

## **Appendix C: Animals**

### *Mammals*

## **SPECIES ACCOUNT: *Antilocapra americana sonoriensis* (Sonoran pronghorn)**

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### ***Species Taxonomic and Listing Information***

**Listing Status:** Endangered, March 11, 1967 (32 FR 4001) and Experimental Population, Nonessential, May 5, 2011 (76 FR 25593).

### **Physical Description**

Pronghorn are proportionately long-legged, small-bodied artiodactyls (hooved mammals having an even number of functional toes), distinguished by large white areas of hair on the rump, sides of face, two bands on the throat, underparts, and partway up the sides of the body. They have slightly curved horns—the males with a single prong projecting forward—and a wooly undercoat overlaid with long, straight, coarse, brittle guard hairs. The animal varies in color from yellowish to tan, but is blackish on the top of the nose. Pronghorn are the swiftest terrestrial mammals in the New World, and can move at speeds of 64 to 72 kilometers per hour (40 to 45 miles per hour) (USFWS 1998).

### **Taxonomy**

Pronghorn are endemic to western North America and are placed within the Family Antilocapridae in Order Artiodactyla, the even-toed ungulates. The genus, *Antilocapra*, contains only the pronghorn species, of which there are four extant subspecies (USFWS 2015). The Sonoran pronghorn is distinguished from the other subspecies by its smaller size, paler color, and distinctive cranial features: 1) Skull narrower in mastoidal, orbital, and zygomatic width. 2) Frontal depression less pronounced. 3) Premaxillae less extended posteriorly along median line. 4) Auditory bullae more flattened, less projecting below level of basioccipital (USFWS 1998).

### **Historical Range**

Sonoran pronghorn historically occurred throughout most of southwestern Arizona, northwestern Sonora, and portions of southeastern California and northeastern Baja California (USFWS 2015).

### **Current Range**

Currently, Sonoran pronghorn only occupy approximately 7.6 percent of their historical range. Their current range is limited to approximately 10,903 square kilometers (km<sup>2</sup>) (4,210 square miles [sq. mi.]), of which 3,781 km<sup>2</sup> (1,460 sq. mi.) are in Mexico and 7,122 km<sup>2</sup> (2,750 sq. mi.) are in the United States. Two of these populations, Kofa and Cabeza Prieta, occur in southwestern Arizona, U.S. The other two populations, Pinacate and Quitovac, occur in northwestern Sonora, Mexico (USFWS 2015).

### **Distinct Population Segments Defined**

No

### **Critical Habitat Designated**

Yes;

### ***Life History***

**Feeding Narrative**

Adult: The Sonoran pronghorn is an herbivore; it consumes various grasses and forbs, and browses on shrubs. Chain fruit cholla is a seasonally important food source for this species. The Sonoran pronghorn has a high bioenergetic requirement; it alternates periods of feeding and resting throughout the day, with continuous feeding in the early morning and late afternoon (NatureServe 2015). Little is known about the feeding competition of this subspecies, although there is a potential for competition from mule deer (*Odocoileus hemionus*) and domestic livestock (USFWS 2015). In winter, the Sonoran pronghorn prefers sparsely vegetated, flat, open spaces that are ideal for swift running and visual detection of predators. In summer, they require denser vegetation that offers thermal cover and moister forage. A mix of these vegetation types is essential to enable the Sonoran pronghorn to use the most suitable vegetation type for the season. The Sonoran pronghorn move nomadically in response to the changes in forage conditions and water availability that result from sporadic rainfall. They require large expanses of contiguous habitat to make these movements and to persist in the harsh desert environment. They also require quality forage, access to water, and a mosaic of suitable vegetation structure (USFWS 2015).

**Reproduction Narrative**

Adult: Sonoran pronghorns are polygamous, and viviparous. They gestate for 250 days. Females generally reach sexual maturity at 16 to 17 months of age; however, records exist of sexual maturation reached at 5 months. Males become sexually mature at 1 year of age. Breeding season is from July through September, and females give birth from February through May. Birthing appears to coincide with spring forage abundance (USFWS 2015). The Sonoran pronghorn reproduce once per year, and give birth to one to two fawns; twins occur more commonly than single births (USFWS 1998). Fawns generally suckle for 4 to 12 weeks. No data are available about the sex ratio, so it is assumed to be 50:50. The lifespan of Sonoran pronghorn in the wild is approximately 10 years. The amount of winter rain and the length of time between winter and summer rains are the most important factors determining fawn survival in Sonoran pronghorn. Adequate moisture is required to provide nutritious forage (USFWS 2015).

**Geographic or Habitat Restraints or Barriers**

Adult: Populations are geographically isolated due to roads and fences; Mexico Highway 2 and Highway 8 and the international boundary fence act as barriers to movement (USFWS 2015).

**Spatial Arrangements of the Population**

Adult: Exist in small herds (USFWS 2015).

**Environmental Specificity**

Adult: Community with key requirements.

**Tolerance Ranges/Thresholds**

Adult: Moderate

**Site Fidelity**

Adult: Low

**Habitat Narrative**

Adult: Sonoran pronghorn are found exclusively in the Lower Colorado River Valley and the Arizona Upland subdivisions of the Sonoran Desertscrub Biome (USFWS 2015). They primarily occupy broad alluvial valleys separated by granite mountains and mesas (NatureServe 2015). In the Lower Colorado River Valley Subdivision, the Sonoran pronghorn most commonly occupy the Creosote-White Bursage series, which is characterized by low, open stands of widely spaced creosotebush (*Larrea tridentata*) and white bursage. In the Arizona Upland Subdivision, Sonoran Pronghorn prefer the Paloverde-Cacti-Mixed Scrub Chain Fruit Cholla vegetation association. Sonoran pronghorn are associated with specific soil associations. Soil association (Gunsight-Rillito-Chuckwalla) is one of the most important explanatory variables for Sonoran pronghorn use areas in a CART model and logistic regression analysis (USFWS 2015). There are four remaining populations, and they are predominantly geographically isolated due to roads and fences. Mexico Highway 2 and the international boundary fence act as barriers to movement between the Pinacate and U.S. subpopulations. Sonoran pronghorn habitat in Mexico is bisected by Highway 8 and associated fences; however, it is unknown how complete a barrier Highway 8 is to pronghorn movements. These barriers were not present historically, and genetic and demographic interchange between pronghorn in Sonora and Arizona likely occurred. Sonoran pronghorn prefer sparsely vegetated, flat, open spaces that are ideal for swift running and visual detection of predators. They require large expanses of a variety of vegetation communities in which to move, based on precipitation, temperature, predation pressure, and high-quality forage availability (USFWS 2015).

***Dispersal/Migration*****Motility/Mobility**

Adult: High

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Migratory

**Dispersal**

Adult: Moderate

**Immigration/Emigration**

Adult: Immigrates/emigrates

**Dependency on Other Individuals or Species for Dispersal**

Adult: No

**Dispersal/Migration Narrative**

Adult: Sonoran pronghorn are a highly mobile subspecies. Their movements correlate with high temperatures and are most likely motivated by the need for the preformed water available in succulent cactus such as chain fruit cholla. Results from aerial telemetry efforts indicate that movements of males range from 30 to 42 kilometers (km) (18.6 to 26.1 miles [mi.]), and that movements of females are within 42 km (26.1 mi.). Although genetic diversity of Sonoran pronghorn is less than other subspecies in the United States, the subspecies is more genetically diverse than the peninsular pronghorn (*Antilocapra americana peninsularis*), and genetic

diversity in Sonoran pronghorn within the United States is not currently low enough to be an immediate concern (USFWS 2015).

**Additional Life History Information**

Adult: Although genetic diversity of Sonoran pronghorn is less than other subspecies in the United States, the subspecies is more genetically diverse than the peninsular pronghorn (*Antilocapra americana peninsularis*), and genetic diversity in Sonoran pronghorn within the United States is not currently low enough to be an immediate concern (USFWS 2015).

***Population Information and Trends*****Population Trends:**

Declining (USFWS 2015)

**Species Trends:**

Declining (USFWS 2015)

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Population Growth Rate:**

Declining (USFWS 2015)

**Number of Populations:**

There are four extant wild populations (USFWS 2015).

**Population Size:**

2017 biennial surveys conducted by AGFD resulted in an estimated 72 and 683 Sonoran pronghorn in the Pinacate and Quitovac populations, respectively (USFWS, 2018).

**Resistance to Disease:**

Low

**Adaptability:**

Low

**Population Narrative:**

Sonoran pronghorn populations show a trend of decline, both within populations and for the subspecies. Currently there are four extant wild populations (USFWS 2015), totaling fewer than 1,000 individuals (NatureServe 2015).

***Threats and Stressors***

**Stressor:** Habitat loss and fragmentation

**Exposure:** Prolonged drought; cattle grazing; fragmentation by fences, railroads, highways, and canals (USFWS 2015).

**Response:** Reduced forage, less dispersal potential.

**Consequence:** Starvation and malnutrition, less genetic diversity.

**Narrative:** Habitat loss and fragmentation have been caused by severe drought, cattle grazing, and fragmentation by railroads, fences, highways, and canals (USFWS 2015).

**Stressor:** Reduced access/reduced availability of water

**Exposure:** Severe drought and habitat fragmentation.

**Response:** Diminished quality of available forage, reduced water availability.

**Consequence:** Death of individuals and greatly reduced populations.

**Narrative:** Reduced access to water and reduced water availability has been caused by both severe drought and habitat barriers. Drought leads to diminished availability of quality forage and water resources, and increases pressure on populations. This pressure has greatly reduced Sonoran pronghorn populations (USFWS 2015).

**Stressor:** Human disturbance

**Exposure:** Railroads, fences, canals, and highways occur throughout the range of the Sonoran pronghorn.

**Response:** Decreased dispersal of populations and decreased access to necessary resources.

**Consequence:** High mortality rates.

**Narrative:** Human disturbance such as railroads, fences, canals, and highways has occurred throughout the range of the Sonoran pronghorn. This limits the dispersal of populations and decreases access to necessary resources, causing high mortality rates in the subspecies (USFWS 2015).

**Stressor:** Agriculture

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Agriculture is a source of habitat loss for the Quitovac population, as the Quitovac area has much private and communal (ejido) land and very few regulations to prevent conversion of habitat to agriculture. Sonoran pronghorn habitat could also be lost to agriculture in the Pinacate population area, but to a much lesser extent than in the Quitovac area. Although the Pinacate population is in a biosphere reserve, some agriculture is allowed. Agriculture is prohibited in the nucleus zone of the bioserve, but in the buffer areas outside the nucleus zone there is less habitat protection and agricultural activities occur on ejidos and private lands (Areas Naturales Protegidas 1995). Agricultural activities are expensive to operate due to the costs of pumping and transporting water, and operate at a subsistence level on ejidos and private farms (Areas Naturales Protegidas 1995). Ejidos and private farms obtain agriculture permits for planting areas that range between 30 and 40 ha (74 and 98 ac) for the production of livestock forage, including alfalfa, wheat, and other forage. However, lack of access to water and dysfunctional hydrologic infrastructure has limited development of agriculture in the bioserve (Areas Naturales Protegidas 1995). The Cabeza Prieta, Kofa, and Saucedo populations occur primarily on public lands and are therefore protected from most major sources of habitat loss (USFWS 2016).

**Stressor:** Climate change

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Our analyses under the ESA include consideration of ongoing and projected changes in climate. The terms “climate” and “climate change” are defined by the Intergovernmental Panel on Climate Change (IPCC). “Climate” refers to the mean and variability of different types of weather conditions over time, with 30 years being a typical period for such measurements, although shorter or longer periods also may be used (Intergovernmental Panel on Climate Change 2007). The term “climate change” thus refers to a change in the mean or variability of one or more measures of climate (e.g., temperature or precipitation) that persists for an extended period, typically decades or longer, whether the change is due to natural variability, human activity, or both (Intergovernmental Panel on Climate Change 2007). Various types of changes in climate can have direct or indirect effects on species. These effects may be positive, neutral, or negative and they may change over time, depending on the species and other relevant considerations, such as the effects of interactions of climate with other variables (e.g., habitat fragmentation) (Intergovernmental Panel on Climate Change 2007). In our analyses, we use our expert judgment to weigh relevant information, including uncertainty, in our consideration of various aspects of climate change (USFWS 2016).

## ***Recovery***

### **Reclassification Criteria:**

Downlisting Criteria: Six criteria must be met to downlist Sonoran pronghorn from endangered to threatened:

1. At least three free-ranging populations are viable. Two of these must be the Cabeza Prieta population and either the Quitovac or Pinacate population. The Recovery Team defines a viable population as one that has less than a 10% probability of extinction over 50 years and a growth rate that is stable or increasing. Furthermore, at least one new population must have been released, in addition to the Kofa subunit (e.g., Saucedo subunit). A population viability analysis (PVA) estimated abundance targets to meet the Recovery Team definition of viability, which is different for each management unit due to different environmental conditions. To be considered viable, a population estimate must meet or exceed the abundance targets and demonstrate a population growth rate that is stable or increasing ( $r \geq 0$ ) for at least five of seven years<sup>1</sup>. Abundance targets for each management unit are estimated from the PVA to be: a) 225 in the Cabeza Prieta Management Unit; b) 150 in the Kofa subunit or a new subunit (Saucedo or other future established subunit); c) 150 in the Pinacate Management Unit; and d) 450 in the Quitovac Management Unit. These population sizes must be estimated by monitoring (i.e., aerial surveys) (USFWS 2016).

2. Within the Cabeza Prieta Management Unit, Pinacate Management Unit, Quitovac Management Unit and the Kofa and Saucedo subunits of the Arizona Reintroduction Management Unit, a minimum of 90% of current Sonoran pronghorn habitat within each unit is retained and contiguous. This Sonoran pronghorn habitat is protected through agency policies, land use regulations and plans, landowner agreements, incentives, and/or other programs and

agreements. The 90% of retained and contiguous Sonoran pronghorn habitat includes key habitat features such as water sources (USFWS 2016).

3. Threats to Sonoran pronghorn habitat quality in three units are stabilized or decreasing as measured by indicators described in Appendix E. Threats must be stabilized or decreased in the three management units that correspond to the three populations that meet the population viability criteria in Recovery Criteria number 1. In particular, the threats of overgrazing; unauthorized routes, roads and trails; invasive plant and animal species threatening Sonoran pronghorn habitat; and spread of shrubby vegetation are minimized through agency policies, land use regulations and plans, landowner agreements, incentives, and/or other programs and agreements (USFWS 2016).

4. Within the Cabeza Prieta Management Unit, Pinacate Management Unit, Quitovac Management Unit, and the Kofa and Saucedo subunits of the Arizona Reintroduction Management Unit, human disturbance is alleviated such that a minimum of 90% of Sonoran pronghorn habitat can be occupied by Sonoran pronghorn (USFWS 2016).

5. Genetic diversity for three populations, as measured by heterozygosity and allelic richness for nuclear DNA markers, has been retained from levels indicated in Culver and Vaughn (2015). These three populations must meet the threshold of viability as described in Downlisting Criterion 1. The minimum level of heterozygosity of any of the three populations must be 49% (i.e., within 20% of the average heterozygosity of population segments (10) estimated by Culver and Vaughn (2015)). The minimum level of allelic richness of any of the three populations must be 1.96 (i.e., within 20% of the average allelic richness of population segments (10) estimated by Culver and Vaughn (2015)) (USFWS 2016).

6. Effective federal, state, tribal, and/or local laws are in place in the recovery conservation units that ensure that killing of Sonoran pronghorn is prohibited or regulated such that viable populations of Sonoran pronghorn can be maintained and are highly unlikely to need the protection of the ESA again (USFWS 2016).

**Delisting Criteria:**

Delisting Criteria: Once the Sonoran pronghorn is downlisted to threatened, the following criteria must be met before the species can be delisted:

1. At least three free-ranging populations are viable. Two of these must be the Cabeza Prieta population and either the Quitovac or Pinacate population. The Recovery Team defines a viable population as one that has less than a 10% probability of extinction over 50 years and a growth rate that is stable or increasing. Furthermore, at least one new population must have been established, in addition to the Kofa subunit (e.g., Saucedo subunit). Established means that the population is stable and is no longer in need of augmentation from a captive breeding program. A PVA estimated abundance targets to meet the Recovery Team's definition of viability, which is different for each management unit due to different environmental conditions. To be considered viable, a population estimate must meet or exceed the abundance targets and demonstrate a population growth rate that is stable or increasing ( $r \geq 0$ ) for at least 10 of 14 years<sup>1</sup>. Abundance targets for each management unit are estimated from the PVA to be: a) 225 in the Cabeza Prieta Management Unit; b) 150 in the Kofa subunit or a new subunit (Saucedo or other future established subunit); c) 150 in the Pinacate Management Unit; and d) 450 in the

Quitovac Management Unit. These population sizes must be estimated by monitoring (i.e., aerial surveys) (USFWS 2016).

2. Delisting criteria 2-6 are the same as downlisting criteria 2-6 (USFWS 2016).

**Recovery Actions:**

- Stabilize, increase, or maintain the number of individuals in existing populations, range wide, where there is adequate habitat (USFWS 2015).
- Assess the quantity and quality of Sonoran pronghorn habitat (USFWS 2015).
- Minimize and mitigate the effects of human disturbance on Sonoran pronghorn (USFWS 2015).
- Identify and address priority Sonoran pronghorn population monitoring needs (USFWS 2015).
- Identify and address priority research needs (USFWS 2015).
- Maintain existing partnerships and develop new partnerships to support Sonoran pronghorn recovery (USFWS 2015).
- Secure adequate funding to implement recovery actions for Sonoran pronghorn (USFWS 2015).
- Practice adaptive management, in which recovery is monitored and recovery tasks are revised by the U.S. Fish and Wildlife Service in coordination with the Recovery Team as new information becomes available (USFWS 2015).

**Conservation Measures and Best Management Practices:**

- Conduct monitoring on Sonoran pronghorn populations, habitat, and threats (USFWS 2015).
- Analyze and share the results of the monitoring (USFWS 2015).
- Compile and exchange information regarding the recovery accomplishments and updates (USFWS 2015).
- Report regularly on Sonoran pronghorn status (USFWS 2015).

**Additional Threshold Information:**

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## **SPECIES ACCOUNT: *Aplodontia rufa nigra* (Point Arena mountain beaver)**

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### ***Species Taxonomic and Listing Information***

**Listing Status:** Endangered; December 12, 1991 (56 FR 64716).

### **Physical Description**

In general, mountain beavers resemble a mix between an overgrown pocket gopher and a muskrat without a tail. Its body is generally stout, compact, and cylindrical; is approximately 30.5 centimeters (12 inches) in length; and weighs 0.45 to 1.8 kilograms (1 to 4 pounds). They have long, stiff whiskers on the nose, small patches of hair at the base of each ear, small eyes and ears, and a cylindrical stump of a tail. Their limbs are short, the forefeet have opposed thumbs, and all digits have long, curved claws. The Point Arena mountain beaver can be distinguished from other mountain beavers by its unique black coloration, a distinct outline of the nasals, and its small size (it is the smallest of the California subspecies) (USFWS 1998).

### **Taxonomy**

The Point Arena mountain beaver was first described as a separate species (*Aplodontia nigra*) in 1914 due to its unique color and anatomical features, and then later as subspecies (*Aplodontia rufa nigra*) of the mountain beaver complex. The Point Arena mountain beaver can be distinguished from other mountain beavers by its unique black coloration, a distinct outline of the nasals, and its small size (it is the smallest of the California subspecies) (USFWS 1998).

### **Historical Range**

Historically, Point Arena mountain beaver colonies were reported in a 12-kilometer (km) (7.5-mile [mi.]) stretch between the town of Point Arena north to Alder Creek, with another record reported another 7 km (4.5 mi.) north at Christianson Ranch (USFWS 1998). All known sites occur within 7.2 km (4.5 mi.) of the coast (USFWS 2009).

### **Current Range**

The Point Arena mountain beaver is known only from an 85-square-km (33-square-mi.) area entirely in western Mendocino County, California. The potential range is considered by the U.S. Fish and Wildlife Service (USFWS) to include the area from 3.2 km (2 mi.) north of Bridgeport Landing to 4.8 km (3 mi.) south of Point Arena, a distance of about 24.6 km (15.3 mi.) (USFWS 2009).

### **Distinct Population Segments Defined**

No

### **Critical Habitat Designated**

No;

### ***Life History***

### **Feeding Narrative**

Adult: The Point Arena mountain beaver is a herbivore with flexible requirements for food, consuming a variety of herbaceous vegetation (including but not limited to succulent plant material, roots, and the bark of woody plants) (USFWS 2009). Many of the plants they consume are unpalatable or toxic to other mammals. Mountain beavers exhibit high activity rates, spending approximately 73 percent of their time foraging or handling food. Mountain beavers are predominantly nocturnal, and collect most of their vegetative material at night within short distances of burrow openings. They may eat vegetation outside of the burrow, but most often collect and cut it to consume later in feeding chambers, often adjacent to the nest chamber. They cut and store about 2.5 times more than they eat, and may stockpile as much as a 2-week supply of forage (USFWS 1998). They reach adult size as yearlings, within a year of becoming independent (Carraway and Verts 1993). Mountain beavers exhibit a behavior called "haystacking," in which they cut bundles of plants and lay them on logs or on the ground to wilt for use later as nesting material, for food storage, or to regulate the moisture content of food by mixing fresh and wilted vegetation (USFWS 1998). Mountain beavers have a simple kidney structure that lacks the anatomical features necessary to concentrate urine effectively, and likely can meet their water needs through metabolic water production and preformed water in food. The inability to concentrate urine and the necessity of a large daily water intake may account for their distribution being limited to areas with rainfall and soil characteristics that promote lush vegetation (USFWS 2009).

#### **Reproduction Narrative**

Adult: Little information is known about the demographics of the Point Arena mountain beaver subspecies—especially information regarding reproduction—and must instead be gleaned from other mountain beaver subspecies (USFWS 1998). Point Arena mountain beavers reach sexual maturity at 2 years of age; however, yearling females ovulate but do not breed. Mountain beavers are monestrous (one litter per year), and display considerable intrapopulation synchrony of estrous timing (Carraway and Verts 1993; USFWS 1998). Nest chambers are constructed belowground, typically deeper than the burrow system, and are lined with an outer layer of coarse vegetation and an inner layer of soft, dry vegetation. The breeding season occurs between late November/early December and May (USFWS 1998), but varies by locale (Carraway and Verts 1993). The gestation period lasts 28 to 30 days, and females nurse their young (USFWS 1998). With litter sizes of two to three young (rarely, four to five) (USFWS 1998) and an expected lifespan of 5 to 6 years (Carraway and Verts 1993), they have a low reproductive capacity of less than 10. Juvenile Point Arena mountain beaver may not be independent until mid-July, perhaps later (USFWS 2009); they are believed to be weaned by 6 to 8 weeks of age, and likely feed on plants carried to the nest by the maternal female (Carraway and Verts 1993).

#### **Geographic or Habitat Restraints or Barriers**

Adult: Gulches and north-facing slopes in narrow coastal valleys (NatureServe 2015; USFWS 1998).

#### **Spatial Arrangements of the Population**

Adult: Clumped; distribution limits are associated with rainfall and soil conditions that promote lush vegetation and high humidity (observed as high as 100 percent) in burrows (USFWS 2009).

#### **Environmental Specificity**

Adult: Narrow; specialist.

**Tolerance Ranges/Thresholds**

Adult: Limited ability to thermoregulate; when exposed to high ambient temperatures, individuals reduce their activity or attempt to escape, and begin panting/salivating. Lethal body temperature is 42 degrees Celsius (108 degrees Fahrenheit). Bright light, warmth, panic, or other conditions may induce a narcoleptic effect (USFWS 1998).

**Site Fidelity**

Adult: High; at least one Point Arena mountain beaver location (at Alder Creek) appears to have been occupied since 1913 (USFWS 2009).

**Dependency on Other Individuals or Species for Habitat**

Adult: Mountain beavers exhibit a "contagious" distribution, in which the presence of one animal in a given area attracts the settlement of others (USFWS 1998).

**Habitat Narrative**

Adult: Point Arena mountain beavers are found in gulches and north-facing slopes in narrow coastal valleys, inhabiting riparian and coastal scrub habitats, where they burrow in moist areas with well-drained soils, a cool thermal regime, abundant food supply, and high percent cover of small-diameter woody material (NatureServe 2015, USFWS 1998). Individuals are able to meet their water needs through metabolic water production, and therefore do not require access to free water (USFWS 2009). Although individuals may share the same contagious and interconnected burrow systems, they do not live in colonies, and are rarely social. Typical burrow territories do not exceed 25 m (80 ft.) and can extend for more than 100 m (330 ft.) in one direction (USFWS 1998). Tunnels run within 0.3 m (1 ft.) of the surface, but can descend up to 1 to 1.5 m (3 to 5 ft.) (USFWS 1998). Mountain beavers exhibit a "contagious" distribution, in which the presence of one animal in a given area attracts the settlement of others (USFWS 1998). The Point Arena mountain beaver has a limited ability to thermoregulate; when exposed to high ambient temperatures, individuals reduce their activity or attempt to escape, and begin panting/salivating. Lethal body temperature is 42 degrees Celsius (108 degrees Fahrenheit). Bright sunlight, warmth, panic, or other conditions may induce a narcolepsy effect, causing individuals to fall asleep (USFWS 1998).

***Dispersal/Migration*****Motility/Mobility**

Adult: Low

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Low

**Immigration/Emigration**

Adult: Emigrates

**Dependency on Other Individuals or Species for Dispersal**

Adult: Mountain beavers exhibit a "contagious" distribution, in which the presence of one animal in a given area attracts the settlement of others. However, they exhibit little social interaction and are not considered colonial, primarily exhibiting solitary behavior except during the short breeding period (USFWS 1998).

**Dispersal/Migration Narrative**

Adult: Mountain beavers exhibit a "contagious" distribution, in which the presence of one animal in a given area attracts the settlement of others, but juvenile dispersal is primarily conducted through excavation in a burrow system. However, some overland migration has been observed, which requires habitat connectivity between suitable, undisturbed habitats. Point Arena mountain beavers are otherwise nonmigratory, and exhibit low dispersal and mobility (USFWS 1998), ranging up to 564 m (1,850 ft.) (USFWS 2009). Point Arena mountain beavers exhibit little social interaction and, although they co-inhabit burrow systems, are not considered colonial (USFWS 1998). Information concerning gene flow, dispersal barriers, dispersal corridors, and potential dispersal distance is limited, and more research is needed (USFWS 2009). However, based on what is known regarding occupied sites and regional land-use patterns, suitable habitat is likely highly fragmented by roads, agricultural use, and residential development (USFWS 2009).

**Additional Life History Information**

Adult: Juvenile dispersal is primarily through excavation in a burrow system, although some overland migration has been seen (USFWS 1998). Maximum dispersal distances for other mountain beaver subspecies are reported as far as 564 m (1,850 ft.) (USFWS 2009).

***Population Information and Trends*****Population Trends:**

Decreasing

**Species Trends:**

Decreasing

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Population Growth Rate:**

Slow

**Number of Populations:**

There are 26 separate populations, which have since been preliminarily aggregated into 14 geographic groups (USFWS 1998; USFWS 2009).

**Population Size:**

Population size is estimated between 200 to 500 (USFWS 1998) to as many as 1,000 individuals (NatureServe 2015).

**Resistance to Disease:**

Unknown (USFWS 1998)

**Adaptability:**

Low

**Additional Population-level Information:**

Historical records of the Point Arena mountain beaver are scarce (USFWS 1998). At the time of the Recovery Plan (USFWS 1998), no data were available on the density of Point Arena mountain beaver populations. Based on population studies of other mountain beaver subspecies (*A. rufa* ssp.), it is likely that many of these small Point Arena mountain beaver sites with fewer than 20 active burrow openings are occupied by only one or two individuals. To date, a total of 262 individual records (points) with burrow systems have been mapped range-wide. The current status of approximately 80 percent of these occurrences is unknown, because they occur on private lands that have not been visited in recent years, and may have been or may be subject to future development (USFWS 2009).

**Population Narrative:**

Historical records of the Point Arena mountain beaver are scarce (USFWS 1998). In 1998, 26 separate populations of Point Arena mountain beaver had been identified (USFWS 1998), with an estimated 200 to 500 (USFWS 1998) to as many as 1,000 individuals (NatureServe 2015). These populations have been preliminarily aggregated into 14 geographic groups. At the time of the Recovery Plan (USFWS 1998), no data were available on the density of Point Arena mountain beaver populations. Based on population studies of other mountain beaver subspecies (*A. rufa* ssp.), it is likely that many of these small Point Arena mountain beaver sites with fewer than 20 active burrow openings are occupied by only one or two individuals. To date, a total of 262 individual records (points) with burrow systems have been mapped range-wide. The current status of approximately 80 percent of these occurrences is unknown, because they occur on private lands that have not been visited in recent years, and may have been or may be subject to future development (USFWS 2009). However, given the species' sensitivity to disturbance (crushing of burrows/vegetation) or catastrophic events, the unlikelihood of immigration/emigration, and the limited genetic diversity, the species shows a low resilience to withstand stochastic events, has a low representation to adapt to changing environmental conditions across the landscape, a low redundancy to withstand catastrophic events, a low resistance to disease, and low adaptability.

**Threats and Stressors**

**Stressor:** Present or threatened destruction, modification, or curtailment of habitat or range

**Exposure:** Direct/indirect

**Response:** Loss/degradation of habitat.

**Consequence:** Degradation of habitat, reduction of quality/quantity of breeding/foraging/upland habitat.

**Narrative:** Urban development, land conversion, and associated activities have led to trash dumping, which attracts predators such as feral and nonferal house pets. Housing developments and associated roads contribute to habitat loss and fragmentation (USFWS 1998; USFWS 2009).

**Stressor:** Predation

**Exposure:** Direct/indirect

**Response:** Mortality

**Consequence:** Higher susceptibility to mortality/extirpation.

**Narrative:** Predators include coyote (*Canis latrans*), bobcat (*Felis rufus*), long-tailed weasel (*Mustela frenata*), spotted skunk (*Spilogale gracilis*), striped skunk (*Mephitis mephitis*), great-horned owl (*Bubo virginianus*), and raptors, as well as domestic and feral dogs and cats (USFWS 1998; USFWS 2009).

**Stressor:** Catastrophic events

**Exposure:** Direct/indirect

**Response:** Mortality, loss/degradation of habitat.

**Consequence:** Mortality, degradation of habitat, reduction of quality/quantity of breeding/foraging/upland habitat.

**Narrative:** Because Point Arena mountain beavers have a clumped and fragmented distribution, they are more vulnerable to localized catastrophic events like storms, fire, flooding, beach erosion, landslides, disease, or prolonged drought (USFWS 1998; USFWS 2009).

**Stressor:** Loss of riparian habitat

**Exposure:** Direct/indirect

**Response:** Loss/degradation of habitat; increased predator activity.

**Consequence:** Degradation or reduction of habitat; increased predation/mortality.

**Narrative:** Unauthorized destruction of riparian habitat through heavy equipment use, vegetation cutting, and/or vegetation burning leads to habitat loss and increased predator activity (USFWS 1998; USFWS 2009).

**Stressor:** Livestock grazing

**Exposure:** Direct/indirect

**Response:** Loss/degradation of habitat.

**Consequence:** Degradation of habitat, reduction of quality/quantity of breeding/foraging/upland habitat.

**Narrative:** Livestock grazing has substantially reduced the extent of historical coastal scrub habitat. Presently, livestock grazing leads to trampling of vegetation, burrows, and runways (USFWS 1998; USFWS 2009).

**Stressor:** Transportation and utility facilities

**Exposure:** Direct/indirect

**Response:** Loss/degradation of habitat.

**Consequence:** Mortality, impeded or eliminated dispersal from natal areas.

**Narrative:** The installation of utilities such as underground fiber optics projects causes noise, vibration, and physical impacts to mountain beaver habitat. New roadways lead to habitat loss; new and existing roadways create higher mortality rates and impede or eliminate the ability of young mountain beavers to disperse from natal areas (USFWS 1998; USFWS 2009).

**Stressor:** Recreation

**Exposure:** Direct/indirect

**Response:** Loss/degradation of habitat.

**Consequence:** Degradation of habitat, reduction of quality/quantity of breeding/foraging/upland habitat.

**Narrative:** Mountain beavers' semi-fossorial habits and anatomy suggest high sensitivity to ground vibration and noise (USFWS 2009). Recreational activities such as camping lead to off-trail exploration and subsequent trampling of vegetation, burrows, and runways. Trail construction and associated facilities (parking lots, interpretive centers, signage) also increase levels of human disturbance and habitat loss (USFWS 1998; USFWS 2009).

**Stressor:** Pest control

**Exposure:** Direct/indirect

**Response:**

**Consequence:** Mortality

**Narrative:** Direct and indirect (where mountain beavers are mistaken for pests like gophers) pest control programs lead to mortality through lethal chemicals (USFWS 1998; USFWS 2009).

**Stressor:** Exotic plants

**Exposure:** Indirect

**Response:** Loss/degradation of habitat.

**Consequence:** Degradation of habitat, reduction of quality/quantity of breeding/foraging/upland habitat.

**Narrative:** Several exotic plant species have become established and have spread rapidly, reducing the quality and quantity of suitable habitat, displacing native vegetation, and completely covering burrow openings to the point that they are no longer occupied. These plants include German ivy (*Senecio mikanioides*), European beachgrass (*Ammophila arenaria*), and ice plant (*Carpobrotus edulis*). The effects of other exotic plants are still unknown (USFWS 1998; USFWS 2009).

**Stressor:** Small population size

**Exposure:** Indirect

**Response:** Decreased ability to respond to changing conditions.

**Consequence:** Reduction in population numbers, increased genetic effects of population bottleneck, higher susceptibility to mortality/extirpation.

**Narrative:** Point Area mountain beaver population numbers may be so low that the effects of inbreeding among closely related individuals could result in an increase in deleterious genes in the population. Moreover, small populations are subject to the effects of genetic drift, the random decline in genetic variation that can occur in small populations. These limit the flexibility of a population to respond to environmental change (USFWS 1998; USFWS 2009).

**Stressor:** Habitat fragmentation

**Exposure:** Direct/indirect

**Response:** Loss/degradation of habitat; genetic isolation.

**Consequence:** Reduced population size; increase probability of genetic drift and inbreeding depression.

**Narrative:** Habitat fragmentation can increase the genetic isolation among populations of mountain beaver, and can reduce population size, thereby increasing the probability of genetic

drift and inbreeding depression. This may result in less variable and adaptable populations of mountain beaver (USFWS 1998; USFWS 2009).

**Stressor:** Global warming

**Exposure:** Direct/indirect

**Response:** Mortality, loss/degradation of habitat.

**Consequence:** Mortality, degradation of habitat, reduction of quality/quantity of breeding/foraging/upland habitat.

**Narrative:** The mountain beaver's unique physiology may make them especially vulnerable to increased drought conditions and temperature. In addition, coastal damage from flooding and extreme storm events such as heavy surf and wind-driven waves could lead to increasing coastal erosion, flooding, and faster cliff retreat, as well as direct mortality to individuals and populations along the coastal bluff edges (USFWS 2009).

### ***Recovery***

#### **Reclassification Criteria:**

At least 16 populations are protected from human-caused disturbance in perpetuity. Each population shall contain at least 20 hectares (ha) (49 acres [ac.]) of suitable habitat, of which at least 10 ha (25 ac.) are occupied habitat.

These populations shall have a mean density of at least four Point Arena mountain beavers per ha (1.6 per ac.) of occupied habitat, unless new data show that a lower density is healthy and stable.

All 16 populations are stable (i.e., no more than a 25 percent change in estimated population size from highest to lowest value) or increasing for a period of 10 years (following attainment of criterion #1), as documented through establishment and implementation of a scientifically acceptable population monitoring program.

The amount of additional habitat needed for population interconnectivity, travel, and dispersal habitat has been determined.

Sufficient information is available to permit adaptive management prescriptions, and any management actions necessary to ensure the continued success of these populations (in criterion #2) have been fully implemented.

#### **Delisting Criteria:**

Thirty populations are protected from disturbance in perpetuity. Each population shall contain at least 20 ha (49 ac.) of suitable habitat, of which at least 10 ha (25 ac.) are occupied habitat.

These populations shall have a mean density of a least four Point Arena mountain beavers per ha (1.6 per ac.) of occupied habitat, unless new data show that a lower density is healthy and stable.

All 30 populations are stable (i.e., no more than a 25 percent change in estimated population size from highest to lowest value) or increasing for a period of at least 15 years (following

attainment of criterion #1), as documented through establishment and implementation of a scientifically acceptable population monitoring program.

Additional habitat needed for population interconnectivity, travel, and dispersal has been protected and is being managed appropriately.

Adaptive management prescriptions have been determined and implemented for all populations.

**Recovery Actions:**

- Protect known populations (USFWS 1998).
- Protect suitable habitat, buffers, and corridors (USFWS 1998).
- Develop management plans and guidelines (USFWS 1998).
- Gather biological and ecological data necessary for conservation of the subspecies (USFWS 1998).
- Determine feasibility of, and need for, relocation (USFWS 1998).
- Monitor existing populations and survey for new ones (USFWS 1998).
- Establish an outreach program (USFWS 1998).
- No formal guidelines containing conservation measures have been developed for this species. The Point Arena Mountain Beaver (*Aplodontia rufa nigra*) 5-Year Review (2009) provides a number of recommendations for actions over the next 5 years; including:
- Continue research to characterize the genetic diversity within and among individual occurrences (USFWS 2009).
- Continue to monitor the established survey grids to estimate abundance, survival rates, and recruitment (USFWS 2009).
- Identify and map suitable habitat, potential dispersal corridors, dispersal barriers, and restoration areas (USFWS 2009).
- Delineate appropriate conservation units for management based on data on gene flow, dispersal barriers, and potential dispersal distances (USFWS 2009).
- Develop and implement a noninvasive sampling program to monitor range-wide trends in abundance and distribution. Also develop a sampling plan to monitor habitat quantity, quality, and threats (USFWS 2009).
- Once sufficient information is gathered, revise the current recovery plan to include updated recovery criteria and tasks (USFWS 2009).
- Identify key areas for protection, such as conservation easements and acquisition; this will enable the USFWS to work with partners when opportunities arise (USFWS 2009).
- Identify sites for vegetation management, such as exotic plant removal or livestock exclosures (USFWS 2009).

***Conservation Measures and Best Management Practices:***

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***Additional Threshold Information:***

- 
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**References**

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## SPECIES ACCOUNT: *Bison bison athabasca* (Wood Bison)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Experimental population/considered extirpated from U.S.; 06/02/1970; Alaska Region (R7) (USFWS, 2016)

### **Physical Description**

Wood bison is the largest native extant terrestrial mammal in North America (Reynolds et al. 2003, p. 1015). Average weight of mature males (age 8) is 910 kilograms (kg) (2,006 pounds (lb)) and the average weight of mature females (age 13) is 440 kg (970 lb) (Reynolds et al. 2003, p. 1015). They have a large triangular head, a thin beard and rudimentary throat mane, and a poorly demarcated cape (Boyd et al. 2010, p. 16). In addition, the highest point of their hump is forward of their front legs; they have reduced chaps on their front legs; and their horns usually extend above the hair on their head (Boyd et al. 2010, p. 16). These physical characteristics distinguish them from the plains bison (Reynolds et al. 2003, p. 1015; Boyd et al. 2010, p. 16) (USFWS, 2012).

### **Taxonomy**

Wood bison (*Bison bison athabasca*) belongs to the family Bovidae, which also includes cattle, sheep, and goats. Debate over the generic name *Bison* continues with some authorities using *Bos* and others using *Bison* depending on the methodology used to determine relationships among members of the tribe Bovini (Asian water buffalo, African buffalo, cattle and their wild relatives, and bison) (Boyd et al. 2010, pp. 13–15). In this discussion, we will use *Bison*, which is consistent with “Wild Mammals of North America” (Reynolds et al. 2003, p. 1010), “Mammal Species of the World” (Wilson and Reeder 2005, p. 689), and the Wood Bison Recovery Team (Gates et al. 2001, p. 25). Wood bison was first described as a subspecies in 1897 (Rhoads 1897, pp. 498–500). One other extant bison subspecies, the plains bison (*B. b. bison*), occurs in the United States and Canada. Based on the historical physical separation and quantifiable behavioral, morphological, and phenological (appearance) differences between the two subspecies, the scientific evidence indicates that subspecific designation is appropriate (van Zyll de Jong et al. 1995, p. 403; FEAP 1990, p. 24; Reynolds et al. 2003, p. 1010; Gates et al. 2010, pp. 15–17) (USFWS, 2012).

### **Historical Range**

The exact extent of the original range of wood bison cannot be determined with certainty based on available information, but was limited to North America (Gates et al. 2001, p. 11). However, historically, the range of the wood bison was generally north of that occupied by the plains bison and included most boreal regions of northern Alberta, northeastern British Columbia east of Cordillera, a small portion of northwestern Saskatchewan, the western Northwest Territories south and west of Great Slave Lake, the Mackenzie River Valley, most of The Yukon Territory, and much of interior Alaska (Reynolds et al. 2003, pp. 1011–1012). Skinner and Kaisen (1947, pp. 158, 164) suggested that the prehistorical U.S. range extended from Alaska to Colorado, and Stephenson et al. (2001, p. 140) concluded that wood bison were present within the boundaries of what is now defined as Alaska until their disappearance during the last few hundred years. Currently, there is a wild population neither in Alaska nor in the continental United States (Harper and Gates 2000, p. 917; Stephenson et al. 2001, p. 140) (USFWS, 2012).

**Critical Habitat Designated**

No;

***Life History*****Feeding Narrative**

Adult: The foraging habitats most favored by wood bison are grass and sedge meadows occurring on alkaline soils. These meadows are typically interspersed among tracts of coniferous forest, stands of poplar or aspen, bogs, fens, and shrublands. Meadows typically represent 5 to 20 percent of the landscape occupied by wood bison (Larter and Gates 1991a, p. 2682; Gates et al. 2001, p. 23). Wet meadows are rarely used in the summer, probably because of the energy required to maneuver through the mud, but they are used in late summer when they become drier, and in the winter when they freeze (Larter and Gates 1991b, pp. 133, 135; Strong and Gates 2009, p. 438). Biology Because wood bison can thrive on coarse grasses and sedges, they occupy a niche within the boreal forest that is not utilized by other northern herbivores such as moose or caribou (Gates et al. 2001, p. 25). Several studies indicate that wood bison prefer sedges (*Carex* spp.), which can comprise up to 98 percent of the winter diet (Reynolds et al. 1978, p. 586; Smith 1990, p. 88; Larter and Gates 1991a, p. 2679; Fortin et al. 2003, pp. 224–225). Seasonally, other important diet items include grasses, willow, and lichen (Reynolds et al. 1978, p. 586; Smith 1990, p. 88; Larter and Gates 1991a, pp. 2680–2681; Fortin et al. 2003, pp. 224–225) (USFWS, 2012).

**Reproduction Narrative**

Adult: The wood bison's breeding season is from July to October. The age of first reproduction depends on nutritional condition and disease status, and is therefore variable (Gates et al. 2010, p. 49). Females typically produce their first calf when they are 3 years old and may be reproductively successful up to age 20 (Wilson et al. 2002, p. 1545). Although capable of reproduction at age 2, males typically do not participate in the rut until they are 5 or 6, and reproductive success is at its maximum between ages 7 and 14 (Wilson et al. 2002, pp. 1538, 1544). Bison have a polygynous mating system, in which one male mates with several females (Wilson et al. 2002, p. 1538). When habitat is adequate and there are no other limiting factors such as disease and predation, wood bison populations have expanded exponentially (FEAP 1990, pp. 34–35; Gates and Larter 1990, p. 233). Consequently, newly introduced populations have the capacity to grow quickly, as demonstrated by the Mackenzie herd (Gates and Larter 1990, p. 235) (USFWS, 2012).

**Tolerance Ranges/Thresholds**

Adult: Moderate (inferred from USFWS, 2012)

**Site Fidelity**

Adult: Moderate (inferred from USFWS, 2012)

**Habitat Narrative**

Adult: The foraging habitats most favored by wood bison are grass and sedge meadows occurring on alkaline soils. These meadows are typically interspersed among tracts of coniferous forest, stands of poplar or aspen, bogs, fens, and shrublands. Meadows typically represent 5 to 20 percent of the landscape occupied by wood bison (Larter and Gates 1991a, p. 2682; Gates et al. 2001, p. 23). Wet meadows are rarely used in the summer, probably because of the energy

required to maneuver through the mud, but they are used in late summer when they become drier, and in the winter when they freeze (Larter and Gates 1991b, pp. 133, 135; Strong and Gates 2009, p. 438) (USFWS, 2012). Moderate ecological integrity of the community, tolerance ranges and site fidelity are inferred based on the habitat this species inhabits and its ability to move long distances in search of food.

### ***Dispersal/Migration***

#### **Motility/Mobility**

Adult: High (USFWS, 2012)

#### **Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory (USFWS, 2012)

#### **Dispersal**

Adult: Released individuals have dispersed as far as 250 km (USFWS, 2012)

#### **Dispersal/Migration Narrative**

Adult: Free-ranging wood bison roam extensively with annual maximum traveling distance from each individual's center-of-activity averaging from 45 to 50 kilometers (km) (28 to 31 miles (mi)) (Chen and Morley 2005, p. 430). However, some captive animals released into the wild have traveled over 250 km (155 mi) (Gates et al. 1992, pp. 151–152). Herds are fluid, and individuals interchange freely (Fuller 1960, p. 15; Wilson et al. 2002, p. 1545). Wood bison travel between favored foraging habitats along direct routes including established trails, roads, river corridors, and transmission lines (Reynolds et al. 1978, p. 587; Mitchell 2002, p. 50). Bison are also powerful swimmers and will cross even large rivers such as the Peace, Slave, Liard, and Nahanni to reach forage, provided that there are low banks for entry and exit (Fuller 1960, p. 5; Mitchell 2002, pp. 32, 50; Larter et al. 2003, pp. 408– 412) (USFWS, 2012).

### ***Population Information and Trends***

#### **Population Trends:**

Unknown (No information found)

### ***Threats and Stressors***

**Stressor:** Loss of foraging habitat: Fire suppression (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Wood bison depend on a landscape that includes sufficient grasslands and meadows for foraging habitat (Larter and Gates 1991b, p. 133). It appears that primarily through fire suppression, there was an overall loss of meadow habitat in Canada through the 1900s. More intensive fire management began in Canada in the early 1900s, with the philosophy that fire was destructive and should be eliminated to protect property and permit proper forest management (Stocks et al. 2003, p. 2). However, wildfire is an integral component of boreal forest ecology (Weber and Flannigan 1997, p. 146; Rupp et al. 2004, p. 213; Soja et al. 2007, p. 277). Without fire, trees encroach on meadows and eventually the meadow habitat is lost and replaced by

forest. Fire alone, or in combination with grazing, can facilitate the conversion and maintenance of grasslands (Lewis 1982, p. 24; Chowns et al. 1997, p. 205; Schwarz and Wein 1997, p. 1369). Burning by Native groups within the range of wood bison was apparently a common practice through the 1940s outside WBNP but ended within the park when it was established in 1922 (Lewis 1982, pp. 22–31; Schwarz and Wein 1997, p. 1369). An examination of aerial photographs taken at WBNP over time showed that a semi-open grassland that covered about 85 ha (210 ac) in 1928 supported a grassland of only 3 ha (7.4 ac) in 1982 (Schwarz and Wein 1997, p. 1369). In addition, a number of sites previously identified as prairie are now dominated by trembling aspen (Schwarz and Wein 1997, p. 1369). Although not quantified, it is likely that because of fire suppression and forest encroachment on meadows, there was a net loss of suitable open meadow habitat for wood bison throughout their range through about 1990. More recently, several factors may be counteracting the loss of open meadow habitat including controlled burns, timber harvest, oil and gas development, agricultural development, and the effects of climate change, as discussed below (USFWS, 2012).

**Stressor:** Loss of foraging habitat: Controlled burns (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Controlled burns have been implemented since 1992 in wood bison habitat in the Northwest Territories to increase meadow habitat (Chowns et al. 1997, p. 206). Approximately 4,400 to 26,900 ha (10,873 to 66,471 ac) were burned from 1992 to 1997, with some sites being burned up to three times (Chowns et al. 1997, pp. 206–207). In addition, lightning fires burned 300,000 ha (741,316 ac), or almost 20 percent of the wood bison range in this area, from 1994 to 1996 (Chowns et al. 1997, p. 209). Plants favored by bison were more abundant in unburned areas and in meadows that had burned only once (Quinlan et al. 2003, p. 348), indicating that prescribed burns must be used judiciously to be effective in creating foraging habitat for wood bison. A study of vegetation recovery and plains bison use after a wildfire near Farewell, Alaska (Campbell and Hinkes 1983, p. 18), showed that grass and sedge-dominated communities increased from 38 percent to approximately 97 percent of the study area. Plains bison use also increased in subsequent years after the fire, and winter distribution of the Farewell herd expanded due to fire-related habitat changes (Campbell and Hinkes 1983, pp. 18–19). Because sedges are important winter forage for wood bison, the amount of such habitat has a major influence on herd size. Newly created habitats will be used by wood bison when these habitats are contiguous with existing summer or winter ranges (Campbell and Hinkes 1983, p. 20). In summary, studies that have looked at the exclusion of fire or the effect of wildfire on wood bison habitat have concluded that fire is a necessary component of the landscape to maintain clearings and create conditions that favor forage preferred by wood bison. Controlled burns can have the same effect as wildfire by creating openings in the forest. However, repeated burns in the same location can be detrimental to creating suitable forage (USFWS, 2012).

**Stressor:** Loss of foraging habitat: Timber harvest (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The volume of timber logged in Canada rose 50 percent from 1970 to 1997; in Alberta, the logging rate increased 423 percent, from 3.4 to 17.8 million meters (m)<sup>3</sup> (120 to 628 million feet (ft)<sup>3</sup>) per year during the same time (Timoney and Lee 2001, p. 394). These values are

conservative because forests logged on private land and those harvested on government land after fire, insect outbreaks, or disease may go unrecorded (Timoney and Lee 2001, p. 395). The primary method of harvest is clearcutting (Timoney and Lee 2001, p. 394). Compared to a closed canopy forest, clearcuts improve the amount of suitable habitat available to wood bison because they create openings and increase the amount of summer forage available. However, the quantity and quality of forage is less than what is found in preferred wood bison foraging habitats, and the increased productivity seen after a clearcut is not maintained, as woody vegetation becomes more dominant over time (Redburn et al. 2008, p. 2233). In addition, clearcuts do not provide adequate winter forage because wood bison's preferred food, sedges, typically do not colonize these areas. Clearcutting is not being used as a management tool to increase wood bison habitat currently, and whatever gains in habitat that have occurred from clearcutting are most likely low. In summary, although timber harvest occurs throughout the range of wood bison, it is unclear to what extent it is creating suitable habitat. Clear cuts can increase summer forage, but they need to be in proximity to sedge meadows (wintering habitat) to increase the annual carrying capacity for wood bison, and the openings created by the clear cuts must be maintained over time. Although timber harvest has the potential to increase the amount of suitable habitat for wood bison, the amount that may have been created is most likely low and is undocumented (USFWS, 2012).

**Stressor:** Loss of foraging habitat: Oil and gas development (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Oil and gas exploration and production in Canada has increased in the last 20 years (Timoney and Lee 2001, pp. 397–398). Seismic mapping to determine the oil and gas reserves below the surface involves cutting paths 5 to 8 m (16.4 to 26 ft) wide across the landscape. The seismic lines become persistent features in the forested boreal landscape (Lee and Boutin 2006, p. 249). Approximately 70 percent of landscape disturbance for nonrenewable resource extraction in Alberta is due to seismic lines (Timoney and Lee 2001, p. 397). There are an estimated 1.5 to 1.8 million km (932,000 to 1,100,000 mi) of seismic lines in Alberta (Timoney and Lee 2001, p. 397). Lee and Boutin (2006, p. 244) found that only 8.2 percent of seismic lines in Alberta's northeastern forested stands recovered to greater than 50 percent woody vegetative cover after 35 years, and 64 percent of these seismic lines maintained a cover of grasses and herbs. In terms of creating forest openings, more suitable foraging habitat, and linear paths, seismic lines may be beneficial for wood bison. However, because vehicular routes were established in 20 percent of the seismic lines, they also become corridors for offroad vehicles, recreationalists, and poachers (Trombulak and Frissell 2000, pp. 19–20; Timoney and Lee 2001, p. 400; Lee and Boutin 2006, p. 244). Although wood bison are known to occupy linear clearings such as roads, and seismic lines have increased dramatically within their range, potentially creating suitable habitat, we do not have documentation of wood bison use of this type of habitat (USFWS, 2012).

**Stressor:** Loss of foraging habitat: Agricultural development (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Agricultural development, including plains bison ranching, is the least compatible land use for wood bison recovery (Harper and Gates 2000, p. 921). Loss of habitat for agricultural

production is a threat to wood bison because of the large areas involved. Agricultural development near Fort St. John and Fort Nelson, British Columbia, has reduced habitat for wood bison, and continuing expansion of agriculture in the north will further limit the ability to meet population recovery objectives (Harper and Gates 2000, p. 921). Based on a conservative estimate of historical habitat only in Canada, Gates et al. (1992, p. 154) estimated that human activities and development exclude wood bison from approximately 34 percent of their historic range. When an updated Canadian historical range (Stephenson et al. 2001, p. 136) and the Alaskan historical range are included in the calculation, the amount of compromised habitat drops to approximately 16.5 percent if only Canada is considered, and 13 percent if the historical habitat in Canada and Alaska are combined (Stephenson 2010, pers. comm.). Sanderson et al. (2002, pp. 894–896; 2008, p. 257) found that the level of human influence in the range occupied by wood bison to be extremely low (less than 10 percent). Although human development and influence is very low over the majority of range occupied by wood bison, we assume that because of human population growth, increased commercial production of plains bison, and increased agricultural production, there will be continued loss of suitable wood bison habitat into the foreseeable future (USFWS, 2012).

**Stressor:** Climate change(USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** One potential effect of climate change may be an increase in anthrax outbreaks because of increased summer air temperatures. Between 1962 and 1993, nine anthrax outbreaks were recorded in northern Canada, killing at least 1,309 wood bison (Dragon et al. 1999, p. 209). Additional outbreaks continued to occur through at least 2010 (GNT 2010, p. 9). Wood bison appear most susceptible to outbreaks when they are stressed, including heat stress and high densities of biting insects (Dragon et al. 1999, p. 212; Gates et al. 2010, p. 28). In addition, if climate change leads to widespread or intense drought, there could be changes in the quality and availability of forage that may cause animals to concentrate around available food and water. These factors could contribute to stress levels and increase susceptibility to anthrax (Dragon et al. 1999, p. 212; Gates et al. 2010, p. 28). Although isolated anthrax outbreaks occur currently, it is possible that outbreaks may become more frequent, become more widespread, or affect a greater number of animals in the future. Thus far, anthrax outbreaks have occurred sporadically when the necessary factors have come together to affect portions of one herd at a time. Anthrax is not currently having a population-level effect, and we do not have enough information to predict with confidence if anthrax will have a population-level effect on wood bison in the future as a result of climate change.

**Stressor:** Disease (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Bovine brucellosis and bovine tuberculosis are listed as threats to this species. Herds are examined annually to insure they are disease free (USFWS, 2012).

**Stressor:** Predation (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Wolf predation can be a significant limiting factor for diseased populations of wood bison (Reynolds et al. 1978, p. 581; Van Camp 1987, p. 25). Wood bison were the principle food of two wolf packs from 1975 to 1977 in the Slave River lowlands (Van Camp 1987, pp. 29, 32). Of the adult and subadult wood bison that died in 1976–1977, wolves killed 31 percent; however, hunters killed 39.3 percent (Van Camp 1987, p. 33). Joly and Messier (2004, p. 1173) found that productivity of the diseased WBNP herd was insufficient to offset losses to both predation and disease, but that in the absence of either factor, positive population growth was possible. Presence of disease likely increased the killing success of wolves through bison debilitation (Joly and Messier 2004, p. 1174). Wood bison evolved with wolves, and we have no data showing that predation by wolves is limiting the recovery of any of the disease-free herds or would cause the extirpation of a herd (ADF&G 2007, p. 98) (USFWS, 2012).

**Stressor:** Accidental mortality (USFWS, 2012)

**Exposure:****Response:****Consequence:**

**Narrative:** Because of continued or increased resource development, tourism, and offroad vehicle use, it is anticipated that mortality from collisions with vehicles will be a source of individual mortality for several populations. Because mortality from road collisions represents a small portion of the total subspecies population, and efforts are made to reduce bison/highway conflicts, this source of mortality is not expected to have a significant impact at the subspecies population level. Spring flooding in the PeaceAthabasca River Delta in 1958, 1961, and 1974 killed approximately 500, 1,100, and 3,000 wood bison, respectively (Reynolds et al. 2003, p. 1029). Autumn flooding in the same area in 1959 killed an estimated 3,000 wood bison (Reynolds et al. 2003, p. 1029). This region is within WBNP where the diseased herds reside. Most likely a small number of animals drown each year when caught by floods or when they break through ice (Soper 1941, p. 403; Larter et al. 2003, p. 411). Large drowning events have not been documented from other rivers, and no large mortality events have been documented in recent years. Drowning is also recognized as a cause of mortality in the Chitek Lake, Mackenzie, and Nahanni herds (Larter et al. 2003, p. 411). Because mortality due to drowning typically affects only a portion of a herd and herd sizes are increasing (see Table 1, above), drowning does not appear to be having a population-level effect on wood bison. Although wood bison are hardy and very cold tolerant (Gates et al. 2010, p. 24), above-average snowfall, long periods of sub-zero temperatures, and midwinter thaws followed by freezing can cause mortality. Such severe winter conditions reduce forage availability (Reynolds et al. 2003, p. 1030). Rain-on-snow events can also form an ice layer that creates a barrier to forage for herbivores (Putkonen 2009, p. 221). Freezing rain in autumn that causes ground-fast ice to form before snow cover accumulates, ice layering in the snow cover, crusting of the snow, and the formation of ground-fast ice in spring increase the energy required to obtain forage or make forage unobtainable (Gunn and Dragon 2002, p. 58). Soper (1941, pp. 403–404) recounts several stories in which excessive snowfall caused mass mortalities of wood bison, and Van Camp and Calef (1987, p. 23) report that 33 percent of the diseased wood bison herd in the Slave River lowlands was lost during the severe winter of 1974–1975. Starvation in bad winters is recognized as a source of mortality for wood bison in the Chitek Lake herd. We have no information indicating that starvation is having a population-level effect on any of the herds currently. Rain-on-snow events may increase in the face of climate change (Rennert et al. 2009, p. 2312). A doubling of carbon dioxide is estimated to cause a 40 percent increase in the area impacted by rain-on-snow events in the Arctic by 2080 (Rennert et

al. 2009, p. 2312). Rain-on-snow events may become more prevalent primarily in northwestern Canada, Alaska, and eastern Russia (Rennert et al. 2009, p. 2312). We have no reports that rain-on-snow events have led to the deaths of bison, but they could be susceptible to starvation by such events (USFWS, 2012).

**Stressor:** Genetic issues (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** It is recognized that genetic diversity in wood bison is relatively low, and that the herds must be managed to maintain genetic diversity. Loss of genetic diversity is a factor that may limit the ability of wood bison to adapt to changing conditions in the future, but the magnitude of that limitation, if it exists, is unknown. Lack of genetic diversity is potentially limiting over the long term, depending on the magnitude of environmental change wood bison may face. Because no effects of inbreeding have been documented and management actions have been shown to be effective, we conclude that loss of genetic diversity is not a threat to wood bison now or in the foreseeable future. Hybridization with plains bison is a threat that most likely will increase in the future. Because of consumer demand for bison meat, we expect commercial bison production will continue to expand, removing suitable habitat for wood bison recovery herds, and increasing the probability that escaped plains bison will be free on the landscape. Hybridization is a threat to wood bison now and in the foreseeable future (USFWS, 2012).

### ***Recovery***

**Recovery Actions:**

- A recovery plan has not been issued for this species.

### **References**

USFWS. 2016. Environmental Conservation Online System (ECOS) – Species Profile. <http://ecos.fws.gov/ecp0/>. Accessed July 2016

U.S. Fish and Wildlife Service. 2012. Endangered and Threatened Wildlife and Plants

Reclassifying the Wood Bison Under the Endangered Species Act as Threatened Throughout Its Range. Final Rule. FR Vol. 77, No. 86. Pages 26191-26212.

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## **SPECIES ACCOUNT: *Brachylagus idahoensis* (Columbia Basin pygmy rabbit (Columbia Basin DPS))**

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### ***Species Taxonomic and Listing Information***

**Listing Status:** Endangered; 11/30/2001; Pacific Region (R1) (USFWS, 2016)

### **Physical Description**

The pygmy rabbit is the smallest leporid in North America, with mean adult weights from 375 to about 500 grams (0.83 to 1.1 pounds), and lengths from 23.5 to 29.5 centimeters (9.3 to 11.6 inches) (Orr 1940; Janson 1946; Wilde 1978; Gahr 1993; WDFW 1995). Females tend to be slightly larger than males. Pygmy rabbits undergo an annual molt. Their overall color is slate-gray tipped with brown. Their legs, chest, and nape (back of neck) are tawny cinnamonbrown, their bellies are whitish, and the entire edges of their ears are pale buff. Their ears are short (3.5 to 5.2 centimeters [1.4 to 2.0 inches]), rounded, and thickly furred outside. Their tails are small (1.5 to 2.4 centimeters [0.6 to 0.9 inch]), uniform in color, and nearly unnoticeable in the wild (Orr 1940; Janson 1946; WDFW 1995). The pygmy rabbit is distinguishable from other rabbit species by its small size, short ears, gray color, small hind legs, and lack of white on the tail (USFWS, 2012).

### **Taxonomy**

The pygmy rabbit is a member of the family Leporidae, which includes hares and rabbits. The species has been placed in several genera since it was first classified in 1891 as *Lepus idahoensis* (Washington Department of Fish and Wildlife [WDFW] 1995). In 1904, it was reclassified and placed in the genus *Brachylagus*, and in 1930, it was again reclassified and placed in the genus *Sylvilagus*. More recent examination of dentition (Hibbard 1963) and analysis of blood proteins (Johnson 1968) suggest that the pygmy rabbit differs significantly from species within either the *Lepus* or *Sylvilagus* genera. The pygmy rabbit is now generally considered to be within the monotypic genus *Brachylagus*, and is again classified as *B. idahoensis* (Green and Flinders 1980a; WDFW 1995). There are no recognized subspecies of the pygmy rabbit (Dalquest 1948; Green and Flinders 1980a) (USFWS, 2012).

### **Historical Range**

Columbia Basin pygmy rabbits were considered rare during the early 20th century (Dalquest 1948), although there is little comprehensive information available regarding their historical abundance (WDFW 1995). Columbia Basin pygmy rabbits were thought to be extirpated from Washington during the mid-20th century, until a possible sighting was documented in Benton County in 1979. Intensive surveys in 1987 and 1988 located five small subpopulations in southern Douglas County. Three of the subpopulations were found on State lands and two were found on private lands (WDFW 1995). With the exception of the Benton County record, Columbia Basin pygmy rabbits have only been found in southern Douglas and northern Grant Counties since the mid-20th century (WDFW 2001a) (USFWS, 2012).

### **Distinct Population Segments Defined**

Columbia Basin DPS

### **Critical Habitat Designated**

Yes;

### ***Life History***

#### **Feeding Narrative**

Adult: Pygmy rabbits occur in the semiarid shrub steppe biome of the Great Basin and adjacent intermountain regions of the western United States. Within this broad biome, pygmy rabbits are typically found in habitat types that include tall, dense stands of sagebrush (*Artemisia* spp.), on which they are highly dependent to provide both food and shelter throughout the year. The pygmy rabbit is one of only two native rabbit species in North America that digs its own burrows and, therefore, is most often found in areas with relatively deep, loose soils that allow burrowing (USFWS, 2012).

#### **Reproduction Narrative**

Adult: Pygmy rabbits begin breeding the year following their birth and, in Washington, breeding occurs from January through June (Gahr 1993). Gestation in captive pygmy rabbits is from 22 to 24 days (Elias 2004), and females can produce from one to four litters per year (Elias 2004). Kits emerge from their burrows at roughly 2 weeks of age, and average litter sizes in captivity were roughly 3.5 kits per litter at the time of emergence (Elias 2004). Breeding appears to be highly synchronous in a given area, and juveniles are often identifiable to cohorts (Wilde 1978; Becker pers. comm. 2012). Information on captive and wild pygmy rabbits indicates that females excavate specialized, cryptic “natal” burrows that are disassociated from their resident burrow systems (P. Swenson, Oregon Zoo, pers. comm. 2001; Elias 2004; Rachlow et al. 2005). Recorded lengths of natal burrows from entrance to nest ranged from 15.5 to 35.5 centimeters (7 to 14 inches). In the wild, natal burrows typically consist of a single entrance under a large sagebrush plant (Rachlow et al. 2005). Females begin to dig and supply nesting material (e.g., plucked fur, grass clippings) to these burrows several days prior to giving birth, and may give birth and nurse their young in the runway to the burrow’s entrance. After nursing, the young return to the burrow and the female fills the burrow entrance with loose soil and otherwise disguises the immediate area to avoid detection (Elias 2004; Rachlow et al. 2005). Captive pygmy rabbit females sometimes construct other “dead-end” burrows that appear to be associated with their natal burrows, and female pygmy rabbits may alter their defecation and latrine habits while pregnant or nursing (P. Swenson, pers. comm. 2001). Ongoing work with captive and wild pygmy rabbits should provide additional information concerning details of their reproductive strategy.

Mortality Rates: The annual mortality rates of adult pygmy rabbits may be as high as 88 percent, and over 50 percent of juveniles may die within roughly 5 weeks of their emergence (Wilde 1978; WDFW 1995). However, the mortality rates of adult and juvenile pygmy rabbits can vary considerably between years, and even between juvenile cohorts within years (Wilde 1978). Starvation and environmental stress likely account for some mortality in wild pygmy rabbits (Wilde 1978), however, predation is generally considered to be the main cause of mortality (Green 1979). Potential predators include fossorial and terrestrial mammals such as badgers, long-tailed weasels (*Mustela frenata*), coyotes (*Canis latrans*), and bobcats (*Felis rufus*), and a variety of avian predators such as great horned owls (*Bubo virginianus*), long-eared owls (*Asio otus*), ferruginous hawks (*Buteo regalis*), northern harriers (*Circus cyaneus*), and common ravens (*Corvus corax*) (Janson 1946; Gashwiler et al. 1960; Green 1978; WDFW 1995; M. Hallet, WDFW, pers. comm. 2002). Population cycles are not known in pygmy rabbits, although local, rapid population declines have been noted in several states (Bradfield 1974; Weiss and Verts 1984; WDFW 1995). After initial declines, pygmy rabbit populations may not have the same capacity

for rapid increases in numbers as other leporids due to their close association with specific components of sagebrush ecosystems, and the relatively limited availability of their preferred habitats (Wilde 1978; Green and Flinders 1980b; WDFW 1995) (USFWS, 2012)

**Tolerance Ranges/Thresholds**

Adult: Low (inferred from USFWS, 2012)

**Site Fidelity**

Adult: High (inferred from USFWS, 2012)

**Habitat Narrative**

Adult: Pygmy rabbits occur in the semiarid shrub steppe biome of the Great Basin and adjacent intermountain regions of the western United States. Within this broad biome, pygmy rabbits are typically found in habitat types that include tall, dense stands of sagebrush (*Artemisia* spp.), on which they are highly dependent to provide both food and shelter throughout the year. The pygmy rabbit is one of only two native rabbit species in North America that digs its own burrows and, therefore, is most often found in areas with relatively deep, loose soils that allow burrowing (USFWS, 2012). High ecological integrity of the community and site fidelity as well as low tolerance ranges are inferred based on the specific habitat needs of the species and the fact that the species has always been thought to be scarce and limited in numbers. The species has also been extirpated from the wild.

***Dispersal/Migration*****Motility/Mobility**

Adult: High (USFWS, 2010)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory (USFWS, 2010)

**Dispersal**

Adult: High (USFWS, 2010)

**Dispersal/Migration Narrative**

Adult: New information from a study in Idaho indicates that pygmy rabbits have a greater dispersal capability than previously known (Rachlow and Estes-Zumpf 2005, pages 1-3). These more recent records indicate that juvenile pygmy rabbits often undertake a single, rapid dispersal movement between 4 to 12 weeks of age, and that some juvenile animals may disperse over 6 miles (10 kilometers) during this period. In addition, adult pygmy rabbits may disperse over 7.5 miles (12 kilometers) between their more restricted, seasonal use sites. While these movements are considerably longer than those previously documented, it should also be noted that they are maximum estimates and there appear to be large differences in the propensity of individual pygmy rabbits to disperse, with many animals remaining relatively sedentary. Reflecting this, median recorded dispersal distances for the Idaho pygmy rabbits were 0.7 mile (1.1 kilometers) and 1.9 miles (3.0 kilometers) for males and females, respectively (USFWS, 2010).

***Population Information and Trends***

**Population Trends:**

Unknown (inferred from USFWS, 2012)

**Population Size:**

157 genetically-identified wild-born rabbits (USFWS 2019)

**Population Narrative:**

Surveys of this last known subpopulation have not detected any animals since before July 2004 (B. Patterson, WDFW, pers. comm. 2004), indicating that the Columbia Basin pygmy rabbit may have been extirpated from the wild. However, due to other priorities and limited access to private lands (see Conservation Actions Implemented), only about 7.7 percent, or 46,000 hectares (113,600 acres) of the potentially suitable shrub steppe habitat that remains within the Columbia Basin (totaling roughly 599,000 hectares [1,479,500 acres]) has been surveyed specifically for pygmy rabbit presence since 2001 (USFWS 2010). Therefore, other wild but as yet unknown pygmy rabbit subpopulations may still be present within the Columbia Basin and ongoing survey effort to detect any that may remain has been identified as a key action in this Recovery Plan. In 2011, reintroduction efforts for the Columbia Basin pygmy rabbit were resumed. New measures being implemented for these ongoing efforts include capturing wild pygmy rabbits from populations outside of the Columbia Basin to include them in the reintroduction program and holding some of the program animals at the release site in large (up to 4-hectare [10-acre]) enclosures (see Conservation Actions Implemented). During the summer of 2011, 16 captive-bred adults and 48 captive-bred kits were released into habitat historically occupied by the species in the Columbia Basin. Even with increased protective measures implemented during release efforts, the captive-bred adults again experienced very high mortality and none are believed to have survived. However, several of the captive-bred kits appear to have developed resident burrow systems and successfully over-wintered at the site. Another 34 captive-bred adults and 32 wild-caught adults (from populations in Oregon and Nevada) were placed in the large enclosures at the release site during the fall of 2011. Many of these animals from the captive-bred and wild-caught groups successfully over-wintered in the enclosures. Another 11 captive-bred adults and 44 wild-caught adults (from populations in Nevada and Utah, plus 1 Idaho male captured in 2009) were placed in the enclosures during spring and summer of 2012. Many of the animals in the large enclosures successfully produced over 130 kits during the 2012 breeding season (see Reintroduction). Finally, as of July 31, 2012, another 103 captive-bred and enclosure-bred kits have been released into habitat historically occupied by the species in the Columbia Basin (USFWS, 2012).

**Threats and Stressors**

**Stressor:** Crop Production (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Dry-land and irrigated crop production has converted and fragmented large portions of the native shrub steppe habitats that were present within the Columbia Basin prior to European settlement in the region (Daubenmire 1988; Franklin and Dyrness 1988; Dobler et al. 1996; WDFW 1995). In addition, urban and rural developments permanently remove native shrub steppe habitats. It has been estimated that nearly 60 percent of the native shrub steppe

habitats originally within the Columbia Basin have been converted to other uses (Dobler et al. 1996). Columbia Basin pygmy rabbits cannot occupy these converted sites and, due to their relatively restricted movements, fragmentation of shrub steppe habitats severely limits their ability to disperse (Katzner and Parker 1997) (USFWS, 2012).

**Stressor:** Fire (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Sagebrush is easily killed by burning, and when fires occur at increased frequency they can remove sagebrush from the vegetation community (Daubenmire 1988; WDFW 1995). Fire frequency has increased over portions of the remaining shrub steppe habitats within the Columbia Basin as a result of various influences, including the establishment of invasive plant species, unimproved road access, and certain recreational activities. Due to their reliance on tall, dense stands of sagebrush and associated shrub steppe vegetation, Columbia Basin pygmy rabbits cannot occupy frequently burned sites (USFWS, 2012).

**Stressor:** Nonnative/invasive plants (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Various nonnative, invasive plant species such as cheat grass and knapweed (*Centaurea* spp.) have become well established throughout the Columbia Basin (Daubenmire 1988; Franklin and Dyrness 1988). Areas with dense cover of cheat grass are apparently avoided by pygmy rabbits in Oregon (Weiss and Verts 1984), and these newly established plant communities often provide fine fuels that can carry fires. Combined with widespread unimproved road access and informal recreational activities that can provide multiple sources of ignition, the establishment of non-native, invasive plant species increases the risk of fire, and reduces the security and suitability of areas that could potentially support the Columbia Basin pygmy rabbit (WDFW 1995) (USFWS, 2012).

**Stressor:** Livestock grazing (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Under certain circumstances, livestock grazing can negatively impact the Columbia Basin pygmy rabbit. The effects may depend on a variety of factors including livestock type, timing and duration of grazing, stocking densities, locations of water or mineral supplement blocks, and other factors that may concentrate livestock use. Impacts to pygmy rabbits may include damage to burrow systems and possible direct mortality to young due to trampling (Rauscher 1997; N. Siegel, Washington State University [WSU], pers. comm. 2001; P. Becker, pers. comm. 2011), altered movement and behavioral patterns (Gahr 1993; Siegel 2002), fewer available burrows (Siegel 2002), and decreased quantity and nutritional quality of forage species in grazed areas (Siegel-Thines et al. 2004) (USFWS, 2012).

**Stressor:** Disease (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** A number of captive Columbia Basin pygmy rabbits died as a result of various diseases, especially coccidiosis and mycobacteriosis (WDFW 2010; Harrenstien et al. 2006; Harrenstien et al. 2011). Coccidiosis is caused by a protozoan (likely *Eimeria* spp.) that occurs in soil and feces, and which invades the intestines and other tissues of animals. Coccidiosis may be most detrimental in neonate pygmy rabbits, as both adult and young animals can apparently remain free of the disease while harboring high levels of coccidia. Various preventive measures that were undertaken (see Conservation Actions Implemented) appear to have been effective at decreasing the incidence of coccidiosis in the captive population. The bacterium that causes mycobacteriosis (*Mycobacterium avium*) commonly exists in soil and water, and can survive for long periods of time in soil. High numbers of the bacterium can also be shed in feces and urine. The incubation period for mycobacteriosis can be weeks to months, and detection of infected individuals is difficult. Comparisons of immune system function (i.e., lymphocyte stimulation and cytokine assays) among pygmy rabbits from the Columbia Basin and populations in Idaho, as well as the riparian brush rabbit (*Sylvilagus bachmani riparius*) and domestic rabbits (*Oryctolagus* spp.), have been undertaken (Harrenstien et al. 2006). In general, Columbia Basin pygmy rabbits had a significantly poorer immune response to mycobacteriosis than pygmy rabbits from Idaho and the other lagomorph species. A partially-ineffective cell-mediated immune response appears to be the most likely cause of their high mortality resulting from mycobacteriosis. A relationship between diminished genetic diversity (see Factor E) and higher susceptibility to mycobacteriosis has been demonstrated in other endangered species (Harrenstien et al. 2006) (USFWS, 2012).

**Stressor:** Predation (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Predation is thought to be the major cause of mortality among pygmy rabbits (Green 1979). However, pygmy rabbits have adapted to the presence of a wide variety of avian and terrestrial predators that occur throughout their historical distribution (Janson 1946; Gashwiler et al. 1960; Green 1978; WDFW 1995). In relatively large, well distributed pygmy rabbit populations, predation is not likely to represent a significant threat to their long-term security. In contrast, due to the extremely small size and localized occurrence of the Columbia Basin pygmy rabbit population, altered predation patterns, or even natural levels of predation, currently represent a significant threat to reestablishment of this population segment in the wild and could impair ongoing conservation efforts (USFWS, 2012).

**Stressor:** Inadequacy of existing regulatory mechanisms (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Washington classification of the pygmy rabbit as a State endangered species makes it illegal to hunt, possess, maliciously harass or kill pygmy rabbits, or to maliciously destroy their nests, unless otherwise authorized by the Washington Wildlife Commission (Revised Code of Washington 77.15.120). However, this State designation does not provide regulatory protection from activities that may incidentally harm the Columbia Basin pygmy rabbit, nor does it provide regulatory mechanisms to protect habitat that may be considered essential to its long-term security. Washington legislation (i.e., House Bill 1309) prescribes ecosystem standards for State-owned agricultural and grazing lands to maintain and restore fish and wildlife habitat by

improving overall ecosystem health. However, these standards do not specifically address protection and conservation of the Columbia Basin pygmy rabbit, and are only mandated for lands under the jurisdiction of WDFW and WDNR. In addition, application of the standards on lands managed by WDNR must be consistent with the agency's fiduciary obligations. Large areas of privately owned land within the historical distribution of the Columbia Basin pygmy rabbit have been withdrawn from crop production and planted to native and nonnative cover under the Federal Conservation Reserve Program administered by USDA. Revegetation standards under this program promote the improvement of habitats potentially used by the Columbia Basin pygmy rabbit. The program also restricts livestock grazing on contract lands except under severe drought conditions (USFWS 2001). However, the measures prescribed under this program do not specifically address conservation of the Columbia Basin pygmy rabbit, participation is voluntary, contracts expire after 10 years, and changes to program requirements and management objectives at each renewal period are common (USDA 1998). Presently, it is unclear what effects recent program changes have had, or future changes may have, on recovery efforts for the Columbia Basin pygmy rabbit. Certain conservation measures developed under the Endangered Species Act, including the template Safe Harbor Agreement and a county-wide Habitat Conservation Plan currently under development, can provide protection and conservation incentives for Columbia Basin pygmy rabbits (see discussion below under section I.G.6, Stakeholder Involvement). These measures would apply only to willing landowners participating in these programs (USFWS, 2012).

**Stressor:** Small population size (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The most immediate concerns for the Columbia Basin pygmy rabbit are associated with the population's extremely small size and possible extirpation from the wild (USFWS 2010). Small populations are highly susceptible to random environmental events (e.g., severe storms, prolonged drought, extreme cold spells), abrupt changes in cover or food resources (e.g., from wildfire or insect infestations), altered predator or parasite populations, disease outbreaks, and fire. Small populations are also more susceptible to demographic and genetic limitations (Shaffer 1981). These threat factors, which may act in concert, include natural variation in survival and reproductive success of individuals, chance disequilibrium of sex ratios, changes in gene frequencies due to genetic drift, and diminished genetic diversity and associated effects due to inbreeding. These influences continue to represent a significant risk to the potential reestablishment of the Columbia Basin pygmy rabbit and its long-term security in the wild (USFWS 2010) (USFWS, 2012).

**Stressor:** Climate change (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** When there is sufficient information available, our analyses conducted pursuant to the ESA include consideration of ongoing and projected changes in climate. The terms "climate" and "climate change" are defined by the Intergovernmental Panel on Climate Change (IPCC). "Climate" refers to the mean and variability of different types of weather conditions over time, with 30 years being a typical period for such measurements, although shorter or longer periods also may be used (IPCC 2007). The term "climate change" thus refers to a change in the mean or

variability of one or more measures of climate (e.g., temperature, precipitation) that persists for an extended period, typically decades or longer, whether the change is due to natural variability, human activity, or both (IPCC 2007). Various types of changes in climate can have direct or indirect effects on species. These effects may be positive, neutral, or negative and they may change over time, depending on the species and other relevant considerations, such as the effects of interactions of climate with other variables (e.g., habitat fragmentation) (IPCC 2007). In our analyses, we use our expert judgment to weigh relevant information, including uncertainty, in our consideration of various aspects of climate change (USFWS, 2012).

### ***Recovery***

#### **Reclassification Criteria:**

We will consider reclassification of the Columbia Basin pygmy rabbit from endangered to threatened status pursuant to the measures prescribed by the ESA if any one of the following criteria is met. 1 – Subpopulations at 2 recovery emphasis areas each have a 5-year average  $N_e$  of at least 375 individuals, and a third recovery emphasis area has been formally established through completion of one or more appropriate conservation agreements and is available for initial reintroduction efforts; or (USFWS, 2012).

2 – A subpopulation at 1 recovery emphasis area has a 5-year average  $N_e$  of at least of 250 individuals, and subpopulations at 2 other recovery emphasis areas each have a 5-year average  $N_e$  of at least 125 individuals; or (USFWS, 2012).

3 – A single subpopulation with a 5-year average  $N_e$  of at least of 750 individuals has been reestablished through dispersal and range expansion from one or more recovery emphasis areas, and appropriate conservation agreements have been reached to include the newly occupied habitats within the recovery emphasis area(s) involved and management measures to maintain identified dispersal corridors have been agreed to and implemented (USFWS, 2012).

#### **Delisting Criteria:**

A minimum 5-year average of at least 2,800 adult Columbia Basin pygmy rabbits in at least 12 populations. Of these, at least 4 populations have 500 or more adults each and at least 8 populations have 100 or more adults each (USFWS 2019).

Habitat security for the 12 populations has been established (WDFW 1995, p. 25) (USFWS 2019).

### ***Conservation Measures and Best Management Practices:***

- 1 – Continue efforts to reestablish a viable population of pygmy rabbits within the Columbia Basin through release of the captive, intercrossed pygmy rabbits combined with simultaneous direct translocation of pygmy rabbits captured from outside of the Columbia Basin. This recommendation addresses recovery actions 2 (management of genetic characteristics) and 4 (reestablishment of wild subpopulations) in the current draft recovery plan for the Columbia Basin pygmy rabbit (USFWS, 2010).
- 2 – Carefully monitor all pygmy rabbit subpopulations within the Columbia Basin to determine their collective genetic status, including estimated levels of introgression, and continue to adjust future management actions based on a comprehensive adaptive management approach. This recommendation addresses recovery actions 3 (monitor wild subpopulations) and 10 (revise Federal

recovery plan) in the current draft recovery plan for the Columbia Basin pygmy rabbit (USFWS, 2010).

- 3 – Investigate if, and the extent to which, pygmy rabbit populations differentiate with regard to their epigenomes. This recommendation addresses recovery actions 2 (management of genetic characteristics) and 3 (monitor wild subpopulations) in the current draft recovery plan for the Columbia Basin pygmy rabbit (USFWS, 2010).
- 4 – Amend the current draft recovery plan for the Columbia Basin pygmy rabbit. This recommendation addresses recovery action 10 (revise Federal recovery plan) in the current draft recovery plan for the Columbia Basin pygmy rabbit (USFWS, 2010).

## References

USFWS. 2016. Environmental Conservation Online System (ECOS) – Species Profile. <http://ecos.fws.gov/ecp0/>. Accessed July 2016

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U.S. Fish and Wildlife Service. 2012. Recovery Plan for the Columbia Basin Distinct Population Segment of the Pygmy Rabbit (*Brachylagus idahoensis*). Portland, Oregon. ix + 109 pp

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## SPECIES ACCOUNT: *Canis lupus* (Gray wolf (all ssp. within U.S.))

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### *Species Taxonomic and Listing Information*

**Commonly-used Acronym:** None

**Listing Status:** Endangered (all states except Minnesota - threatened; Northern Rocky Mt. pop. excepted): 02/17/22

### **Physical Description**

The gray wolf is the largest wild member of the Canidae, or dog family, with adults ranging from 18 to 80 kilograms (40 to 175 pounds), depending on sex and subspecies (68 FR 15804). The total length of the gray wolf is about 205 centimeters (cm) (80 inches [in.], with a tail of up to 50 cm (20 in.) in length, and a nose pad averaging 3.1 cm (1.2 in.) or more in diameter. The upper canine is more than 1.2 cm (0.5 in.) in anteroposterior diameter at the base, not extending below the level of the anterior mental foramen when the lower jaw is in place. The condylobasal length of the skull is 20.3 to 26.9 cm (8 to 10.6 in.) (NatureServe 2015). The wolves' fur color is frequently a grizzled gray, but it can vary from pure white to coal black. Wolves may appear similar to coyotes (*Canis latrans*) and some domestic dog breeds (such as the German shepherd or Siberian husky) (*C. familiaris*). However, wolves' longer legs, larger feet, wider head and snout, and straight tail distinguish them from both coyotes and dogs (68 FR 15804).

### **Taxonomy**

Gray wolf taxonomy has undergone substantial revisions in recent years, including a major taxonomic revision in which the number of recognized gray wolf subspecies in North America was reduced from 24 to five: eastern wolf (*C. l. lycaon*), Mexican wolf (*C. l. baileyi*), Arctic wolf (*C. l. arctos*), northern timber wolf (*C. l. occidentalis*), and plains wolf (*C. l. nubilus*) (Chambers et al. 2012). The Mexican wolf is considered a morphologically distinct and valid subspecies, based on skull morphometrics and unique genetic markers, and is listed—and therefore treated separately—from the remaining gray wolves in North America (80 FR 2488). The gray wolf differs from the coyote in its larger nose pad, more rounded ears, larger anteroposterior diameter of upper canine at gum level, larger heel pad on the forefoot, longer skull, and relatively shorter canines. Also, the gray wolf holds the tail high when running, while the coyote holds it low. In some parts of central and eastern North America, the coyote approaches the wolf in certain characteristics, due to interbreeding. The gray wolf differs from the red wolf in its larger size, longer skull, and in certain features of the molars; however, the red wolf actually may be a coyote-gray wolf hybrid. The gray wolf differs from the domestic dog in its generally larger size, broader nose pad, more massive skull with heavier teeth, relatively longer rostrum, supraoccipital shield which is larger and projects farther posteriorly, and longer and narrower front foot track (NatureServe 2015).

### **Historical Range**

Until the molecular genetics studies of the last few years, the range of the gray wolf prior to European settlement was generally believed to include most of North America. The only areas that were believed to have lacked gray wolf populations are southern and interior Greenland, the coastal regions of Mexico, all of Central America south of Mexico, coastal and parts of California, the extremely arid deserts and the mountaintops of the western United States, and parts of the eastern and southeastern United States. However, some authorities question the

reported historical absence of gray wolves from parts of California. Authors are inconsistent in their views of the precise boundary of historical gray wolf ranges in the eastern and southeastern United States (68 FR 15804). The U.S. Fish and Wildlife Service (USFWS) views the historical range of the gray wolf as the central and western United States, including portions of the Western Great Lakes region, the Great Plains, portions of the Rocky Mountains, the Intermountain West, the Pacific Northwest, and portions of the Southwest. All or parts of 29 southern and eastern states (Maine, Massachusetts, Connecticut, New Hampshire, Rhode Island, Vermont, New York, New Jersey, Pennsylvania, Delaware, Maryland, Virginia, North Carolina, South Carolina, Georgia, Florida, Ohio [the part outside the Western Great Lakes DPS], West Virginia, Kentucky, Tennessee, Alabama, Mississippi, Louisiana, Texas [east of Interstate Highway 35], Oklahoma [east of Interstate Highway 35 and southeast of Interstate Highway 44 north of Oklahoma City], Arkansas, Missouri [southeast of Interstate Highway 44 and southeast of Interstate Highway 70 east of St. Louis], Indiana [the part outside the Western Great Lakes DPS], and Illinois [the part outside the Western Great Lakes DPS]) were not within the gray wolf's historical range (76 FR 26086).

**Current Range**

The present-day geographical area of the eastern (aka Minnesota)/Western Great Lakes DPS is described as all of Minnesota, Wisconsin, and Michigan; the portion of North Dakota north and east of the Missouri River upstream to Lake Sakakawea and east of the centerline of Highway 83 from Lake Sakakawea to the Canadian border; the portion of South Dakota north and east of the Missouri River; the portions of Iowa, Illinois, and Indiana north of the centerline of Interstate Highway 80; and the portion of Ohio north of the centerline of Interstate Highway 80 and west of the Maumee River at Toledo (50 FR 81666). The present-day geographical area of the western (aka Wyoming)/northern Rocky Mountains DPS includes California, northern Colorado, Idaho, Montana, Oregon, northern Utah, Washington, and Wyoming (77 FR 55530).

**Distinct Population Segments Defined**

Yes: Gray Wolf includes all subspecies and areas outside of DPSs identified below): (AL, AR, AZ, CA, CO, CT, DE, FL, GA, KS, KY, LA, MA, MD, ME, MO, MS, NC, NE, NH, NJ, NM, NV, NY, OK, PA, RI, SC, TN, TX, VA, VT, and WV, and portions of IA, IN, IL, ND, OH, OR, SD, UT, and WA), March 9, 1978 (43 FR 9607) Northern Rocky Mountain DPS (western): (excluding Wyoming), April 2, 2009 (74 FR 15123); (Wyoming), April 2, 2009 (74 FR 15123) Western Great Lakes DPS: (eastern) (Minnesota), March 9, 1978 (43 FR 9607); (excluding Minnesota), April 2, 2009 (74 FR 15070)

**Critical Habitat Designated**

Yes; 6/9/1977.

**Legal Description**

On March 9, 1978, the Service issued a final rulemaking which provides for the reclassification of the may wolf in the United States and Mexico, and for the determination of critical habitat for species of gray wolf in Michigan and Minnesota. The reclassification is considered to accurately express the current status of the gray wolf, based solely on an evaluation of the best available biological data. The special regulations being established in Minnesota are deemed necessary and advisable to provide for the future well-being of the species.

**Critical Habitat Designation**

Michigan. Isle Royale National Park. Minnesota. Areas of land, water, and airspace in Beltrami, Cook, Itasca, Koochiching, Lake, Lake of the Woods, Roseau, and St. Louis Counties, with boundaries (4th and 5th Principal meridians) identical to those of zones 1, 2, and 3, as delineated in 50 CFR 17.40(d)(1).

Zone 1 - 4,488 square miles. Beginning at the point of intersection of United States and Canadian boundaries in Section 22, Township 71 North, Range 22 West, in Rainy Lake, then proceeding along the west side of Sections 22, 21, and 34 in said Township and Sections 3, 10, 15, 22, 27 and 24 in Township 70 North, Range 22 West and Sections 3 and 10 in Township 69 North, Range 22 West; then east along the south boundaries of Sections 10, 11, and 12 in said Township; then south along the Koochiching and St. Louis counties line to Highway 53; thence southeasterly along State Highway 53 to the junction with County Route 765; thence easterly along County Route 765 to the junction with Kabetogama Lake in Ash River Bay; thence along the south boundary of Section 33 in Township 69 North, Range 19 West, to the junction with the Moose River; thence southeasterly along the Moose River to Moose Lake; thence along the western shore of Moose Lake to the river between Moose Lake and Long Lake; thence along the said river to Long Lake; thence along the east shore of Long Lake to the drainage on the southeast side of Long Lake in NE1/4. Section 16, Township 67 North, Range 1a West; thence along the said drainage southeast side and subsequently northeasterly to Marion Lake, the drainage being in Section 17 and 16, Township 67 North, Range 16 West; thence along the west shoreline of Marion Lake proceeding southeasterly to the Moose Creek: thence along Moose Creek to Flap Creek: thence southeasterly along Flap Creek to the Vermilion River: thence southerly along the Vermilion River to Vermilion Lake: thence along the Superior National Forest boundary in a southeasterly direction through Vermilion Lake passing these points: Oak Narrows, Muskrat Gchannel, South of Pine Island, to Hoodo Point and the junction with County Route 697; thence southeasterly on County Route 697 to the junction with State Highway 169: thence easterly along State Highway 169 to the junction with State Highway 1: thence easterly along State Highway 1 to the junction with the Erie Railroad tracks at Murphy City: thence easterly along the Erie Railroad tracks to the junction with Lake Superior at Taconite Harbor; thence northeasterly along the North Shore of Lake Superior to the Canadian Border; thence westerly along the Canadian Border to the point of beginning in Rainy Lake.

Zone 2 - 1,856 square miles. Beginning at the intersection of the Erie Mining Co. Railroad and State Highway 1 (Murphy City); thence southeasterly on State Highway 1 to the junction with County Road 4: thence southwesterly on County Road 4 to the State Snowmobile Trail (formerly the Alger-Smith Railroad); thence southwesterly to the intersection of the Old Railroad Grade and Reserve Mining Co. Railroad in Section 33 of Township 56 North, Range 9 West; thence northwesterly along the Railroad to Forest Road 107; thence westerly along Forest Road 107 to Forest Road 203; thence westerly along Forest Road 203 to the junction with County Route 2; thence in a northerly direction on County Route 2 to the junction with Forest Road 122; thence in a westerly direction along Forest Road 122 to the Junction with the Duluth, Missable and Iron Range Railroad; thence in a southwesterly direction along the said railroad tracks to the junction with County Route 14; thence in a northwesterly direction along County Route 14 to the junction with County Route 55; thence in a westerly direction along County Route 55 to the junction with County Route 44; thence in a southerly direction along County Route 44 to the junction with County Route 266; thence in a southeasterly direction along County Route 266 and subsequently in a westerly direction to the junction with County Road 44; thence in a northerly direction on County Road 44 to the junction with Township Road 2615; thence westerly along Township Road

2615 to Alden Lake; thence northwesterly across Alden Lake to the Inlet of the Cloquet River; thence northerly along the Cloquet River to the junction with Carrol Trail-State Forestry Road; thence west along the Carrol Trail to the junction with County Route 4 and County Route 49; thence west along County Route 49 to the junction with the Duluth, Winnipeg and Pacific Railroad; thence in a northerly direction along said Railroad to the junction with the Whiteface River; thence in a northeasterly direction along the Whiteface River to the Whiteface Reservoir; thence along the western shore of the Whiteface Reservoir to the junction with County Route 340; thence north along County Route 340 to the junction with County Route 16; thence east along County Route 16 to the junction with County Route 346; thence in a northerly direction along County Route 346 to the junction with County Route 569; thence along County Route 569 to the junction with County Route 565; thence in a westerly direction along County Route 565 to the junction with County Route 110; thence in a westerly direction along County Route 110 to the junction with County Route 100; thence in a north and subsequent west direction along County Route 100 to the junction with State Highway 135; thence in a northerly direction along State Highway 135 to the junction with State Highway 169 at Tower; thence in an easterly direction along the southern boundary of Zone 1 to the point of beginning of Zone 2 at the junction of the Erie Railroad Tracks and State Highway 1.

Zone 3 - 3,501 square miles. Beginning at the junction of State Highway 11 and State Highway 65; thence southeasterly along State Highway 65 to the junction with State Highway 1; thence westerly along State Highway 1 to the junction with State Highway 72; thence north along State Highway 72 to the junction with an un-numbered township road beginning in the northeast corner of Section 25, Township 155 North, Range 31 West; thence westerly along the said road for approximately seven (7) miles to the junction with SFR 95; thence westerly along SFR 95 and continuing west through the southern boundary of Sections 36 through 31, Township 155 North, Range 33 West, through Sections 36 through 31, Township 155 North, Range 34 West, through Sections 36 through 31, Township 155 North, Range 35 West, through Sections 36 and 35, Township 155 North, Range 36 West to the junction with State Highway 69, thence northwesterly along State Highway 69 to the junction with County Route 44; thence northerly along County Route 44 to the junction with County Route 704; thence northerly along County Route 704 to the junction with SFR 49; thence northerly along SFR 49 to the junction with SFR 57; thence easterly along SFR 57 to the junction with SFR 63; thence south along SFR 63 to the junction with SFR 70; thence easterly along SFR 70 to the junction with County Route 87; thence easterly along County Route 87 to the junction with County Route 1; thence south along County Route 1 to the junction with County Route 16; thence easterly along County Route 16 to the junction with State Highway 72; thence south on State Highway 72 to the junction with a gravel road (unnumbered County District Road) on the north side of Section 31, Township 158 North, Range 30 West; thence east on said District Road to the junction with SFR 62; thence easterly on SFR 62 to the junction with SFR 175; thence south on SFR 175 to the junction with County Route 101; thence easterly on County Route 101 to the junction with County Route 11; thence easterly on County Route 11 to the junction with State Highway 11; thence easterly on State Highway 11 to the junction with State Highway 65, the point of beginning.

#### **Primary Constituent Elements/Physical or Biological Features**

The Minnesota population represents the last significant element of a species that once occupied a vastly larger range in the lower 48 States, and long-term trends may be working against the wolf. To quote the Recovery Plan. "Future circumstances are unpredictable and those that now exist could change drastically. For example, widespread industrialization, mineral exploitation,

and general development could threaten much of the wolf's remaining range, making regulation increasingly significant to the populations left. Additional roads, railroads, power lines, mines and tourist facilities could further carve up much of northern Minnesota. This would disrupt the natural repopulation of depleted areas by wolves and promote higher human densities which would compete with wolves for their wild prey." Moreover, in recent years there has been a decline in deer, the main prey species, in parts of the primary range of the wolf.

PCEs are not described. Based on the text above, it can be inferred that (1) undeveloped lands in Minnesota and (2) deer are major constituent elements for this species.

### **Special Management Considerations or Protections**

The provisions for predator control state that wolves may be taken by authorized Federal or State employees in zones 2, 3, 4, and 5, if such wolves commit significant depredations on lawfully present domestic animals. Few, if any, of these wolves will be taken in zones 2 and 3 which have practically no livestock, and nearly all will be taken in zone 4. Essentially then, the wolf population in zones 1, 2, and 3 will not be affected by the depredation control activity. The population in zone 4 might be held below biological potential, but would continue to exist in reasonable numbers. The control of depredating wolves in zone 4 will reduce conflicts with human interests and should create a more favorable public attitude that would be of overall benefit to the wolf.

### ***Life History***

#### **Feeding Narrative**

Adult: Wolves primarily are predators of medium and large mammals. Wild prey species in North America include white-tailed deer (*Odocoileus virginianus*), mule deer (*O. hemionus*), moose (*Alces alces*), elk (*Cervus canadensis*), woodland caribou (*Rangifer caribou*), barren ground caribou (*R. arcticus*), bison (*Bison bison*), muskox (*Ovibos moschatus*), bighorn sheep (*Ovis canadensis*), Dall sheep (*O. dalli*), mountain goat (*Oreamnos americanus*), beaver (*Castor canadensis*), and snowshoe hare (*Lepus americanus*), with small mammals, birds, and large invertebrates sometimes being taken. In the Midwest, during the last 22 years, wolves have also killed domestic animals, including horses (*Equus caballus*), cattle (*Bos taurus*), sheep (*Ovis aries*), goats (*Capra hircus*), llamas (*Lama glama*), pigs (*Sus scrofa*), geese (*Anser sp.*), ducks (*Anas sp.*), turkeys (*Meleagris gallopavo*), chickens (*Gallus sp.*), pheasants (*Phasianus colchicus*), dogs (*Canis domesticus*), and cats (*Felis catus*). Since 1987, wolves in the northern Rocky Mountains of Montana, Idaho, and Wyoming have also killed domestic animals, including llamas, horses, cattle, sheep, and dogs (68 FR 15804). Gray wolves hunt in packs during nighttime/crepuscular hours, and wolf density is positively correlated to the amount of ungulate biomass available and the vulnerability of ungulates to predation (68 FR 15804). Young wolves achieve near-adult size at between 8 and 10 months of age (Mech 1974). Wolves are social animals, typically living and hunting in packs of two to 12 wolves—primarily family groups consisting of a breeding pair, their pups from the current year, offspring from the previous year, and occasionally an unrelated wolf (68 FR 15804).

#### **Reproduction Narrative**

Adult: Gray wolves are monogamous, breeding once per year, with an average of six to seven (and up to ten) pups per season. Courtship begins between January and April. After a gestation period of 63 days, the young are born blind in a den, typically a hole in the ground, or in a rock

crevice, hollow log, or an overturned stump. Pups are weaned in about 5 weeks, and young leave the den at about 3 months old. Some offspring remain with the pack; others disperse as they mature. Adults reach sexual maturity during their second or third year, and live for up to 10 to 16 years of age in the wild. Gray wolves have a high fitness, with an average reproductive capacity of 35 offspring in a lifetime. Only the dominant male and dominant female mate and rear offspring. Lone wolves generally do not successfully rear young, but they may if food is abundant (NatureServe 2015; Mech 1974; 68 FR 15804).

**Geographic or Habitat Restraints or Barriers**

Adult: Gray wolves occur only where human population density and persecution level are low and prey densities are high (NatureServe 2015).

**Spatial Arrangements of the Population**

Adult: Clumped

**Environmental Specificity**

Adult: Broad/generalist where key requirements are present.

**Site Fidelity**

Adult: Moderate

**Dependency on Other Individuals or Species for Habitat**

Adult: Abundance of ungulate prey (NatureServe 2015).

**Habitat Narrative**

Adult: Gray wolves show no particular habitat preferences, and can be found in such vegetation types as alpine, desert, conifer forest, hardwood forest, mixed forest, grassland/herbaceous, savanna, shrubland/chaparral, tundra, conifer woodland, hardwood woodland, and mixed woodland (NatureServe 2015). However, they are generally found only where human population density and persecution level are low and prey densities of ungulates are high (NatureServe 2015).

***Dispersal/Migration*****Motility/Mobility**

Adult: High

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Gray wolves have an annual home range (territory) of up to several hundred square kilometers (km<sup>2</sup>) (hundred square miles [sq. mi.]). Young gray wolves disperse from natal to new territories between the ages of 1 and 2 years, typically between February-April and October-November (NatureServe 2015).

**Immigration/Emigration**

Adult: Immigration/emigration.

**Dependency on Other Individuals or Species for Dispersal**

Adult: No

**Dispersal/Migration Narrative**

Adult: Gray wolves are highly mobile and readily disperse or migrate hundreds of kilometers (hundred or more miles). Gray wolves have an annual home range (territory) of up to several hundred km<sup>2</sup>. Young gray wolves disperse from natal to new territories (hundreds of kilometers [hundred or more miles]) between the ages of 1 and 2 years, typically between February-April and October-November; 35 percent of known-age wolves remained in their natal territory for more than 2 years (NatureServe 2015).

***Population Information and Trends*****Population Trends:**

Northern Rocky Mountain DPS (western) Idaho – increasing (IDFG 2015); Montana -decreasing (Bradley et al. 2015); Wyoming – increasing (WGFD et al. 2015); Washington – increasing (Becker et al. 2015); and Oregon – increasing (ODFW 2015). Western Great Lakes DPS (eastern) Minnesota – declining-to-stable due to hunting and depredation control; Wisconsin – declining due to human-caused mortality; Michigan – declining for unstated reasons.

**Species Trends:**

Increasing to stable to declining, depending on location.

**Resiliency:**

High

**Representation:**

High

**Redundancy:**

Moderate

**Population Growth Rate:**

Dependent on DPS, state, and localized conditions.

**Number of Populations:**

Two populations: northern Rocky Mountains, Western Great Lakes (not including the Mexican gray wolf) in the conterminous United States (USFWS 2012). 81 to more than 300 occurrences (NatureServe 2015).

**Population Size:**

10,000 to >1,000,000 individuals (worldwide) (NatureServe 2015). Northern Rocky Mountain DPS (western) As of December 2014, the minimum year-end wolf population in the Northern Rocky Mountain DPS totaled 1,782 individuals, with a state-by-state breakdown as follows: Idaho – 770 individuals; Montana – 554 individuals; Oregon – 77 individuals; Washington – 48 individuals; and Wyoming – 333 individuals (USFWS 2015). Western Great Lakes DPS (eastern) The minimum year-end wolf population in the Western Great Lakes DPS totaled 3,722

individuals, with a state-by-state breakdown as follows: Michigan (Upper Peninsula – late winter 2013-14) – 636 individuals; Isle Royale (January 2015) – 3 individuals; Minnesota (2013-2014) – 2,423 individuals; and Wisconsin (late winter 2013-14) – 660 individuals (USFWS 2014).

**Resistance to Disease:**

High

**Adaptability:**

High

**Additional Population-level Information:**

Because the species is wide-ranging, it is difficult to estimate the number of distinct occurrences (NatureServe 2015).

**Population Narrative:**

There are two populations of the gray wolf in the conterminous United States, not including the Mexican gray wolf: northern Rocky Mountains and Western Great Lakes (USFWS 2012). There are anywhere between 81 and more than 300 occurrences; however, because the species is wide-ranging, it is difficult to estimate the number of distinct occurrences (NatureServe 2015). The gray wolf has rebounded from the brink of extinction to exceed population targets by as much as 300 percent. Today, there are at least 5,521 gray wolves in the contiguous United States. Wolf numbers continue to be robust, stable, and self-sustaining (USFWS 2015). Outside the Northern Rocky Mountain DPS, there are 20 individuals in Washington. Population numbers and trends for each DPS are provided below (USFWS et al. 2015). Northern Rocky Mountain DPS (western) As of December 2014, the minimum year-end wolf population in the Northern Rocky Mountain DPS totaled 1,782 individuals, with a state-by-state breakdown as follows: Idaho – 770 individuals; Montana – 554 individuals; Oregon – 77 individuals; Washington – 48 individuals; and Wyoming – 333 individuals (USFWS et al. 2015). In the Northern Rocky Mountains DPS, state population trends are as follows: Idaho – increasing (IDFG 2015); Montana – decreasing (Bradley et al. 2015); Wyoming – increasing (WGFD et al. 2015); Washington – increasing (Becker et al. 2015); and Oregon – increasing (ODFW 2015). Western Great Lakes DPS (eastern) The minimum year-end wolf population in the Western Great Lakes DPS totaled 3,722 individuals, with a state-by-state breakdown as follows: Michigan (Upper Peninsula – late winter 2013-14) – 636 individuals; Isle Royale (January 2015) – 3 individuals; Minnesota (2013-2014) – 2,423 individuals; and Wisconsin (late winter 2013-14) – 660 individuals (USFWS 2014). In the Western Great Lakes DPS, state population trends are as follows: Minnesota – declining-to-stable due to hunting and depredation control; Wisconsin – declining due to human-caused mortality; Michigan – declining for unstated reasons. Mexican Gray Wolf-Experimental Population As of December 2014, there were 109 individuals in Arizona and New Mexico (USFWS et al. 2015).

**Threats and Stressors**

**Stressor:** See narrative.

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Recent analysis of the current threats and stressors to the species are not available. In an effort to identify the factors that may affect the species, the threats and stressors identified in

the Revised Recovery Plan for the Eastern Timber Wolf and Northern Rocky Mountain Recovery Plan are summarized below (USFWS 1987; USFWS 1992). Five main factors are critical to the long-term survival of the eastern timber wolf: (1) large tracts of wild land with low human densities and minimal accessibility by humans; (2) ecologically sound management; (3) availability of adequate wild prey; (4) adequate understanding of wolf ecology and management; and (5) maintenance of populations that are either free of, or resistant to, parasites and diseases new to wolves, or are large enough to successfully contend with their adverse effects (USFWS 1987).

**Stressor:** Development

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Development has multiple effects on wolves: (1) increased human presence increases the chance of direct killing of wolves; (2) although undocumented, unnatural structures, sounds, and smells might deter wolves from inhabiting an area; (3) artificial corridors such as paved roads, powerlines, fences along interstate highways, and railroads may prevent or minimize dispersal; (4) increased human presence increases chances of introducing new diseases and parasites to wolves via pets; and (5) reduced prey species abundance and diversity reduce wolf food supply (USFWS 1987).

**Stressor:** Human density and accessibility

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Nowhere in the United States is there an area where the eastern timber wolf will not be affected by human activity. Wherever people reside in wolf country, they will have domestic livestock and/or pets that may be subject to wolf attack. Public education about the wolf, and the preservation of large tracts of wild land with low human densities and minimal accessibility, will help preserve the wolf. Human activity and exploitation of wildlife increase with accessibility. This is especially true for wolves, which are strongly affected by roads in the following ways: (1) direct mortality via vehicles; (2) roads allow access by hunters and trappers, some of whom deliberately and/or accidentally kill wolves; and (3) major highways are barriers to dispersal (USFWS 1992).

**Stressor:** Ecological sound management

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Ecologically sound management includes: (1) protection where needed to help restore the eastern timber wolf to areas of its original range and to preserve a naturally functioning population that can serve as a living museum, as a scientific subject, and as a reservoir to repopulate adjacent areas; (2) depredation control where wolves are killing domestic animals; (3) restocking of wolves into suitable areas of their former range, when feasible; (4) continued research and monitoring of wolf populations; and (5) provision of adequate prey diversity and numbers through habitat and population management and reintroductions where appropriate (USFWS 1992). The USFWS recommends that in Michigan and Wisconsin, and in Zone 1, 2, 3, and 4 of Minnesota, strict protection should be afforded the wolf. Legal protection, however, is only as effective as the public acceptance of laws and regulations needed for wolf management, and

the degree of law enforcement devoted to it. Law enforcement is especially needed during fall and winter hunting and trapping seasons, generally September through March. Besides more rigorous and timely enforcement of the laws actually protecting the wolf, additional enforcement is also necessary to ensure that vehicles, including off-road vehicles, be kept off roads restricted against their use. Even the regular presence of law enforcement agents in wolf areas is a valuable deterrent to violations (USFWS 1992).

**Stressor:** Wild prey

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The wolf is dependent on a continual supply of deer, moose, and beaver. Therefore, one of the most important aspects of this plan is to maintain habitat in a high carrying capacity for prey. The most feasible method of doing this is through commercial and noncommercial timber sales and habitat improvement projects for these species. Such programs require temporary roads, but these can later be obliterated or gated. In protected areas such as Voyageurs National Park or the Boundary Waters Canoe Area where timber sales are prohibited or restricted, the prescribed use of fire may produce the mosaic of habitats necessary for a diversity of prey species (USFWS 1992).

**Stressor:** Public education

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Because of the degree of misunderstanding about wolf ecology, population dynamics, and management, concerted efforts aimed at providing public information and education have been implemented. Nevertheless, considerable misinformation still exists among several segments of the Minnesota and Michigan population. Therefore, concerted information and education are still strongly needed (USFWS 1987).

**Stressor:** Disease and parasites

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** In recent years, a number of new diseases and parasites have been clearly documented as occurring in wolf populations in Minnesota, Wisconsin, and Michigan. Heartworm, canine parvovirus (CPV), and Lyme disease each have the potential to become limiting factors acting on survival, reproduction, and dispersal of large numbers of wolves, and thus may determine the fate of isolated wolf populations. Wolf populations will be able to survive only if they are somehow able to contend with these new threats (USFWS 1987). Recent studies have shown that gray wolves, especially juveniles, are susceptible to CPV and distemper. Because survival of juvenile wolves is critical to successful recovery, developing a comprehensive health monitoring program for translocated and naturally reestablishing wolves is essential to minimize the risk of disease (USFWS 1992).

## ***Recovery***

**Reclassification Criteria:**

Consistent with assurances provided in the 1978 reclassification of the gray wolf in the conterminous United States (43 FR 9607, March 9, 1978), three gray wolf recovery programs in the following regions of the country were implemented: the Western Great Lakes (Minnesota, Michigan, and Wisconsin, administered by USFWS' Great Lakes, Big Rivers Region), the Northern Rocky Mountains (Idaho, Montana, and Wyoming, administered by USFWS' Mountain-Prairie Region and Pacific Region), and the Southwest (Arizona, New Mexico, Texas, Oklahoma, Mexico, administered by USFWS' Southwest Region). Recovery plans were developed in each of these areas to organize and prioritize recovery criteria and actions appropriate to the unique local circumstances of the gray wolf. Thus, the three gray wolf recovery programs have functioned independently from one another since their inceptions (USFWS 2012).

#### Western Great Lakes/Eastern Timber Wolf

The primary objective of the Recovery Plan for the Eastern Timber Wolf is to maintain and reestablish viable populations of the eastern timber wolf in as much of its former range as is feasible (USFWS 1992).

When condition 1 of the Delisting Criteria is met and there are 80 wolves (based on late winter counts) in Wisconsin for a minimum of 3 consecutive years, the eastern timber wolf should be downlisted to threatened in Wisconsin. At that time, consideration may also be given to the downlisting of the Michigan wolf population (USFWS 1992).

#### Northern Rocky Mountain

To reclassify the Northern Rocky Mountain wolf to threatened status over its entire range by securing and maintaining a minimum of 10 breeding pairs in each of two recovery areas for a minimum of 3 successive years (USFWS 1987).

To reclassify the Northern Rocky Mountain wolf to threatened status in an individual recovery area by securing and maintaining a minimum of 10 breeding pairs in the recovery area for a minimum of 3 successive years. Consideration will also be given to reclassifying such a population to threatened under similarity of appearance after this objective for the population has been achieved and verified, special regulations are established, and a state management plan is in place for that population (USFWS 1987).

#### Southwest/Mexican

Need to develop reclassification criteria.

#### **Delisting Criteria:**

Consistent with assurances provided in the 1978 reclassification of the gray wolf in the conterminous United States (43 FR 9607, March 9, 1978), three gray wolf recovery programs in the following regions of the country were implemented: the Western Great Lakes (Minnesota, Michigan, and Wisconsin, administered by USFWS' Great Lakes, Big Rivers Region), the Northern Rocky Mountains (Idaho, Montana, and Wyoming, administered by USFWS' Mountain-Prairie Region and Pacific Region), and the Southwest (Arizona, New Mexico, Texas, Oklahoma, Mexico, administered by USFWS' Southwest Region). Recovery plans were developed in each of these areas to organize and prioritize recovery criteria and actions appropriate to the unique local

circumstances of the gray wolf. Thus, the three gray wolf recovery programs have functioned independently from one another since their inceptions (USFWS 2012).

#### Western Great Lakes/Eastern Timber Wolf

The Revised Recovery Plan for the Eastern Timber Wolf identified the following two recovery criteria necessary to delist the species.

- 1) the survival of the wolf in Minnesota is ensured (USFWS 1992); and
- 2) at least one viable population of eastern timber wolves outside Minnesota and Isle Royale in the contiguous 48 states of the United States is reestablished (USFWS 1992).

A viable population of eastern timber wolves outside of Minnesota must meet one of the following two descriptions, based on late winter counts: 1. An isolated eastern timber wolf population in the United States must average at least one wolf per 50 sq. mi. (a self-sustaining population of at least 200 wolves) distributed in a minimum area of at least 25,600 contiguous km<sup>2</sup> (10,000 sq. mi.) of suitable habitat over a period of 5 successive years; or 2. An eastern timber wolf population in the United States, located within 160 kilometers (100 miles [mi.]) of a self-sustaining wolf population (as described in item 1), must average at least one wolf per 128 km<sup>2</sup> (50 sq. mi.) or consist of 100 wolves distributed in an area of at least 12,800 contiguous km<sup>2</sup> (5,000 sq. mi.) of suitable habitat over a period of 5 consecutive years. These 100 wolves do not have to be evenly distributed (USFWS 1992).

#### Northern Rocky Mountain

To remove the Northern Rocky Mountain wolf from the endangered and threatened species list by securing and maintaining a minimum of 10 breeding pairs in each of the three recovery areas for a minimum of 3 successive years (USFWS 1987).

#### Southwest/Mexican

Need to develop delisting criteria.

#### **Recovery Actions:**

- Consistent with assurances provided in the 1978 reclassification of the gray wolf in the conterminous United States (43 FR 9607, March 9, 1978), three gray wolf recovery programs in the following regions of the country were implemented: the Western Great Lakes (Minnesota, Michigan, and Wisconsin, administered by USFWS' Great Lakes, Big Rivers Region), the Northern Rocky Mountains (Idaho, Montana, and Wyoming, administered by USFWS' Mountain-Prairie Region and Pacific Region), and the Southwest (Arizona, New Mexico, Texas, Oklahoma, Mexico, administered by USFWS' Southwest Region). Recovery plans were developed in each of these areas to organize and prioritize recovery criteria and actions appropriate to the unique local circumstances of the gray wolf. Thus, the three gray wolf recovery programs have functioned independently from one another since their inceptions (USFWS 2012).
- Western Great Lakes/Eastern Timber Wolf

- The Revised Recovery Plan for the Eastern Timber Wolf provided the following recovery actions.
- Ensure perpetuation of the eastern timber wolf population at levels optimal to the various parts of its present Minnesota range (optimum level includes biological carrying capacity and compatibility with humans): Zone 1, to fluctuate naturally; Zones 2 and 3, one wolf per 10 sq. mi., Zone 4, one wolf per 50 sq. mi., Zone 5, no wolves (USFWS 1992).
- Enhance and reestablish a viable wolf population in Michigan (excluding Isle Royale) and Wisconsin (USFWS 1992).
- Continue management to perpetuate natural conditions for the eastern timber wolf on Isle Royale National Park, Michigan (USFWS 1992).
- Reestablish the wolf population in Adirondack Mountains (New York), northwestern Main/adjacent New Hampshire, and/or northeastern Maine (USFWS 1992).
- Create a Coordination Committee of state and federal representatives to implement the Eastern Timber Wolf Recovery Plan (USFWS 1992).
- Northern Rocky Mountain
- The Northern Rocky Mountain Wolf Recovery Plan provided the following recovery actions.
- Determine the present status and distribution of gray wolves in the Northern Rocky Mountains, and devise a systematic approach for compiling observations and other data on the Northern Rocky Mountain wolf (USFWS 1987).
- Evaluate and verify the population goals for a threatened and fully recovered population established in the current objectives (USFWS 1987).
- Delineate recovery areas and identify and develop conservations strategies and management plan(s) to ensure perpetuation of the Northern Rocky Mountain wolf (USFWS 1987).
- Monitor gray wolf populations, habitat, and prey (USFWS 1987).
- Develop and initiate information and education programs (USFWS 1987).
- Southwest/Mexican
- Inventory and evaluate remaining gene pool (USFWS 1982).
- Protect remaining gene pool (USFWS 1982).
- Reestablish and maintain viable wild populations of Mexican wolves in at least two areas in Mexico and/or adjoining areas of southwestern United States (USFWS 1982).
- If efforts fail to establish and maintain viable wild populations of Mexican wolves anywhere in Mexico or the United States, declare the subspecies extinct in the wild and maintain remaining captive Mexican wolves in captivity, managing captive populations so as to prevent extinction of the subspecies and, if possible, genetic degeneration. For this task, the exact mechanisms and assignment of responsibilities are to be determined at the time by agreement between USFWS and Dirección General de la Fauna Silverstre after recommendations are obtained from the Mexican Wolf Recovery Team, American Association of Zoological Parks and Aquariums, and International Species Inventory Systems (USFWS 1982).
- Monitor progress of agencies, groups, and individuals with assigned responsibilities to ensure that tasks are accomplished in the recommended order of priorities and by the target dates (USFWS 1982).

***Conservation Measures and Best Management Practices:***

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***Additional Threshold Information:***

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## SPECIES ACCOUNT: *Canis lupus baileyi* (Mexican gray wolf)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered/Experimental Population; 04/28/1976, 01/24/1998; Southwest Region (R2) (USFWS, 2016)

### **Physical Description**

The Mexican wolf is the smallest extant gray wolf in North America. Adults weigh 23 to 41 kilograms (50 to 90 pounds), with a length of 1.5 to 1.8 meters (m) (5 to 6 feet [ft.]) and height at shoulder of 63 to 81 centimeters (25 to 32 inches). Mexican wolves are typically a patchy black, brown-to-cinnamon, and cream color, with primarily light underparts. Solid black or white coloration, as seen in other North American gray wolves, does not exist in Mexican wolves (80 FR 2488).

### **Taxonomy**

The Mexican wolf subspecies, *Canis lupus baileyi*, was originally described in 1929 as *Canis nubilus baileyi*, with a distribution of “southern and western Arizona, southern New Mexico, and the Sierra Madre and adjoining tableland of Mexico as far south, at least, as southern Durango.” In the first comprehensive treatment of North American wolves in 1944, *C. n. baileyi* was renamed as a subspecies of *lupus* (i.e., *C. l. baileyi*). Since that time, gray wolf taxonomy has undergone substantial revision, including a major taxonomic revision in which the number of recognized gray wolf subspecies in North America was reduced from 24 to 5, with the Mexican wolf, *C. l. baileyi*, being recognized as a subspecies ranging throughout most of Mexico to just north of the Gila River in southern Arizona and New Mexico. The Mexican wolf is considered a morphologically distinct and valid subspecies, based on skull morphometrics and unique genetic markers (80 FR 2488).

### **Historical Range**

Prior to the late 1800s, the Mexican wolf inhabited the southwestern United States and Mexico. In Mexico, Mexican wolves ranged from the northern border of the country, southward through the Sierra Madre Oriental and Occidental and the altiplano (high plains), to the Neovolcanic Axis (a volcanic belt that runs east-west across central-southern Mexico), although wolf distribution may not have been continuous through this entire region. The Mexican wolf is the only subspecies known to have inhabited Mexico. In the United States, Mexican wolves (and, in some areas, *Canis lupus nubilus* and the previously recognized subspecies *C. l. monstrabilis*, *C. l. mogollonensis*, and *C. l. youngi*) inhabited montane forests and woodlands in portions of New Mexico, Arizona, and Texas. In southern Arizona, Mexican wolves inhabited the Santa Rita, Tumacacori, Atascosa-Pajarito, Patagonia, Chiricahua, Huachuca, Pinaleno, and Catalina Mountains, west to the Baboquivaris and east into New Mexico. In central and northern Arizona, the Mexican wolf and other subspecies of gray wolf were interspersed. The Mexican wolf and other subspecies were present throughout New Mexico, with the exception of low desert areas, documented as numerous or persisting in areas including the Mogollon, Elk, Tularosa, Diablo, Pinos Altos, Black Range, Datil, Gallinas, San Mateo, Mount Taylor, Animas, and Sacramento mountains (80 FR 2488).

### **Current Range**

It is estimated that breeding populations of Mexican wolves were extirpated from the United States by 1942. Poisons and trapping techniques during the 1950s and 1960s eliminated remaining Mexican wolves north of the United States-Mexico border, although occasional reports of wolves crossing into the United States from Mexico persisted into the 1960s. Wolf distribution in northern Mexico contracted to encompass the Sierra Madre Occidental in Chihuahua, Sonora, and Durango, as well as a disjunct population in western Coahuila (from the Sierra del Carmen westward). When the Mexican wolf was listed as endangered under the Endangered Species Act (ESA) in 1976, no wild populations were known to remain in the United States or Mexico. A survey to determine the status and distribution of wolves in Mexico in 1977 identified three general areas where wolves were recorded as still present in the Sierra Madre Occidental: (1) Northern Chihuahua and Sonora border (at least eight wolves); (2) western Durango (at least 20 wolves in two areas); and (3) a small area in southern Zacatecas. Although occasional anecdotal reports have been made during the last three decades that a few wild wolves still inhabit forested areas in Mexico, no publicly available documented verification exists. Several individual Mexican wolves captured in the wild in Mexico became the basis for the captive-breeding program that has enabled the reintroduction to the wild. As a result of the Mexican Wolf Experimental Population Area captive breeding program, Mexican wolves have been released or translocated in central Arizona, New Mexico, and a small portion of northwestern Texas. As of 2013, a single wild population of a minimum of 83 Mexican wolves inhabited the United States in central Arizona and New Mexico (80 FR 2488).

**Critical Habitat Designated**

Yes;

***Life History*****Feeding Narrative**

Adult: Mexican wolves are carnivores whose prey include medium to large ungulates like elk (*Cervus* spp.), white-tailed deer (*Odocoileus virginianus*), and mule deer (*Odocoileus hemionus*) (80 FR 2488). When ungulate populations are low or seasonally unavailable, wolves may eat alternative prey such as lagomorphs (hares and rabbits), rodents, and carrion. Reintroduced wolves in Arizona and New Mexico subsist primarily on elk and sometimes take livestock, deer, rodents, or lagomorphs (NatureServe 2015). Mexican wolves hunt in packs during nighttime/crepuscular hours, and wolf density is positively correlated with the amount of ungulate biomass available and the vulnerability of ungulates to predation (80 FR 2488).

**Reproduction Narrative**

Adult: Mexican wolves are monogamous, breeding once per year with an average of four to six pups (up to eleven) per season. Courtship begins between January and April, and after a gestation period of 63 days, the young are born blind in a den, typically a hole in the ground, or in a rock crevice, hollow log, or overturned stump. Pups are weaned in about 5 to 7 weeks, and young leave the den at about 3 months old. Some offspring remain with the pack, others disperse as they mature. Adults reach sexual maturity during their second or third year, and live for up to 10 to 16 years of age in the wild. Mexican wolves have a high fitness, with an average reproductive capacity of 35 offspring in a lifetime (NatureServe 2015; Mech 1974; USFWS 1982).

**Geographic or Habitat Restraints or Barriers**

Adult: Occur only where human population density and persecution level are low and prey densities are high (NatureServe 2015).

**Spatial Arrangements of the Population**

Adult: Clumped

**Environmental Specificity**

Adult: Broad/generalist where key requirements are present.

**Site Fidelity**

Adult: Moderate

**Dependency on Other Individuals or Species for Habitat**

Adult: Abundance of ungulate prey (80 FR 2488).

**Habitat Narrative**

Adult: Historically, Mexican wolves occupied montane woodlands characterized by sparsely to densely forested mountainous terrain consisting of evergreen oaks (*Quercus* spp.) or pinyon (*Pinus edulis*) and juniper (*Juniperus* spp.) to higher elevation pine (*Pinus* spp.), mixed-conifer forests, and adjacent grasslands at elevations of 1,219 to 1,524 m (4,000 to 5,000 ft.) (80 FR 2488). In the present-day, Mexican wolves in Arizona and New Mexico inhabit evergreen pine-oak woodlands (i.e., Madrean woodlands), pinyon-juniper woodlands (i.e., Great Basin conifer forests), and mixed-conifer montane forests (i.e., Rocky Mountain, or petran, forests) (80 FR 2488). Mexican wolves are generally found only where human population density and persecution level are low and prey densities of ungulates are high (NatureServe 2015).

***Dispersal/Migration*****Motility/Mobility**

Adult: High

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Mexican wolves probably are similar to other grey wolf subspecies, having an annual home range of up to several hundred square kilometers (km) (70 to 150 square miles [mi.]); and occasionally moving several hundred km (60 to 180 mi.), especially when dispersing young. Historical information indicates that a hunting runway of 113 km (70 mi.) would be traversed about every 9 days (80 FR 2488). Like gray wolves, young Mexican wolves disperse from natal to new territories between the ages of 1 and 2 years, typically between February and April, and October and November (NatureServe 2015).

**Immigration/Emigration**

Adult: Unlikely

**Dependency on Other Individuals or Species for Dispersal**

Adult: No

**Dispersal/Migration Narrative**

Adult: Mexican wolves probably are similar to other grey wolf subspecies, having an annual home range of up to several hundred square km (70 to 150 square mi.); and occasionally moving several hundred km (70 to 150 square mi.), especially when dispersing young. Historical information indicates that a hunting runway of 113 km (70 mi.) would be traversed about every 9 days. Like gray wolves, young Mexican wolves disperse from natal to new territories between the ages of 1 and 2 years, typically between February and April, and October and November (NatureServe 2015).

***Population Information and Trends*****Population Trends:**

Decreasing

**Species Trends:**

Decreasing

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Population Growth Rate:**

Slow

**Number of Populations:**

When the Mexican wolf was listed as endangered under the ESA in 1976, no wild populations were known to remain in the United States or Mexico. A survey to determine the status and distribution of wolves in Mexico in 1977 identified three general areas where wolves were recorded as still present in the Sierra Madre Occidental: (1) Northern Chihuahua and Sonora border (at least eight wolves); (2) western Durango (at least 20 wolves in two areas); and (3) a small area in southern Zacatecas. Although occasional anecdotal reports have been made during the last three decades that a few wild wolves still inhabit forested areas in Mexico, no publicly available documented verification exists. Several individual Mexican wolves captured in the wild in Mexico became the basis for the captive-breeding program that has enabled the reintroduction to the wild. As a result of the Mexican Wolf Experimental Population Area captive breeding program, Mexican wolves have been released or translocated in central Arizona, New Mexico, and a small portion of northwestern Texas. As of 2013, a single wild population of a minimum of 83 Mexican wolves inhabited the United States in central Arizona and New Mexico (80 FR 2488).

**Population Size:**

In 2012, fewer than 100 individuals were recorded in the wild in Arizona and New Mexico. The total wild population (including Mexico) is estimated to be between 50 and 250 individuals (NatureServe 2015).

**Resistance to Disease:**

Low

**Adaptability:**

Moderate

**Additional Population-level Information:**

Wild populations are being supported through the establishment and success of an ongoing captive-breeding program.

**Population Narrative:**

When the Mexican wolf was listed as endangered under the ESA in 1976, no wild populations were known to remain in the United States or Mexico. A survey to determine the status and distribution of wolves in Mexico in 1977 identified three general areas where wolves were recorded as still present in the Sierra Madre Occidental: (1) Northern Chihuahua and Sonora border (at least eight wolves); (2) western Durango (at least 20 wolves in two areas); and (3) a small area in southern Zacatecas. Although occasional anecdotal reports have been made during the last three decades that a few wild wolves still inhabit forested areas in Mexico, no publicly available documented verification exists. Several individual Mexican wolves captured in the wild in Mexico became the basis for the captive-breeding program that has enabled the reintroduction to the wild. Nevertheless, the Mexican wolf is showing decreasing population and species-level trends as a result of illegal shooting and small population sizes. As a result of the Mexican Wolf Experimental Population Area captive breeding program, Mexican wolves have been released or translocated in central Arizona, New Mexico, and a small portion of northwestern Texas. As of 2013, a single wild population of a minimum of 83 Mexican wolves inhabited the United States in central Arizona and New Mexico. However, these wild populations are being supported through the establishment and success of the ongoing captive-breeding program (80 FR 2488). Total wild population (including Mexico) is estimated to be between 50 and 250 individuals (NatureServe 2015). Given the threats of illegal shooting, inbreeding, loss of adaptive potential, loss of heterozygosity, and small population size, the species shows a low resilience to withstand stochastic events, has a low representation to adapt to changing environmental conditions across the landscape, a low redundancy to withstand catastrophic events, a low resistance to disease, and moderate adaptability.

***Threats and Stressors***

**Stressor:** Illegal Shooting

**Exposure:** Direct

**Response:** Increased incidence of mortality in all life stages.

**Consequence:** Reduction in population numbers, decreased reproductive success, increased genetic effects of population bottleneck, higher susceptibility to mortality/extirpation.

**Narrative:** Illegal mortalities as a result of shooting are one of the biggest threats the Mexican wolf faces. Death by illegal shooting can be intentional or accidental (mistaken for coyote), but

fluctuates between zero to as much as 15 percent of the known population in a given year (80 FR 2488).

**Stressor:** Inadequacy of Existing Regulatory Mechanisms

**Exposure:** Indirect

**Response:** Increased incidence of hunting/killing, increased incidence of mortality in all life stages.

**Consequence:** Reduction in population numbers, decreased reproductive success, increased genetic effects of population bottleneck, higher susceptibility to mortality/extirpation.

**Narrative:** Regulatory mechanisms to prohibit and penalize illegal killing exist under the ESA, but illegal shooting of wild Mexican wolves in the United States persists. The U.S. Fish and Wildlife Service concludes that, absent the protection of the ESA, killing of wolves in the United States would increase, potentially drastically, because state penalties are less severe than current federal penalties. In regard to regulatory protection for the Mexican wolf in Mexico, the recent poisoning of several reintroduced wolves suggests that illegal killing may be a challenge for that country's reintroduction efforts as well. Therefore, in the absence of the ESA, existing regulatory mechanisms will not act as an effective deterrent to the illegal killing of Mexican wolves in the United States, and this inadequacy will significantly affect the Mexican wolf (80 FR 2488).

**Stressor:** Inbreeding, Loss of Adaptive Potential, Loss of Heterozygosity, and Small Population Size

**Exposure:** Indirect/direct

**Response:** Increased decline in fitness/viability.

**Consequence:** Reduction in population numbers, decreased reproductive success, increased genetic effects of population bottleneck, higher susceptibility to mortality/extirpation.

**Narrative:** Inbreeding and loss of heterozygosity have the potential to affect viability-related fitness traits in Mexican wolves and, therefore, to affect the persistence of the subspecies in the wild in the near term; loss of genetic variation (adaptive potential) significantly affects the likelihood of persistence of the Mexican wolf over longer time frames (80 FR 2488).

## ***Recovery***

### **Reclassification Criteria:**

Option 1: The Mexican wolf will be considered for downlisting when: a) The United States population average over a 4-year period is greater than or equal to 320 Mexican wolves; and b) Gene diversity available from the captive population has been incorporated in the United States population through the scheduled releases of wolves surviving to breeding age as identified in delisting criteria (USFWS 2018).

Option 2: The Mexican wolf will be considered for downlisting when a minimum of two populations (one in the United States and one in Mexico) meet abundance and genetic criteria as follows: a) Each population average over the same 4-year period is greater than or equal to 150 wolves with an annual positive population growth rate; and b) Gene diversity available from the captive population has been incorporated into both the United States and Mexico populations through the scheduled releases of wolves surviving to breeding age as identified in delisting criteria (USFWS 2018).

### **Delisting Criteria:**

The Mexican wolf will be considered for delisting when (USFWS 2017): 1) A minimum of two populations meet all abundance and genetic criteria as follows: United States a) The population average over an 8-year period is greater than or equal to 320 wolves (e.g., annual wolf abundance of 200, 240, 288, 344, 412, 380, 355, and 342 averages 320 wolves); b) The population must exceed 320 wolves each of the last 3 years of the 8- year period; c) The annual population growth rate averaged over the 8-year period is stable or increasing (e.g., annual averages of 1.2, 1.2, 1.2, 1.2, 1.2, 0.9, 0.9, and 1.0 averages 1.1); and d) Gene diversity available from the captive population has been incorporated into the United States population through scheduled releases of a sufficient number of wolves to result in 22 released Mexican wolves surviving to breeding age in the United States population. "Surviving to breeding age" means a pup that lives two years to the age of breeding or an adult or subadult that lives for a year following its release. "Scheduled releases" means captive releases and translocations that achieve genetic representation, as described in Rationale for Recovery Criteria. Mexico a) The population average over an 8-year period is greater than or equal to 200 wolves; b) The population must exceed 200 wolves each of the last three years of the 8- year period; c) The annual population growth rate averaged over the 8-year period is stable or increasing; and d) Gene diversity available from the captive population has been incorporated into the Mexico population through scheduled releases of a sufficient number of wolves that results in 37 released Mexican wolves surviving to breeding age in the Mexico population. "Surviving to breeding age" means a pup that lives 2 years to the age of breeding or an adult or subadult that lives for a year following its release. "Scheduled releases" means captive releases and translocations that achieve genetic representation, as described in Rationale for Recovery Criteria. -and- 2) States and Tribes will ensure regulatory mechanisms are in place to prohibit or regulate human-caused mortality of Mexican wolves in those areas necessary for recovery such that the Service determines at least 320 Mexican wolves are likely to be maintained in the United States in the absence of Federal ESA protections. In addition, Mexico will ensure regulatory mechanisms are in place to protect Mexican wolves from human-caused mortality, such that the Service determines at least 200 Mexican wolves are likely to be maintained in Mexico. To ensure we are making expeditious progress toward recovery, we will evaluate our progress at 5 and 10 years after implementation of the recovery plan and subsequently adjust our management as needed. In addition, we will conduct 5-year species status reviews required under the Section 4(c)(2) of the ESA. We developed recovery actions for the Mexican wolf for each objective, which include: surveying and monitoring Mexican wolf populations in the wild; conducting releases (including cross-fostering) and translocations of Mexican wolves; reducing human-caused mortality of Mexican wolves; reducing Mexican wolf-livestock conflicts; developing and implementing plans for releases, cross-fostering, and translocations; monitoring and managing Mexican wolf health and genetic health; maintaining habitat; maintaining and enhancing connectivity within and between Mexican wolf populations; maintaining and improving the status of native prey populations; managing the Mexican wolf captive breeding population; conducting education and outreach; managing the recovery program; coordinating binational recovery efforts; and developing adequate regulations and management and monitoring plans to maintain viable Mexican wolf populations after delisting. We expect to recover the Mexican wolf within 25-35 years. The total estimated cost of implementing this plan through year 2043, the estimated recovery date of the Mexican wolf, is \$178,439,000. The estimated cost to implement the first 5 years of recovery actions (i.e., intermediate steps toward the goal of recovery) is \$38,455,000. This cost includes those borne by governmental agencies and nongovernmental organizations in the United States and Mexico.

**Recovery Actions:**

- Inventory and evaluate remaining gene pool (USFWS 1982).
- Protect remaining gene pool (USFWS 1982).
- Re-establish and maintain viable wild populations of Mexican wolves in at least two areas in Mexico and/or adjoining areas of southwestern United States (USFWS 1982).
- If efforts fail to establish and maintain viable wild populations of Mexican wolves anywhere in Mexico or the United States, declare subspecies extinct in wild and maintain remaining captive Mexican wolves in captivity, managing captive populations so as to prevent extinction of the subspecies and, if possible, genetic degeneration. For this task, the exact mechanisms and assignment of responsibilities are to be determined at the time by agreement between U.S. Fish and Wildlife Service and Dirección General de la Fauna Silvestre after recommendations are obtained from the Mexican Wolf Recovery Team, American Association of Zoological Parks and Aquariums, and International Species Inventory Systems (USFWS 1982).
- Monitor progress of agencies, groups, and individuals with assigned responsibilities to ensure that tasks are accomplished in the recommended order of priorities and by the target dates (USFWS 1982).
- Our recovery goal is to conserve and protect the Mexican wolf and its habitat so that its long-term survival is secured, populations are capable of enduring threats, and it can be removed from the list of threatened and endangered species. Recovery objectives for the Mexican wolf are (USFWS 2017): 1. Increase the size of two Mexican wolf populations; 2. Improve gene diversity and maintain the health of Mexican wolves; 3. Ensure adequate habitat availability to support viable Mexican wolf populations; 4. Maintain the Mexican Wolf Species Survival Plan (SSP) captive breeding program to improve the status of wild populations; 5. Promote Mexican wolf conservation through education and outreach programs; and 6. Ensure recovery success.

***Conservation Measures and Best Management Practices:***

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***Additional Threshold Information:***

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## SPECIES ACCOUNT: *Canis rufus* (Red wolf)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered/Experimental Population, Non-Essential; 03/11/1967, 11/19/1986; Southeast Region (R4) (USFWS, 2016)

### **Physical Description**

The dorsal pelage is mainly gray with interspersed blackish hairs and sometimes yellowish or reddish hairs, especially on the legs and underparts. Nose pad is more than 25 mm wide. Total length 136-165 cm (Whitaker 1996, Whitaker and Hamilton 1998). LENGTH:165 cm. WEIGHT: 40900 g (NatureServe, 2015).

### **Taxonomy**

Based on recent genetic studies, Wilson et al. (2000) concluded that the eastern timber wolf (*Canis lupus lycaon*) and the red wolf (*Canis rufus*) are sister taxa and are best considered to be conspecific. Additionally, Wilson et al. found that these two taxa form a North American lineage with the coyote (*Canis latrans*) that is distinct from that of the gray wolf (*Canis lupus*), which is Eurasian in origin. Wilson et al. (2000, 2003) proposed that the eastern timber wolf (*Canis lycaon*) be recognized as a species distinct from the gray wolf (*C. lupus*). In contrast, Nowak (2002) presented an analysis of cranial morphology of recent and Pleistocene *Canis* and concluded that *Canis rufus* is a valid species and that *lycaon* may be a hybrid between *Canis rufus* and western *Canis lupus* (NatureServe, 2015).

### **Historical Range**

Delineation of the range of this wolf is hampered by a paucity of specimens and debate over the taxonomic status of the species (see Nowak 2002). Data presented by Nowak (2002) indicate a range extending from Maine to Florida and eastward (south of the Great Lakes) to Illinois, Oklahoma, and Texas, based on pre-1918 complete skulls (fragmentary archaeological and paleontological material dating back 10,000 years do not change this much). Basically, if the red wolf is regarded as a valid species, its range historically was essentially confined to the southeastern United States. Assuming the traditional view that the red wolf is a valid species and that it still exists in unhybridized form, the species was, until recent reintroductions, extinct in the wild since early 1980s (or mid-1970s, Rennie 1991). Formerly it was believed to have occurred from central Texas eastward to the coasts of Florida and Georgia and north to North Carolina, and along the Mississippi River Valley north to southern Illinois, and occasionally in Mexico. The last remnant population along Texas/Louisiana coast was rendered functionally extinct due to hybridization with coyote. A reintroduced population now occurs in an area of roughly 6,900 square kilometers in northeastern North Carolina (reintroduction in Great Smoky Mountains National Park failed and has been terminated; Federal Register, 8 October 1998) (NatureServe, 2015).

### **Current Range**

Propagation populations currently exist on two islands: Bulls Island, Cape Romain National Wildlife Refuge, South Carolina; and St. Vincent National Wildlife Refuge, Florida. Other red wolves exist in many captive-breeding facilities (NatureServe, 2015). The only population of red wolves known to exist in the wild on the Albemarle Peninsula of northeastern North Carolina (USFWS, 2007).

**Distinct Population Segments Defined**

No

**Critical Habitat Designated**

Yes;

***Life History*****Feeding Narrative**

Adult: Opportunistic. Diet consists of a variety of invertebrates and vertebrates (rabbits, rodents, deer, birds, etc.). Particularly favors marsh rabbits (*SYLVILAGUS AQUATICUS*), nutria (*MYOCASTOR COYPUS*), and carrion. Not considered a threat to livestock (does not hunt in packs), but may prey on unattended young calves, pigs, and barnyard fowl (Matthews and Moseley 1990). Adults and immatures are carnivorous, piscivorous, and invertivorous. It is primarily nocturnal (NatureServe, 2015).

**Reproduction Narrative**

Adult: Mates in January-February. Gestation lasts 60-63 days. Litter of 3-12 (average 6-7) is born in March-May. One litter per year. Young are born in a den in a hollow log, in a burrow, or in similar secluded sites (NatureServe, 2015). Age of breeding can be 2 years and up (USFWS, 2007). The sex ratio is biased, currently showing 14 percent more females than males. Red wolves are monestrous and typically persist as monogamous pairs (USFWS, 1989).

**Spatial Arrangements of the Population**

Adult: Small groups, scattered individuals (NatureServe, 2015)

**Environmental Specificity**

Adult: Broad (inferred from NatureServe, 2015)

**Habitat Narrative**

Adult: Suitable habitat for this habitat generalist includes upland and lowland forests, shrublands, and coastal prairies and marshes; areas with heavy vegetative cover. More social than coyote but less so than gray wolf; typically travels and forages in small family groups or alone. Formerly density probably not more than 1 per 2 sq. miles (NatureServe, 2015).

***Dispersal/Migration*****Motility/Mobility**

Adult: High (NatureServe, 2015))

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory (NatureServe, 2015)

**Dispersal**

Adult: High (NatureServe, 2015)

**Dispersal/Migration Narrative**

Adult: Home ranges variously reported as 65 to 130 square kilometers (Riley and McBride 1975), 117 (males) and 78 square kilometers (females, Carley 1979); 100-200 sq. km mentioned by Lowman 1975; varies with conditions. Wolves are highly mobile and readily disperse hundreds of kilometers. This species is non-migratory (NatureServe, 2015). Large areas of habitat of at least 170,000 acres in size are required by this species (USFWS, 1989).

### ***Population Information and Trends***

#### **Population Trends:**

Decline of >90% (NatureServe, 2015)

#### **Species Trends:**

Decreasing (USFWS, 2007)

#### **Resiliency:**

Given the very low numbers in the NEP (3 breeding pairs; N approximately 44), without substantial intervention (e.g., releases and management of coyote introgression), extirpation will likely occur within as few as eight years (Faust et al. 2016, p. 15). Faust et al. (2016, p. 3) suggested that the NEP could avoid extirpation and be viable (<10% chance of extirpation in 125 years) as a population with intervention, which might include reduction of the NEP mortality rate, increase in breeding rates (which would require reducing breeding season mortality), and releases from the Species Survival Plan (SSP) captive population for approximately 15 years followed by releases to maintain genetic health after that. However, the starting value (i.e., number of animals) for the population is now lower (44 wolves) than was initially modeled, and there is now an increased risk of stochastically-driven dynamics given the smaller population size (i.e., variability in the environment could have a stronger effect on the remaining population, than initially projected). All in all, without significant intervention, wild red wolves in the NEP could be extirpated in the near-term. If interventions described in Faust et al. (2016) are carried out, which could produce a viable population on the Albemarle Peninsula, substantial additional efforts and financial resources will be needed to facilitate population expansion in North Carolina. Modelling indicates landscape-level factors that affect habitat (e.g., particularly sea-level rise and increased flooding) will result in substantial changes to the habitat on the peninsula in the next 125 years, which could push wolves further west from where they currently occur. If this happens, they would encounter more development (e.g., Greenville area), as indicated by the urban development model results. Whether their natural mobility as a species will allow the red wolf to locate suitable habitat in a changing landscape is still unclear, but coyotes will likely use the same habitats and are more adaptable with regard to human development and infrastructure. Without sufficient wolf mates on the landscape, hybridization would likely continue to occur and coyotes already vastly outnumber wolves on both the peninsula and areas west of the current NEP so, intensive management and significant additional resources would be necessary. With regard to the SSP captive population, current gene diversity for the managed population is 88.87% and is equivalent to the genetic diversity of a population descended from only approximately five founders. This is one of the biggest challenges with this species because the current gene diversity is very low. The main objective for the captive population is to maintain this diversity in the long term. Faust et al. (2016, p. 3) discussed that “[w]hile the SSP [captive population] has been maintained at a relatively large population size of more than 150 animals for over 20 years, it needs to increase breeding and

increase its population size/space to ensure long-term viability and its ability to serve as a strong source for animals to release to the wild (USFWS 2018).”

**Representation:**

The SSP captive population represents the genetic fail-safe for the entire population and any future recovery potential for the species. However, only twelve of the original fourteen lines are still represented and Faust et al. (2016) provide several scenarios through which the SSP captive population could be expanded, genetic diversity (of the remaining 12 lines) maintained, and future release efforts supported. While any future reintroductions would require a consideration of SSP capacity to support these efforts, it is clear that the SSP captive population has maintained a genetically-diverse stock, within the limits of the remaining 12 founder lines, from which to grow the population and release into the wild. This report presents the best available scientific information to date on the status and management of the red wolf. This report is expected to be a living document that can be edited and peer-reviewed regularly to keep it current with the best available science. We expect to use this report for future recovery planning activities, management efforts, species status review (i.e., 5-year reviews), and other conservation activities that depend on the most current science (USFWS 2018).

**Redundancy:**

Redundancy is having sufficient numbers of resilient populations for the species to withstand catastrophic events. The single NEP of red wolves could be extirpated in approximately 8 to 37 years (Faust et al. 2016, p. 15). Without new reintroduction sites the species is unlikely to have significant redundancy in the wild. Some level of redundancy is present in captivity because the species is held at multiple facilities throughout the U.S. However, this does not constitute a viable wild population. Therefore, at present and into the future, there is no redundancy of red wolves in the wild (USFWS 2018).

**Population Growth Rate:**

8% per year (USFWS, 1989)

**Number of Populations:**

1 (USFWS, 2007)

**Population Size:**

100 - 130 (USFWS, 2007)

**Minimum Viable Population Size:**

2,000 (USFWS, 1989)

**Resistance to Disease:**

Low (inferred from USFWS, 2007; see threats)

**Adaptability:**

Low (inferred from NatureServe, 2015)

**Population Narrative:**

Assuming an historical range throughout southeastern North America (e.g., Nowak 2002), the extent of occurrence, area of occupancy, and abundance of this species have undergone a

drastic decline over the long term. This species has experienced a long-term decline of >90%. As of 2005, there were about 100 red wolves in the wild on 6,900 square kilometers in northeastern North Carolina. The number of reproductive individuals probably is not greater than 50. The captive population included around 165 wolves. This species occurs in the wild in one major location (plus two islands that serve as propagation areas). Genetic data (see Wayne and Jenks 1991, Wayne 1992, Wayne et al. 1998, Reich et al. 1999) indicate that existing populations of what have been called red wolves have no unique genetic characteristics and most likely are a product of hybridization between *Canis lupus* and *C. latrans*. A reintroduced population now occurs in an area of roughly 6,900 square kilometers in northeastern North Carolina. Attempted reintroduction in Great Smoky Mountains National Park failed, probably due to parvovirus and other common canine diseases, internal and external parasites, poor nutrition caused by low food availability, and predation (Federal Register, 8 October 1998) (NatureServe, 2015). There is only one population currently existing in the wild. The species status is declining based on the 2006 and 2007 Recovery Data Calls. Recent calendar year counts for red wolves in the wild population fluctuate between approximately 100 to 130 red wolves (USFWS, 2007). If the genetic models prescribe an  $N_e$  of 500 to achieve some set of genetic objectives, the MVP might have to be 2,000. While over the history of the population the average growth rate has been about 8 percent per year (i.e.,  $\lambda = 1.08$ ), it is expected that the potential for increase could be expanded to 20 percent per year (USFWS, 1989).

### ***Threats and Stressors***

**Stressor:** Habitat loss and fragmentation (USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** For centuries, fragmentation in red wolf historic range has come in the form of habitat conversion and land development by humans. Proposed development projects on the Albemarle Peninsula will have short-term and long-term effects on red wolves in the NEP unless potential effects are addressed early via planning, designs, and project implementation. Barriers to dispersal that fragment habitat (e.g., highways, airports, or large fenced areas) can have long-term effects upon genetic diversity. For restored populations of small size, such as the red wolf NEP, fragmenting barriers can magnify these genetic effects and potentially dampen or reverse population growth to a greater degree. Fragmentation contributed to the initial decline of the red wolf species. Now, fragmentation threatens red wolves in the North Carolina NEP via proposed barriers and habitat conversion on both public and private land (USFWS, 2007).

**Stressor:** Disease (USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Because canid diseases can spread quickly, they can cause serious setbacks in red wolf recovery. Canid diseases remain a serious threat to the red wolf NEP and to captive red wolves. Acton and colleagues found that titers against parvovirus are not detectable in a large portion of vaccinated red wolves, indicating the NEP is still very much at risk to CPV2 parvovirus. This is important because poor pup survival from parvovirus caused the Service in 1998 to discontinue the Great Smoky Mountains red wolf NEP (Henry 1998). Numerous diseases and other ailments have been documented during the past thirty years in individual red wolves. During 2007, the

Service observed eye entropion in three young captive program red wolves being held at Alligator River NWR. Other physical anomalies were observed in captive red wolves in recent years, such as progressive retinal atrophy, malocclusion and undescended testicles (Waddell, Pers. Comm. 2007). Heartworms, hookworms (*Ancylostoma caninum*), and sarcoptic mange, are serious concerns, but heartworms and hookworms have so far not been identified as a significant source of mortality in the NEP (USFWS 1990; Phillips and Scheck 1991). Tick paralysis was reported by Beyer and Grossman (1997), while Rothschild et al. (2001) reported arthritis, and Harrenstein et al. (1997) reported antibody responses to canine distemper and canine parvovirus indicating prior exposure. Penrose et al. (2000) reported the Lyme disease causing bacteria *Borrelia burgdorferi* in a red wolf. Neiffer et al. (1999) reported abdominal disease involving cecal inversion and colocolic intussusception. Kearns et al. (2000) reported dermatosis. Acton et al. (2000) surveyed necropsy results in 62 captive program red wolves for the period of 1992 to 1996. They documented numerous ailments in individual red wolves of many different ages. Of 22 neonatal deaths, major causes included parental trauma, parasitic pneumonia, and septicemia (systemic bacteria often found in the blood). Two juvenile red wolves died of cardiovascular anomalies or systemic parasitism. Of 38 adult red wolf deaths, causes included neoplasia and gastrointestinal diseases. Of the fatal neoplasm conditions, 50% were lymphosarcoma (USFWS, 2007).

**Stressor:** Gunshot and vehicle mortality (USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Gunshot mortality is a serious threat to red wolves in the North Carolina NEP. Preliminary figures generated in 2006 and 2007 (D. Murray unpublished data) showed that a wild red wolf is 7.2 times more likely to be killed by gunshot during the hunting season than during the nonhunting season. Whether accidental by licensed hunters, or illegal, gunshot mortality since 2004 is hampering the ability of the red wolf NEP to continue its upward trend in growth. Since 2004, gunshot mortality has reduced the number of breeding pairs and pups in the NEP and otherwise removed growth potential. Vehicle strike mortality significantly impacts the red wolf NEP in North Carolina. From 270 known red wolf mortalities recorded for the NEP between 1987 and 2006, vehicle mortality was calculated to be 17.4 percent (D. Murray 2007, unpublished data) (USFWS, 2007).

**Stressor:** Hybridization (USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Introgression of coyote genes continues to be a threat to the red wolf across its historic range (USFWS, 2007).

**Stressor:** Climate change (USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Natural weather events and global climate change will play growing roles in long-term survival and recovery of red wolves. The red wolf NEP in North Carolina is subject to annual tropical storm activity. In fact, Hurricane Isabel resulted in the deaths of two captive red wolves

during September of 2003, with no noticeable long-term impacts observed in the NEP. However, the NEP and associated prey species remain vulnerable to sea level rise and flooding related to climate change and hurricanes. Additional long-term changes in habitat availability, prey abundance, and other ecological or landscape factors will occur with climate change (Parry et al. 2007) (USFWS, 2007).

### ***Recovery***

#### **Reclassification Criteria:**

Not available

#### **Delisting Criteria:**

Establish and maintain at least three reintroduction projects within the historic range of the red wolf. This must be paralleled by the cooperation and assistance of at least 30 captive-breeding facilities in the United States. Human attitudes regarding red wolves must be addressed through education processes. The establishment of 220 red wolves in wild situations and the maintenance of 330 in captivity would provide for genetic stability and maintain the species. For the foreseeable future it is not considered feasible to either delist or downlist this species (USFWS, 1989).

#### **Recovery Actions:**

- Maintain and evaluate existing wild populations (USFWS, 1989).
- Establish new populations in the wild (USFWS, 1989).
- Expand captive-breeding capabilities (USFWS, 1989).
- Expand cryopreservation capabilities (USFWS, 1989).

### ***Conservation Measures and Best Management Practices:***

- Develop an effective disease prevention and management plan for red wolves and other canid species in northeastern North Carolina (USFWS, 2007).
- Expand the number of facilities participating in the Red Wolf Species Survival Plan to continue to meet genetic diversity objectives and to aid in establishing any future additional red wolf populations. Support Tacoma Metroparks and the Point Defiance Zoo and Aquarium in Washington with relocation and reconstruction of the flagship red wolf captive breeding facility located there. Enhance partnerships in the Red Wolf Species Survival Plan with staff at facilities across North America to enhance red wolf captive breeding (USFWS, 2007).
- Identify and evaluate land areas in red wolf historic range that could be considered for potential establishment of second and third wild red wolf populations. Examine biological and human factors important in identifying new restoration locations. Evaluate site selection concepts offered by states, scientists, and partners (Knowlton 2007 in litt.; Kyle et al. 2007; Van Manen et al. 2000; Defenders of Wildlife 2005 in litt.; Scott et al. 2005; Stoskopf 2007 in litt.; Murray 2007 in litt.; among others). Biologists have known since the first wolf was released in North Carolina and based on the recovery plan for the red wolf, that the species cannot be recovered by restoring it only to the Albemarle Peninsula. Before release of red wolves in North Carolina, the Service recognized the impacts this action would have and cooperated extensively with the State and local communities in order to be able to initiate an important recovery action while maintaining flexibility to ensure human safety and activities would be considered. One of the objectives to attain the red wolf's recovery is to restore and expand the red wolf into other suitable habitats within its historic range.

The Service's immediate focus is on its recovery efforts for the red wolf NEP. The Service would like to explore the feasibility of restoration of other populations and intends to work in cooperation with States, partners, and local communities (USFWS, 2007).

- Work collaboratively with the U.S. Department of Agriculture Wildlife Services in support of efforts by the NCWRC to develop a cooperative statewide canid management plan or policy. With NCWRC leadership, develop a plan or policy concurrent with developing new state and federal regulations which address the most pressing canid issues in the State of North Carolina. Include the issues of landowner needs, hunter stewardship, trapping opportunities, wolf management areas, and canid disease management. Focus on the illegal import, illegal release, and fox pen hunting of invasive eastern coyotes, with safeguards ensuring wolves are not hunted in fox pens. Focus on elimination of eastern coyotes from the Albemarle Peninsula to the extent feasible. Include in the cooperative plan provisions to effectively manage wolves, coyotes, wolf-dog hybrids, foxes and exotic variations of these animals (USFWS, 2007).
- Develop cooperative actions which result in significant reduction of the portion of red wolf mortality attributed indirectly or directly to people. Work with the North Carolina Wildlife Resources Commission and the North Carolina Department of Transportation to develop cooperative measures which reduce the loss of red wolves caused by gunshot and vehicles strikes. Develop and implement educational outreach measures to highlight to people and local communities we need their assistance in reducing red wolf mortality. Encourage managers of large development projects and partners on the Peninsula to work with us in incorporating red wolf recovery concerns. Develop mutually beneficial landowner incentive measures. Explore potential joint state and federal law enforcement measures (USFWS, 2007).
- Draft a new recovery plan and species survival plan for the red wolf. These plans should incorporate significant advances in science and information developed since approval of the 1990 Red Wolf Recovery/Species Survival Plan. The 1982, 1984 and 1990 plans were written to identify measures which ensure immediate survival of red wolves in captivity and in the red wolf NEP. Many tasks in these early plans associated with captive rearing and restoration into the wild are completed or ongoing with significant gains in survival pulling the red wolf away from the brink of extinction. After 20 years of restoration and management of red wolves in the wild and in captivity, we must set new recovery goals, objectives, criteria, tasks and research needs. These should focus on population management, restoration in historic range, expanded captive breeding, reduction of new threats, long-term conservation, delisting, and down-listing (USFWS, 2007).
- Establish a human dimensions sub-team and a community stakeholder group to advise the Service and Red Wolf Recovery Implementation Team scientists on human factors and issues important in successful red wolf recovery (USFWS, 2007).
- Maintain at least two locations which fulfill the vital restoration roles of island propagation sites that contribute directly to both wild red wolf population(s) and captive breeding. The two sites currently with such capabilities are St. Vincent NWR in Florida and Cape Romain NWR in South Carolina (USFWS, 2007).
- Launch studies of wolf/coyote interaction and monitoring to identify additional long-term strategies for wolf and coyote management, with focus on the western end of the red wolf NEP (USFWS, 2007).
- Consider updating the red wolf 4(d) rule in cooperation with the State to reflect additional strength and flexibility needed for landowners, land managers, hunters, trappers, communities, red wolves and law enforcement officers. Another option is to identify alternate conservation incentive agreements with land owners and land managers (USFWS, 2007).

- Engage further science in the discussion of relationships between red wolves and Algonquin wolves and whether or not they should be managed together across a broader geographic continuum (USFWS, 2007).
- Launch enhanced, expanded and new efforts to educate local communities and visitors about red wolf conservation and ecosystem values. Share red wolf conservation values with children, families, other stakeholders and the general public. Enhance partnerships developing ecotourism values for local communities proximal to the wild red wolf population(s). Assist partners in their efforts to promote ecotourism and establish an education center emphasizing red wolf, refuge, farming, hunting and other natural resource community values (USFWS, 2007).
- Evaluate how the effects of climate change will influence red wolf recovery. Develop plans which address the effects of climate change via strategies in long-term conservation (USFWS, 2007).
- Continue to implement and further develop the red wolf adaptive management plan for wild red wolf population(s), based on regular evaluations and recommendations by scientists from the Red Wolf Recovery Implementation Team (USFWS, 2007).

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## SPECIES ACCOUNT: *Corynorhinus (=Plecotus) townsendii ingens* (Ozark big-eared bat)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; 11/30/1979; Southwest Region (R2) (USFWS, 2016)

### **Physical Description**

The Ozark big-eared bat is medium-sized with distinctively large ears (30 – 39 mm; 1.2 – 1.5-inches long) that connect at the base across the forehead. The tragus (i.e., fleshy prominence in front of the external ear opening) is long (11 – 17mm; 0.43 – 0.67 inches) and pointed. Prominent lumps occur on either side of the face (Kunz and Martin, 1982). The long fur is light to dark brown on the back and paler tan underneath due to the brown base and tan to buff tip of the ventral hairs (Barbour and Davis, 1969; Kunz and Martin, 1982; Tumlison, 1995). The Ozark big-eared bat is the largest and reddest of the five subspecies of *C. townsendii*. The bat has a wingspan of 305 - 330 mm (12 - 13 inches), a forearm length of 39 - 48 mm (1.5 - 1.9 inches), and weighs from 5 - 13 grams (0.2 - 0.5 ounces) (Kunz and Martin, 1982). The toe hairs do not extend beyond the claws.

### **Taxonomy**

The Ozark big-eared bat belongs to the plain-nosed bat family, Vespertilionidae. The vesper bats are the second largest mammalian family after the Muridae (Old World rats and mice). The genus name of the Ozark big-eared bat at the time of listing was *Plecotus* based on the revised taxonomy of North American bats by Handley (1959). Handley determined that the three species of North American big-eared bats did not differ enough morphologically from the European species of the genus *Plecotus* to warrant unique generic status. The bats were considered members of the genus *Plecotus* and subgenus *Corynorhinus*. *Corynorhinus* was subsequently elevated from subgenera to full generic status and *Plecotus* was limited to species of the Palearctic as a result of additional studies based on morphology, karyotype, and mitochondrial DNA (Bogdanowicz et al., 1998; Fedyk and Ruprect, 1983; Qumsiyeh and Bickham, 1993; Stock, 1983; Tumlison and Douglas, 1992; Volleth and Heller, 1994). A recent study on the phylogeny of North American big-eared bats using mitochondrial and nuclear DNA sequences confirmed the designation of three *Corynorhinus* species and corroborates the subspecies classification *Corynorhinus townsendii ingens* (Piaggio and Perkins, 2005).

### **Current Range**

The Ozark big-eared bat was federally-listed as endangered in 1979 due to its small population size, reduced and limited distribution, and vulnerability to human disturbance. Disturbance of hibernating bats causes the loss of critical fat stores and increases the probability of starvation during the winter, while disturbance at maternity roosts can result in loss of young. The bat also is listed as endangered by the States of Oklahoma, Arkansas, and Missouri (although the species is believed to have been extirpated from Missouri). The Ozark big-eared bat is endemic to the Ozark Highlands and Boston Mountains ecoregions (Omernik, 1987) where it occurs in oak-hickory hardwood forests (Clark, 1991; Leslie and Clark, 2002; and U.S. Fish and Wildlife Service, 1995). The current range of the Ozark big-eared bat includes northeastern Oklahoma and northwestern Arkansas. In Oklahoma (i.e., the project area), Ozark big-eared bats currently are known to occur in Adair, Cherokee, and Sequoyah counties. They were historically known from

two caves in Delaware County, but have not been observed there recently. Twelve caves considered essential for the continued existence of the Ozark big-eared bat (i.e., used by colonies of Ozark big-eared bats for maternity sites and/or hibernacula) occur in Oklahoma. In Arkansas, the Ozark big-eared bat is known to occur in Crawford, Marion, Searcy, Washington, and Franklin counties. Seven essential caves occur in Arkansas. The Ozark Highlands ecoregion is under considerable development pressure and is one of the fastest growing areas in the country due to relatively inexpensive land prices and the aesthetics of the area. For example, the human population of Washington and Benton County, Arkansas, and Adair and Cherokee counties, Oklahoma, increased 39.0 percent, 59.0 percent, 14.2 percent, and 24.9 percent, respectively, from 1990 to 2000. Over the same period, the human population within the states of Oklahoma and Arkansas, and within the United States increased by only 9.7 percent, 13.7 percent, and 13.2 percent respectively (U.S. Census Bureau, 2001). The Oklahoma Department of Commerce (ODOC) projects the human population of Adair and Cherokee counties, Oklahoma, to grow by about 35 percent over the next 23 years (ODOC, 2002). Based on population estimates since 1997, when the most recently discovered essential maternity site was added to the annual monitoring efforts, the overall long-term population trend appears to be slightly increasing. However, during a five-year review on the current status of the Ozark big-eared, the Service (2008) determined that neither the down nor de-listing criteria identified in the current recovery plan (USWFS, 1995) had been met, and that significant threats to this species remain. Although additional essential caves have been discovered and protected since the time of listing, not all known caves have been afforded some form of protection (e.g., a cave gate/grill, signs, fee-title purchase, conservation easement, etc). Population trends of all individual colonies at essential caves are not well explained by available monitoring data.

**Critical Habitat Designated**

No;

***Life History*****Feeding Narrative**

Adult: They primarily feed on moths, but also are known to eat beetles and other flying insects (USFWS, 1995; Leslie and Clark, 2002; Dodd, 2006).

**Reproduction Narrative**

Adult: Ozark big-eared bats mate during fall and winter. Females become reproductively active during their first fall (Kunz and Martin, 1982; U. S. Fish and Wildlife Service, 1995), while young males do not reach sexual maturity until their second fall (Kunz and Martin, 1982). Females store sperm in their reproductive tract during the winter hibernation period.. Ozark big-eared bats give birth to a single offspring in May or June after a two-three month gestation period (Kunz and Martin, 1982; Clark et al., 2002). Young bats grow quite rapidly and are capable of flight at three weeks and are weaned by six weeks (Kunz and Martin, 1982). Maternity colonies begin to break up in August (Kunz and Martin, 1982; Clark et al., 1996; Wethington et al., 1996). Males are solitary during the summer maternity period (Kunz and Martin, 1982; Harvey and Barkley, 1990; Clark et al., 1993). Little else is known about their summer habitats (U. S. Fish and Wildlife Service, 1995). The maximum life span of the Ozark big-eared bat is estimated to be about 16 years based on recovery of banded bats (Paradiso and Greenhall, 1967; Harvey, 1992).

**Environmental Specificity**

Adult: Hibernacula and maternity caves

**Habitat Narrative**

Adult: The Ozark big-eared bat is an insectivorous bat that uses caves year-round. Colonies typically begin to form at hibernacula in October and November (Clark et al., 1996 and 2002). Both sexes hibernate together in clusters that typically range from 2 -135 individuals (Clark et al., 1993, 1997 and 2002). Ozark big-eared bats also will hibernate singly (Clark et al., 1996, 1997, and 2002) and in larger groups that have consisted of up to about 400 individuals. The Ozark big-eared bat is known to exhibit winter activity (Kunz and Martin, 1982; Clark et al., 2002). Insect activity typically is very low during cold nights. Winter activity, therefore, may not be for foraging. Activity likely occurs in order to relocate within the same hibernaculum or among hibernacula to find a more thermally stable location when temperatures at the initial location become too extreme (Kunz and Martin, 1982; Harvey and Barkely, 1990). Ozark big-eared bats also may be seeking open water to drink (Avery, 1985; Clark et al., 2002; Speakman and Racey, 1989). Hibernating colonies gradually begin to break up in spring from April through May (Clark et al., 2002). Females also become pregnant during this time (Kunz and Martin, 1982) and slowly begin to congregate at warm maternity caves to give birth and rear their young over the summer (Clark et al., 1993, 1996, and 2002). Distances between hibernacula and summer caves are known to range from 6.5 to 65 km (4 to 40 miles). The exact timing of the formation of maternity colonies varies between years, but usually occurs between late April and early June (Clark et al., 2002; U. S. Fish and Wildlife Service, 1995). Like other temperate bats, the species exhibits strong roost fidelity, returning to the same maternity sites and hibernacula year after year (Kunz and Martin, 1982; Clark et al., 1996; Weyandt et al., 2005). Ozark big-eared bats typically emerge from their caves to forage shortly after sunset (Clark et al., 1993 and 2002). They primarily feed on moths, but also are known to eat beetles and other flying insects (USFWS, 1995; Leslie and Clark, 2002; Dodd, 2006). Forested habitats are an important source of food for the Ozark big-eared bat. A recent study on the diet of the Ozark big-eared bat and prey abundance in Arkansas found that the bats prey on a wide diversity of moth species, and that most of the species are dependent upon woody forest plants as a host (Dodd, 2006). The study also found a positive correlation between woody species richness and moth occurrence. Conservation of the Ozark big-eared bat, therefore, requires not only protection of important caves but also forested habitat that supports abundant and diverse moth populations (Leslie and Clark, 2002; Dodd, 2006; Dodd and Lacki, 2007). Conservation practices that encourage a diversity of woody forest plant species (e.g., prescribed fire, selective thinning) to provide a rich prey base of moths should benefit Ozark big-eared bat colonies. Females forage relatively close to the maternity cave (about 1.0 – 2.0 km; 0.6 – 1.2 miles) during the early and middle portions of the maternity season. Female bats likely forage only short distances from the cave in order to return several times during the night to take care of flightless young. As the season progresses, average distance to foraging sites (up to about 7.3 km; 4.5 miles) increases (Clark et al., 1993; Harvey, 1992). Foraging farther distances from the cave later in the summer may reduce competition with newly volant young that have begun to forage. The Ozark big-eared bat has been shown to selectively forage in both edge and forested habitats and also to use habitats in proportion to their availability. A radio telemetry study of the foraging activity of females during the maternity season, for example, found that females used edge habitats more than expected (Clark et al., 1993). Another study, however, found that males selected forested areas during late summer (i.e., September) while females failed to show preference for foraging habitat (Wethington et al., 1996). Based on wing-loading characteristics (i.e., the ratio of weight to wing

area), the Ozark big-eared bat is considered a highly maneuverable flier. Ozark big-eared bats are well adapted to forage in either a cluttered environment such as the interior of a forest or a relatively more open area, such as edge habitats (Farney and Fleharty, 1969; Leslie and Clark, 2002; Clark et al., 2003; Wethington et al., 1996). The Ozark big-eared bat, therefore, is not as restricted in its selection of foraging habitats as other less maneuverable species. Selection of foraging habitat by this subspecies may change seasonally and likely is due to both foraging efficiency and the availability of prey (Clark et al., 1993; Dodd, 2006; Wethington et al., 1996). Edge habitat may be selected at times of high moth abundance because it is relatively less costly to forage there as compared to the more cluttered forest interior and woodland moths are abundant enough that the probability of encounter is high. However, during times of reduced moth abundance, Ozark big-eared bats may move into the forest interior to forage where the occurrence of their preferred prey is relatively higher (Dodd, 2006).

***Dispersal/Migration*****Motility/Mobility**

Adult: High

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory

**Dispersal/Migration Narrative**

Adult: Relatively sedentary; movements generally are not more than about 64 km. (NatureServe, 2015). Distances between hibernacula and summer caves are known to range from 6.5 to 65 km (4 to 40 miles).

***Population Information and Trends*****Population Trends:**

Increasing

**Population Size:**

~1,800

**Population Narrative:**

Ozark big-eared bat populations at essential hibernacula and maternity sites have been monitored using minimal census techniques since each essential site was discovered to obtain estimates on colony size and population trends (Puckette, 2008; Harvey et al., 2006). Monitoring data reveal a disparity between summer and winter population estimates. Numbers of Ozark big-eared bats estimated from summer maternity counts are larger than those found during winter hibernacula counts. For example, for the last year in which a representative count of both Ozark big-eared bat hibernacula and maternity sites occurred (2003), 701 bats were counted at hibernacula while maternity counts resulted in an estimate of about 1,600 bats. This indicates there likely are major hibernacula that have not yet been located. Population estimates and trends are therefore based on maternity colony counts. The Service recently completed a 5-year review for the Ozark big-eared bat (USFWS, 2008). Five-year reviews are assessments of the best scientific and commercial data currently available for a listed species, and are used to determine whether or not a change in the federal classification of a species is

warranted. The 5-year document examined abundance and population trends for data collected through the 2006 maternity season. The document contains the most recent summary of information pertaining to population size, variability, and stability. Therefore, information from that analysis is summarized here. Data collected from the 2008 maternity colony surveys also are utilized here for estimates of current population size (Although the 2009 data are available, the 2008 estimates are used to estimate population size and trends because counts were not conducted at all maternity caves during the 2009 maternity season.). At the time of listing, the Ozark big-eared bat was known from only a few caves in northwestern Arkansas, southwestern Missouri, and northeastern Oklahoma. The entire population was estimated to consist of about 100-200 individuals (Figure 2). Since listing, additional caves used by maternity colonies in the summer and as hibernacula have been discovered in Oklahoma and Arkansas. The population currently is estimated to consist of about 1,800 individual bats with about 400 in Arkansas and 1,400 in Oklahoma. Census counts indicate that the overall population has experienced a slightly increasing trend since 1997, when the last discovered essential maternity site from which we have several years of population data (a maternity cave in Arkansas) were added to the annual counts. The overall population estimate has averaged about 1,500 bats between 1997 and 2008. An increasing population trend is observed over this time period when the data from Arkansas is considered alone. In contrast, estimates from exit count data for Oklahoma indicate that the population size in Oklahoma has experienced an overall slightly declining trend since 1987, the first year in which annual monitoring efforts included all known essential maternity sites from the state. The apparent declining trend in Oklahoma may be attributable to movement among caves, including sites not known to us, and not an actual decrease in bat numbers, and due to the difficulty in monitoring bats at certain caves.

### ***Threats and Stressors***

**Stressor:** Vandalism/Human activity

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Vandalism and unauthorized human activity at maternity roosts and hibernacula still occur even at gated and signed caves. Therefore, human disturbance remain a serious threat. The disparity between summer and winter counts indicates there likely are more caves of importance to the Ozark big-eared bat of which the bat conservation community is not yet aware. A prerequisite to protecting these sites is knowledge of their location, so the need to continue search efforts for unknown Ozark big-eared bat caves continues. Current and future human population growth and development within the Ozark big-eared bat's range will result in the loss and fragmentation of foraging habitat. In addition to protecting the caves used by the Ozark big-eared bat, it will become increasingly important to protect and restore foraging habitat around these caves as development pressures increase in the future (Leslie and Clark, 2002; Wethington et al., 1996). A recent genetics study provides further insight into the need to protect each maternity colony. Weyandt et al. (2005) examined population genetic variability and found that maternally inherited markers differed among sites, indicating very strong site fidelity and limited dispersal by females and high natal philopatry. Due to the natural tendency for limited dispersal by female Ozark big-eared bats and the apparent corresponding lack of connectivity among colonies, caves that experience a local extinction are unlikely to be naturally re-colonized. These results suggest that failure to protect a maternity site may result in the loss of genetic variation.

**Stressor:** Climate change

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Climate change could have a significant impact on all temperate region bats, including the Ozark big-eared bat. Projected changes in climate could impact bats by adversely affecting their food supply and the internal roosting temperature of caves (Bogan, 2003). The Ozark big-eared bat preys on a wide diversity of moth species, but most of the moth species are dependent upon woody forest plants as a host. Climate change may affect the Ozark big-eared bat by impacting plant resources which could alter the timing and abundance of moth prey. Ozark big-eared bats have specific cave microclimate requirements. Only those caves with appropriate microclimates are used as maternity roosts and hibernacula. Changes in the internal roosting temperature of caves may change the suitability of certain caves. Changes in food resources and cave microclimates may affect hibernation periods, and the birth and survival of pups.

**Stressor:** White-nose syndrom

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** White-nose syndrome (WNS) is a new bat malady first observed in four caves in New York during the winter of 2006-2007 that potentially could affect the Ozark big-eared bat in the near future. The fungus *Geomyces destructans* is believed to be the causative agent of WNS, which frequently results in the deaths of infected hibernating bats. The fungus thrives in the cold and humid conditions characteristic of caves, and affected bats have the fungus growing around their nose or other bare surfaces including the wings. WNS currently is known from 11 States in the northeastern and eastern United States and two Canadian Provinces. Experts estimate that over 1,000,000 bats have died due to WNS during the past 4 years. The primary mode of disease transmission is believed to be bat-to-bat contact. Research is ongoing to determine whether all bats that come into contact with the fungus will develop the disease. Although mortality attributable to WNS has not occurred within Oklahoma, the fungus associated with WNS recently was documented on a single cave myotis *Myotis velifer* collected alive from a cave on May 3, 2010, in northwestern Oklahoma. The fungus also was found on gray bats in Missouri during the spring of 2010, a species that co-occurs in caves with the Ozark big-eared bat. Should WNS move into the range of the Ozark big-eared bat (and should Ozark big-eared bats prove to be susceptible to the disease), the potential impact would be severe due to the high mortality rate of affected bats in the northeastern and eastern United States, and the small population size and limited distribution of the Ozark big-eared bat.

## ***Recovery***

### ***Conservation Measures and Best Management Practices:***

- Section 7(a)(1) of the Act directs federal agencies to use their authorities to further the purposes of the Act by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse impacts of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information. A primary goal of HFRP is to promote recovery of listed

species. No additional conservation recommendations are necessary due to the inherent benefits that will occur during implementation of HFRP and the associated HRP.

**References**

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## **SPECIES ACCOUNT: *Corynorhinus (=Plecotus) townsendii virginianus* (Virginia big-eared bat)**

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### ***Species Taxonomic and Listing Information***

**Commonly-used Acronym:** VBEB

**Listing Status:** Endangered; Northeast Region (R5) (USFWS, 2015)

### **Physical Description**

*Plecotus townsendii* is a medium-sized bat with forearms measuring 39 to 48 millimeters (mm) long and weighing 7 to 12 grams. Total body length is 98 mm, the tail is 46 mm, and the hind foot is 11 mm long. This bat's long ears (over 2.5 centimeters) and facial glands on either side of the snout are quite distinctive. Fur is light to dark brown depending upon the age of the individual and the subspecies (USFWS, 2015).

### **Taxonomy**

Formerly included in the genus *Plecotus* (see taxonomic comments for *C. townsendii*). (NatureServe, 2015)

### **Historical Range**

Historically known from Appalachian Mountains in Virginia, West Virginia, North Carolina, and eastern Kentucky. (NatureServe, 2015)

### **Current Range**

Presently occurs in decreased numbers throughout much of the historic range. Largest colonies are in several caves in Pendleton County, West Virginia; some caves serve as both hibernation and maternity sites, others are primarily maternity caves. Colonies occur also in Lee County and surrounding counties, Kentucky (the best known site being Stillhouse Cave); in Bath, Highland, Rockingham, Bland, and Tazewell counties, Virginia (Dalton 1987); and in Avery and Watauga counties, North Carolina (including Black Rock Cliffs Cave) See Matthews and Moseley (1990) and Handley (1991). (NatureServe, 2015). Since the 2008 status review, there are new county records for: Watauga, North Carolinawhere a new maternity cave has been documented (Weber et al. 2016); Carter and Johnson Counties, Tennessee where VBEBs were tracked to day roost sites from known caves in adjacent counties (Weber et al. 2016); Bath County, Virginia where 4 VBEBs were found hibernating in 2 caves (VDGIF 2017 data);and Pulaski County, Kentucky where one male VBEB was found hibernating. This site is located 32.8 miles from the closest known VBEB site (Kiser 2016) (USFWS 2019).

### **Distinct Population Segments Defined**

No

### **Critical Habitat Designated**

Yes; 11/30/1979.

### **Legal Description**

On November 30, 1979, the Service determined the Virginia big-eared bat to be an endangered species and determined five caves in West Virginia to be critical habitat.

**Critical Habitat Designation**

The critical habitat designation for *Corynorhinus (=Plecotus) townsendii virginianus* includes five caves in Pendleton and Tucker Counties, West Virginia.

Pendleton County: Cave mountain Cave, Hellhole Cave, Hoffman School Cave, Sinnit Cave.

Tucker county: Cave Hollow Cave.

**Primary Constituent Elements/Physical or Biological Features**

Not specified. The Virginia big-eared bat depends on the maintenance of precise conditions in these caves which it must use for hibernating sites in the winter and for nurseries in the summer.

**Special Management Considerations or Protections**

Activities that may adversely modify critical habitat include: 1. Any action which would substantially alter the physical structure, temperature, humidity, or air flow of the designated caves could adversely modify critical habitat since the Virginia big-eared bat depends on the maintenance of precise conditions in these caves which it must use for hibernating sites in the winter and for nurseries in the summer. 2. Any action which would result in disturbance of the bats in their hibernating or nursery caves would adversely affect critical habitat since the species is highly tolerant of human disturbance. Such activity might include blasting or construction in or near the designated caves, or increasing human access to the caves.

***Life History*****Feeding Narrative**

Adult: Feeds principally on moths. Forages over fields and woods, with individuals routinely traveling 3-5 miles from roost cave to foraging area (End. Sp. Tech. Bull., Sept./Dec. 1991). In eastern West Virginia, Lepidoptera was the most important insect order in the diet, followed by Coleoptera, Diptera, and Hymenoptera; compared to availability, selectively consumed Lepidoptera and avoided Coleoptera; forest insect comprised a substantial part of the diet (Sample and Whitmore 1993).; Food Habits: Invertivore (Adult, Immature). Activity usually begins well into the night, late relative to other bats. After an initial feeding period, roosts and rests during the night, presumably before a later feeding bout. Commonly arouses in winter, changing position within a hibernaculum or moving to a nearby cave or mine.; (NatureServe, 2015). Townsend's big-eared bat feeds principally on small moths (Microlepidoptera), averaging 6mm in length (range 3 to 10 mm), and also may take other insects, including representatives of Neuroptera, Coleoptera, Diptera, and Hymenoptera (Hamilton, 1943; Ross, 1967; Whitaker et al, 1977). Howell (1920) noted that Townsend's big-eared bat captured insects from leaves and other places. However, Bell (in Kunz and Martin, 1982) noted that big-eared bats feed mostly in the air along forested edges and should not be regarded as foliage gleaners (USFWS, 1984).

**Reproduction Narrative**

Adult: Maximum longevity reported for this species is 16 years 5 months, based on recoveries of banded bats in California (Paradiso and Greenha11, 1967). The following is a summary by Kunz and Martin (1982) of Pearson's work: Estrus and subsequent copulation begin in autumn and

the peak of copulations occurs from November through February, although some females apparently mate before arriving at hibernacula. Young females are reproductively active and mate in their first autumn. Spermatozoa are stored in the reproductive tracts of females until spring, when ovulation, fertilization, and gestation occur. Ovulation may occur either before or after females leave hibernation. Development of a single embryo takes place in the right uterine horn. The length of gestation varies from 56 to 100 days, depending on spring temperatures and the varying amounts of torpor experienced by different individuals. Parturition occurs in late spring and early summer, followed by an anestrous period. In adult males, spermatogenesis occurs during the summer, reaching maximum activity in September. By late September and early October, the testes of adults begin to atrophy, coinciding with the appearance of sperm in the enlarging epididymides. The accessory glands reach full size in late October. Copulation is preceded by a ritualized precopulatory behavior characterized by the production of audible vocalizations, followed by head nuzzling which may be directed at either torpid or active individuals. Young males fail to reach sexual maturity in their first autumn. As in other bats, baby Townsend's big-eared bats are large at birth, weighing nearly 25% of their mother's post-partum mass. Newborn bats are naked and their large ears lie over their unopened eyes for the first few days. Within a few hours after birth they can produce audible 'chirps' which may play an important role in mother-infant recognition. At the age of one week, young bats are capable of producing adult-like audible 'squawks'. Young bats grow rapidly, nearly reaching adult forearm size in one month. They are capable of flight at 2.5 to 3 weeks and are fully weaned by 6 weeks (USFWS, 1984).

**Spatial Arrangements of the Population**

Adult: Clumped (NatureServe, 2015)

**Environmental Specificity**

Adult: Narrow/specialist (NatureServe, 2015)

**Tolerance Ranges/Thresholds**

Adult: Low (NatureServe, 2015)

**Site Fidelity**

Adult: High (NatureServe, 2015)

**Habitat Narrative**

Adult: Species inhabits caves typically in limestone karst regions dominated by mature hardwood forests of hickory, beech, maple, and hemlock (Matthews and Moseley 1990). Prefers cool, well-ventilated caves for hibernation (Matthews and Moseley 1990); roost sites are often near cave entrances or in places where there is considerable air movement (Handley 1991). Males and females hibernate together. In summer, males occur singly or in groups in caves (Handley 1991). In eastern Kentucky, feeding roosts were in cliffs adjacent to two maternity roosts and one bachelor roost (Burford and Lacki 1998). Individuals may move from one roost to another at any season. Maternity colonies settle deep within caves, far from entrance (Matthews and Moseley 1990); these caves are warmer than those used for hibernation. In Kentucky, used limestone caves, except in one instance in which a sandstone rock shelter was used (Lacki et al. 1994). In Kentucky, often detected in old fields and above cliffs (Burford and Lacki 1995) (NatureServe, 2015). High ecological integrity of the population and site fidelity as

well as low tolerance ranges are inferred based on the species need to inhabit caves that are not disturbed.

***Dispersal/Migration*****Motility/Mobility**

Adult: High (NatureServe, 2015)

**Dispersal/Migration Narrative**

Adult: Fairly sedentary; not known to migrate more than about 64 km between hibernation and maternity caves (Matthews and Moseley 1990) (NatureServe, 2015).

***Population Information and Trends*****Resiliency:**

Moderate (inferred from NatureServe, 2015)

**Representation:**

Moderate (inferred from NatureServe, 2015)

**Redundancy:**

Moderate (inferred from NatureServe, 2015)

**Number of Populations:**

6 - 20 (NatureServe, 2015)

**Population Size:**

The current total population of the species is approximately 19,574 bats in hibernacula and 11,778 within the known maternity sites (USFWS, 2019). The large majority of these bats are currently concentrated in 10 hibernacula and 18 maternity sites distributed among 4 genetically distinct populations located in geographically distinct regions (USFWS, 2019).

**Population Narrative:**

Total known population was 3,866 bats (11 colonies) in 1984 (Bagley and Jacobs 1985), about 10,000 in the late 1980s (Dalton 1987). In 1987, the total West Virginia population was 8000, based on a count of about 3500 females, up almost one-third since 1983 (Matthews and Moseley 1990). A 1991 count of the 9 summer colonies in West Virginia yielded 4455 individuals, a 15 percent increase from the 1990 count and a 20 percent increase from 1984 (End. Sp. Tech. Bull., Sept./Dec. 1991); the count was basically unchanged in early 1992, but in May 1992 the largest known maternity colony (1300 individuals) of this subspecies was discovered (1992 End. Sp. Tech. Bull. 17(12):18). The largest known concentration of this species is in Hellhole Cave, West Virginia; the count for the 1994-1995 season was 6378 individuals (End. Sp. Bull. 20(4):21). As of the mid-1990s, West Virginia/North Carolina population was more than 13,000 (1994 End. Sp. Tech. Bull. 19(5):14). In Kentucky, the hibernating population in Stillhouse Cave increased from 1700 in 1982 to 2600 in 1987 (Matthews and Moseley 1990). Virginia population in the 1980s was about 2000 and stable (Dalton 1987, Handley 1991). Total population in 1997 probably was less than 20,000 (Pupke 1997). Known from about 15 caves in 4 states (Kentucky-1, Virginia-2, West Virginia-11, North Carolina-1). Other colonies have either

declined or disappeared (NatureServe, 2015). Moderate resiliency, representation and redundancy are inferred based on the number of known population and their relatively dispersed geography.

### ***Threats and Stressors***

**Stressor:** Development (USFWS, 2008)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Foraging habitat in some areas have been lost to development (USFWS, 2008).

**Stressor:** Mining activities (USFWS, 2008)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Mining activities could potentially impact some of the caves that support this species (USFWS, 2008).

**Stressor:** Protection of cave habitat (USFWS, 2008)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Most caves are protected (gated). However, some privately owned caves are not (USFWS, 2008).

**Stressor:** Rock climbing (USFWS, 2008)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Rock climbing activity has increased in an area of Kentucky which has potential habitat. Inadvertent disturbance could occur (USFWS, 2008).

**Stressor:** White nosed syndrome (USFWS, 2008)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals/loss of populations

**Narrative:** Although not known to occur in VBEB habitats this species is susceptible to possible infestation (USFWS, 2008).

**Stressor:** Predation (USFWS, 2008)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** Predation by house cats and black rat snakes has been documented (USFWS, 2008).

**Stressor:** Wind turbines (USFWS, 2008)

**Exposure:**

**Response:****Consequence:** Loss of individuals**Narrative:** The anticipated development of wind turbines near hibernacula maternity colony caves is a threat to this species (USFWS, 2008).**Stressor:** Vandalism (USFWS, 2008)**Exposure:****Response:****Consequence:** Loss of habitat**Narrative:** Even when gates are locked, cave vandalism can be a problem (USFWS, 2008).**Stressor:** Road mortalities (USFWS, 2008)**Exposure:****Response:****Consequence:** Loss of individuals**Narrative:** VBEB has become more susceptible to road mortalities as areas around their habitat become more developed (USFWS, 2008).**Stressor:** Oil and brine separation plants (USFWS, 2008)**Exposure:****Response:****Consequence:** Loss of individuals**Narrative:** Oil and brine separation plants are a threat to this species (USFWS, 2008).**Recovery****Reclassification Criteria:**

A minimum number of maternity, hibernation, and bachelor sites and total abundance for each MU are attained as described in table 1 of the Recovery Plan Draft Amendment 1 (USFWS, 2019)

For each MU, total population numbers for both hibernacula and maternity sites are stable or increasing for a timeframe approximately equal to the lifespan of a VBEB, (approximately 16 years), which encompasses multiple VBEB generations, and meet or exceed the minimum population numbers listed in table 1 for the most recent half of that timeframe. Numbers shall be based on biennial monitoring of hibernation sites and annual monitoring of maternity sites using Service-approved protocols (USFWS, 2019).

For all sites needed to support the minimum population numbers and distribution specified in table 1, long-term management agreements are in place (finalized and fully implemented) with responsible land and resource management entities. Long-term protection is defined to include: a) The site is located on state or Federal lands with an established long-term management plan, or it is located on private lands with a signed enforceable management agreement that will transfer to new owners; and b) The management plan or agreement specifies that the area will be maintained for the benefit of the VBEB and ensures that habitat (including both the surface and subsurface features) sufficient to support all life functions at all life stages of the populations that utilize the area will be conserved; and c) Human access to the site is controlled by the installation of gates or fences, unless the site is located in a sufficiently remote location such that access violations are not expected. In addition, the site must be closed to access

during all periods VBEs are expected to be present (except for access needed to manage or monitor the bats or the site). Signs are placed at the site to indicate access is prohibited (USFWS, 2019).

Long-term management agreements are in place to protect features essential to all identified key foraging areas (USFWS, 2019).

**Delisting Criteria:**

Delisting for the VBE may be considered when criteria 1 through 4 above are maintained and when all of the following additional criteria are met: Within each MU, all sites needed to support the minimum population numbers and distribution as specified in table 1 are connected by habitats that support travel between sites (USFWS, 2019).

Long-term mechanisms are in place to deter, monitor, detect, and enforce access violations; maintain any gates, fences, and other access controls; and ameliorate adverse effects (including predation) for all sites required to meet criterion 1. Effective monitoring programs are in place to detect access violations and damage to any gates or other access controls in a timely manner. Responsible management entities are identified and accountable for maintaining and repairing access controls, and for regulating and controlling threats from predation (USFWS, 2019).

**Recovery Actions:**

- Review of the existing list of recovery actions as well as the 2008 and 2019 status reviews indicates that some recovery actions have been completed (USFWS, 2019).

***Conservation Measures and Best Management Practices:***

- **Continue Monitoring Population Trends at All Essential Sites:** One recovery criterion for downlisting in the 1995 recovery plan requires maintenance of a stable or increasing OBEB population at all known essential caves over a 10-year period. Results from a recent genetic study (Weyandt et al. 2005) corroborate the importance of monitoring the population trends at each colony. The research suggests very strong site fidelity and limited dispersal by females, and high natal philopatry. These results suggest that failure to protect a maternity site may result in the loss of genetic variation. Each essential site should continue to be monitored over the next 10 years to determine population trends. The hibernacula that are difficult to monitor without disturbing the bats should be monitored every three years (USFWS, 2008).
- **Acquire Essential Caves and Important Foraging Habitat for Additions to the Ozark Plateau NWR:** The Ozark Plateau NWR was approved, in 2005, to expand up to 15,000 acres in Adair, Delaware, Ottawa, Sequoyah, Craig, Mayes, and Cherokee counties, Oklahoma. The Environmental Assessment for the approved Expansion of the Ozark Plateau NWR (Service 2002) includes a land protection plan that identifies: 1) important known habitat for the OBEB in need of long-term protection, 2) the preferred type of protection for each tract, 3) the minimum type of protection deemed necessary, and 4) a protection priority classification for each site. Protecting additional OBEB caves and foraging areas through fee title acquisition and conservation easements would help minimize future destruction and modification of cave and foraging habitats. Adding cave sites to the refuge also would facilitate monitoring of the sites and help regulate human entry for scientific, recreational, and educational purposes. Additional OBEB essential and limited-use caves and surrounding foraging areas need protection through acquisition and/or other measures such as cave gating. Important sites that currently are not afforded protection include essential caves AD-17, AD-24, AD-25, AD-T1, and WA-5202, as well as numerous limited-use caves. These sites could be acquired by

the Service as additions to the refuge or by other natural resource agencies and conservation groups through fee title acquisition or through conservation easements when sellers or donors are willing. The development of voluntary cooperative agreements and cave management plans to protect forested foraging habitat and caves also are potential conservation measures that can be pursued to prevent habitat loss and modification (USFWS, 2008).

- **Increase Staff and Funding Levels at the Ozark Plateau NWR:** Refuge responsibilities are extensive and include developing and maintaining positive landowner relations, developing and implementing cooperative agreements with landowners, working with state and federal agencies, universities, and non-profit organizations, constructing cave gates and fences, repair and maintenance of cave gates and fences, habitat enhancement and restoration (e.g., timber thinning, planting, prescribed burns, etc.), maintenance of roads and buildings, annual monitoring of bat populations, cavefish and cave crayfish monitoring, identifying important tracts for future acquisition, placement and maintenance of interpretative and warning signs at cave entrances, law enforcement, mapping essential caves, facilitating important research, developing and implementing plans for scientific, educational, and other public use, actively preparing proposals for funding from the Service and other agency and private sources for management and acquisition, and preparing important planning documents. Inadequate funding and insufficient staffing at the Ozark Plateau NWR would only continue to make refuge management, and, hence, meeting an OBEB recovery criterion difficult. The Southwest Region's "National Wildlife Refuge System Work Plan" for FY 2007 – 2009 identifies the Ozark Plateau NWR as a Tier 1 focus refuge for the Region. This classification implies that staff and funding from refuges classified as Tier 2 (Targeted Reduction Refuges) and Tier 3 (Satellite Refuges) would be shifted to the Ozark Plateau NWR. Increasing staffing and funding levels would help ensure sufficient operation of the refuge and facilitate recovery of the OBEB. Filling the following positions would facilitate more efficient operation of the Refuge: 1) Refuge Manager, 2) Fish and Wildlife Biologist, and 3) Administrative Assistant (USFWS, 2008).
- **Develop Voluntary Cooperative Agreements with Private Landowners:** The OBEB is known to forage up to 5 miles from cave sites. Efforts to protect foraging habitat should focus on areas within a 5-mile radius from known caves (Harvey 1992, Clark et al. 1993, Wethington et al. 1996). Most surface foraging habitat occurs on private land. Although acquisition in fee title is the most secure and long-term means of protecting OBEB caves and foraging habitat, purchase of all areas necessary for the recovery of the OBEB likely would not be possible due to the large area used by OBEBs. Therefore, working with private landowners has and will continue to be an important recovery tool. The Service's Partners for Fish and Wildlife Program is designed to work cooperatively with private landowners to protect and enhance fish and wildlife resources. The Partner's Program has provided financial assistance for the construction of cave gates in Oklahoma. Where possible, the Partner's Program should continue to be used to protect cave sites from human disturbance through financial and technical assistance. In addition, a number of important caves on private land have been gated with funds from Section 6 of the Endangered Species Act in cooperation with Oklahoma Department of Wildlife Conservation and Rogers State University. This program is popular with private landowners and has been very successful and should continue. Establishing relationships with private landowners also could facilitate the development of voluntary cooperative agreements to protect forested foraging habitat. Potential avenues for these voluntary agreements include the development of Safe Harbor Agreements and TNC's Natural Area Registry Program (USFWS, 2008).
- **Facilitate Management by Other Agencies and Groups:** The Service has worked closely with several state and federal agencies, tribes, universities, and non-profit organizations to protect and manage OBEB habitats, including the ODWC, ANHC, AGFC, the Cherokee Nation, Ozark National Forest, the Oklahoma and Arkansas Chapters of TNC, City of Tulsa, Land Legacy, and the local chapter of the NSS (Tulsa Regional Oklahoma Grotto). Universities involved include Rogers State University,

Oklahoma State University, University of Oklahoma, Northeastern State University, Southeastern Oklahoma State University, University of Central Oklahoma, University of Arkansas, and Arkansas State University. The Service should continue to coordinate management efforts with other agencies and organizations. Essential foraging habitat that is available from willing sellers should be identified for future purchase by the States of Oklahoma and Arkansas through the Recovery Land Acquisition Program and other mechanisms. Landowners of important tracts that are not for sale should be approached regarding conservation easements and possible voluntary cooperative agreements, such as The Nature Conservancy's Natural Area Registry Program (USFWS, 2008).

- **Fine-Tune and Standardize Annual Monitoring at Maternity Colonies:** The population trend analysis at all known essential caves revealed a statistically significant trend at only four of the 15 sites analyzed. The inability to determine whether the population was increasing, decreasing, or stable at most of the essential sites is likely attributable to several possible factors, including movements of bats among the caves and other life history traits that make monitoring more difficult. Additionally, surveyors conducting exit counts in mid-June could unknowingly count only adult females in some years and females plus newly volant young in others. As the climate warms the bats may be reproducing earlier in the year and the young flying earlier. Fine-tuning and standardizing the monitoring approach likely will facilitate collection of more comparable data and enhance efforts to determine population trends at known sites (USFWS, 2008).
- **Investigate the Feasibility of Gating AD-24 and/or -25 to Minimize Human Disturbance:** Apparent declines of the AD-13/24/25 colony may be attributable to movement among caves, as discussed above. Human disturbance could be a contributing factor to the potential movement. Although AD-13 is gated to prevent unnecessary human disturbance and vandalism, neither AD-24 nor -25 are afforded such protection. The landowner of these sites should be contacted regarding implementation of this conservation measure (USFWS, 2008).
- **Assess the Ownership and Protective Status of All Known Limited-Use Sites:** Limited-use sites should be afforded protection. These sites provide important habitat for small groups of bats and solitary males during the summer. An assessment of the ownership and protective status (e.g., gated, cooperative landowner agreement, etc.) for each site should be determined. Conservation easements, fee title acquisitions, and cooperative landowner agreements should be sought on all unprotected sites (USFWS, 2008).
- **Re-Visit Historic and Possible OBEB Caves in Missouri:** An OBEB survey was conducted at 34 sites in Missouri during the summer and fall of 1999 (Elliott et al. 1999). During this survey, evidence of OBEB use, in the form of neatly clipped moth wings, was discovered at two cave sites. A list of the sites from the survey effort is available from the MDC. At a minimum, the two sites with evidence of use should be re-visited periodically. The Oklahoma Ecological Services Field Office currently is working with the Missouri Ecological Services Field Office to investigate possible funding sources and the availability of qualified biologists to conduct an OBEB survey in Missouri within the next few years (USFWS, 2008).
- **Continue to Search for Caves of Importance:** The possibility of finding new essential and limited-use OBEB sites in the Ozarks still exists. For example, in the summer of 2006, a cluster of 15 OBEBs was discovered in a sandstone talus crack on a private in-holding within the Ozark National Forest. Additionally, annual monitoring efforts at maternity sites and hibernacula present a disparity between summer and winter population estimates. Numbers of OBEBs estimated from summer maternity counts are larger than those found during winter hibernacula counts. This indicates there are likely major hibernacula being used by OBEB that have not yet been located. Therefore, searches for unknown maternity sites, limited-use sites, and hibernacula should continue throughout the Ozarks in Oklahoma and Arkansas. Additionally, evidence of possible OBEB occurrence in the form of neatly clipped moth wings and guano has been found in many caves in Oklahoma, Arkansas, and

Missouri. These sites should be revisited to determine whether they are caves of importance. Equipment, such as the Anabat detector that can be placed near cave entrances to record and help identify echolocating bats, may prove valuable in this effort. Should re-visitation of historic or possible sites in Missouri find OBEs, search efforts should be intensified in Missouri (USFWS, 2008).

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## SPECIES ACCOUNT: *Cynomys parvidens* (Utah prairie dog)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Threatened; 06/04/1973; Mountain-Prairie Region (R6) (USFWS, 2016)

#### **Physical Description**

A large rodent (prairie dog). Summer pelage is reddish (tawny olive to clay) dorsally, mixed with black-tipped hairs; short, white-tipped tail the terminal half of which has a white center; total length 31-36 cm (Whitaker 1996) (NatureServe, 2015). The Utah prairie dog (*Cynomys parvidens*) is the smallest species of prairie dog. Individuals are typically 12 to 14 inches (in) long (Hollister 1916) and weigh 1.4 to 3.1 pounds (Wright-Smith 1978) (USFWS, 2015).

#### **Taxonomy**

There are five species of prairie dogs native to North America (Hoogland 2003). Taxonomically, prairie dogs (*Cynomys* spp.) are divided into two subgenera: the white-tail and black-tail. The Utah prairie dog (*C. parvidens*) is a member of the white-tail group, subgenus *Leucocrossuromys*. Other members of this group, which also occur in Utah, are the white-tailed prairie dog (*C. leucurus*) and the Gunnison prairie dog (*C. gunnisoni*). The Utah prairie dog is recognized as a distinct species (Zeveloff 1988; Hoogland 1995), but is most closely related to the white-tailed prairie dog. These two species may have once belonged to a single interbreeding species (Pizzimenti 1975). They are now separated by ecological and physiographic barriers and exhibit genetic differences (USFWS, 2015).

#### **Historical Range**

The type locality for the Utah prairie dog is Buckskin Valley in Iron County, Utah (Pizzimenti and Collier 1975). Historically, Utah prairie dog colonies were found as far west as Pine and Buckskin Valleys in Beaver and Iron Counties, and may have occurred as far north as Nephi, southeast to Bryce Canyon National Park, east to the foothills of the Aquarius Plateau, and south to the northern borders of Kane and Washington Counties (Pizzimenti and Collier 1975) (USFWS, 2015).

#### **Current Range**

The range is restricted to an area of about 1,850 square kilometers in southern Utah. Prior to control programs, the range reportedly extended from Pine and Buckskin valleys in Beaver and Iron counties (perhaps west to Modena in Iron County), north to at least Salina Canyon and near Gunnison in Sevier County (possibly to Nephi), south to Bryce Canyon National Park, and east to the foothills of the Aquarius Plateau (Collier 1974, Pizzimenti and Collier 1975, McDonald 1997). More recently, this species occurred in substantial populations in only three areas: the Awapa Plateau, along the east fork of the Sevier River, and in eastern Iron County; the grass and Sevier river valleys, plus three small, widely separated mountain valleys have small populations (Collier 1974, Pizzimenti and Collier 1975). The species is scarce or absent in the Aquarius Plateau, Fremont and Paria valleys, and Salina Canyon (Collier 1974, Pizzimenti and Collier 1975) (NatureServe, 2015). Today, Utah prairie dogs are limited to the central and southwestern quarter of Utah in Beaver, Garfield, Iron, Kane, Piute, Sevier, and Wayne Counties (USFWS, 2012).

#### **Distinct Population Segments Defined**

No

**Critical Habitat Designated**

No;

***Life History*****Feeding Narrative**

Adult: Feeds primarily on grasses, alfalfa, leafy aster, European glorybind, wild buckwheats in seed, flowers and seeds of shrubs, and insects when available; also may consume cattle feces; generally prefers flowers and seeds over leaves. Inactive and torpid in severe winter conditions. Adults emerge and begin foraging from mid-March to early April, enter dormancy mid-July to mid-August; juveniles enter dormancy from early October to mid-November; low elevation colonies (below 7000 ft) generally are two weeks earlier than higher elevation colonies (Spahr et al. 1991) (NatureServe, 2015). Grasses are a staple of the annual diet (Crocker-Bedford and Spillett 1981; Hasenyager 1984) (USFWS, 2012).

**Reproduction Narrative**

Adult: Reproduce slowly, relative to other rodents (Hoogland 2001). Females produce only one litter per year, but the probability of weaning a litter each year is only 67% (Hoogland 2001). Although all females copulate as yearlings, only 49% of males do so (Hoogland 2001). For females that wean offspring, mean litter size at first emergence from the nursery burrow is 3.88 (Hoogland 2001). Mating occurs in March or April. Gestation lasts about 1 month. Young are born in late April or early May. Litter size is 2-10 (average 3-5). Young emerge above ground at 6 weeks (late May to early June), weaned in about 7 weeks, first breed at about 2 years. Survivorship in first year less than 50%; only 30% remain alive at the end of their second year (Hoogland 2001) (NatureServe, 2015). One half to two thirds of Utah prairie dog's adult population is female (Mackley et al. 1988). Due to their limited reproductive rates, short life span and high mortality rates, numbers of individuals counted within a colony can fluctuate greatly throughout the year with low points in the spring and peaks in the late summer when adults and pups are above ground. The adult females play the major role in caring for young, they are also the primary ones that provide warning of danger (Wright Smith 1978) (USFWS, 2015).

**Geographic or Habitat Restraints or Barriers**

Adult: Brushy vegetation, occurs at 5,400 - 9,500 ft. elevation (USFWS, 2015)

**Spatial Arrangements of the Population**

Adult: 2.5/ha to more than 74/ha; colonial (NatureServe, 2015)

**Habitat Narrative**

Adult: Habitat consists of grasslands, in level mountain valleys, in areas with deep well-drained soil and vegetation that prairie dogs can see over or through. Prairie dogs dig underground burrow systems, in which the young are born. Population densities are extremely variable, ranging from a mean of less than 2.5/ha to more than 74/ha (Pizzimenti and Collier 1975). Lives in colonies ("towns"). Colony structure is dynamic in size and location; social units within colony comprise a dominant male, several females, and the young of the past 2 years (Matthews and Moseley 1990) (NatureServe, 2015). They often select colony sites in swales where the

vegetation can remain moist even in drought conditions (Collier 1975; Crocker Bedford and Spillet 1981). Utah prairie dogs are found in elevations from 5,400 ft on valley floors up to 9,500 ft in mountain habitats. Vegetation must be of short stature to allow the prairie dogs to see approaching predators as well as have visual contact with other prairie dogs in the colony (Collier 1975; Crocker-Bedford and Spillet 1981). Prairie dogs will avoid areas where brushy species dominate, and will eventually decline or disappear in areas invaded by brush (Collier 1975; Player and Urness 1983). Well-drained soils are a habitat requirement for Utah prairie dogs to excavate burrow sites (USFWS, 2015).

***Dispersal/Migration*****Motility/Mobility**

Adult: Moderate (inferred from USFWS, 2015)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory (NatureServe, 2015)

**Dispersal**

Adult: Low (USFWS, 2015)

**Immigration/Emigration**

Adult: Emigrates from natal and breeding areas (USFWS, 2015)

**Dispersal/Migration Narrative**

Adult: This species is non-migratory (NatureServe, 2015). Traditionally, it was thought that natal dispersal (movement of first year animals away from their area of birth) and breeding dispersal (emigration of sexually mature individuals from the area where they copulated) were male-biased, leading to higher mortality rates to young males from predation (Hoogland 2003). However, recent genetic work in a range wide study showed that of the Utah prairie dogs that dispersed, 25 percent were adult females (Brown 2009). Young male Utah prairie dogs disperse in the late summer with average dispersal events of 0.35 mile (mi), long-distance dispersal events of up to 0.75 mi, and unusually long-distance dispersals of 4 mi (Mackley et al. 1988; Brown et al. 2011). In the summer of 2014 the Utah Division of Wildlife Resources documented a recently translocated individual traveling upwards of 10 miles (USFWS, 2015).

***Population Information and Trends*****Population Trends:**

Decline of > 90% (NatureServe, 2015)

**Species Trends:**

Increasing (USFWS, 2015)

**Resiliency:**

Low (inferred from NatureServe, 2015)

**Redundancy:**

Moderate (inferred from NatureServe, 2015)

**Number of Populations:**

~24 (NatureServe, 2015)

**Population Size:**

11,431 (USFWS, 2015)

**Population Narrative:**

Population in 1920 (before control programs) has been estimated at 95,000 (USFWS 1990). Historical area of occupancy has declined from about 1,800 square kilometers historically to only about 28 square kilometers today. This species has experienced a long-term decline of > 90%. On a broad scale, USFWS (1991) mapped about two dozen subpopulations (distinct patches of occupied habitat) (NatureServe, 2015). Rangewide adult counts were as high as 11,431 in the 2014 spring census count (Utah Division of Wildlife Resources (UDWR 2010a, UDWR 2015) with a low count of 1,866 in 1976. Counts of adult Utah prairie dogs from 2010 to 2014 are 5,642; 6,640; 7,979; 7,270; and 11,431 respectively (5 year average= 7,792) (UDWR 2010a, UDWR 2012, UDWR 2014, UDWR 2015) (USFWS, 2015).

***Threats and Stressors***

**Stressor:** Poisoning (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The major historical decline was primarily a result of intensive poisoning efforts. For example, in 1971, poisoning "annihilated" one of the few remaining large colonies (near Loa, Wayne County) (Pizzimenti and Collier 1975). In 1972, the largest colony (Enoch, Iron County) was reduced from more than 1,000 individuals to fewer than 50, apparently from poisoning (Pizzimenti and Collier 1975) (USFWS, 2015).

**Stressor:** Sylvatic plague (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Utah prairie dog populations are susceptible to sylvatic plague (*Yersinia pestis*), a bacterium introduced to the North American continent in the late 1800's (Cully et al. 1993). There is a limited understanding of the variables that determine when sylvatic plague will impact prairie dog populations. Fleas are the vectors that spread the disease and can be brought into the vicinity of a prairie dog colony by a suite of mammals. Plague outbreaks generally occur when populations increase to high densities causing increased stress among individuals and easier transmission of disease between individuals (USFWS, 2015).

**Stressor:** Habitat destruction (USFWS, 2015; 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Recent threats include habitat destruction resulting from residential and agricultural development on private lands (USFWS, 2015). Grazing occurs in almost all mapped and occupied

Utah prairie dog habitat including private, State, and Federal lands. Impacts from over-grazing can include decreased habitat quality resulting from increases in invasive plants and decreased vegetation diversity (Collier and Spillett 1973). OHV recreation is an increasingly common use of public lands. It is likely that OHV use results in habitat fragmentation and reduced connectivity across the species' range, increasing the likelihood of local extirpations. Direct mortality may occur as a result of collision or burrow collapse. Repeated OHV disturbances may reduce the foraging time of Utah prairie dogs and negatively affect weight gain, resulting in decreased overwinter survival. Energy resource exploration and development activities within the range of the Utah prairie dog primarily include wind and oil and gas development. These facilities can result in the loss and fragmentation of Utah prairie dog habitat and increased predation due to added perching locations for raptors. Resulting impacts to prairie dogs from oil and gas development may include direct mortality from vehicles; direct mortality associated with increased access by recreational shooters who use the new roads (Gordon et al. 2003); increased disturbance responses from increased human activity; direct loss and fragmentation of habitat and forage resources during exploration, drilling, and production; and indirect loss of forage resources from invasive nonnative plant species (Seglund and Schnurr 2009) (USFWS, 2012).

**Stressor:** Illegal killing (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Deliberate (illegal) poisoning and shooting by ranchers and farmers concerned about agricultural damage. See Iron County Commission and Utah Division of Wildlife Resources (1998) for further information and references (NatureServe, 2015; USFWS, 2015).

**Stressor:** Drought (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Drought may reduce prairie dog food resources and cause population declines in colonies on drier sites (USFWS, 2015).

**Stressor:** Inadequacy of regulatory mechanisms (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The available Federal and State regulatory mechanisms would provide some protection, but are inadequate to conserve the Utah prairie dog in the absence of the ESA's protections (USFWS, 2012).

**Stressor:** Genetic diversity (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Genetic variance within Utah prairie dog populations is low – less than half that commonly observed for black-tailed prairie dogs (Chesser 1984; Ritchie and Brown 2005; Brown 2009a). This may be the result of genetic drift in small populations (Chesser 1984). Genetic diversity can be negatively impacted by periodic population bottlenecks (e.g., caused by plague

epizootics), and by land uses that fragment Utah prairie dog colonies, decreasing dispersal and genetic exchange. Reduced gene flow between populations could be a concern for long-term population viability (Cooke 1993) (USFWS, 2012).

**Stressor:** Climate change (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The climate in southern Utah has become progressively drier over the last several thousand years, which has led to the gradual transition of grass-dominated ecosystems to those dominated by shrubs. Continued vegetation shifts may result in reduced prairie dog habitat quantity and quality over time. Thus, climate change has emerged as a significant concern for the Utah prairie dog, particularly in regard to the potential for increasingly prolonged drought cycles (USFWS, 2012).

**Stressor:** Vegetation community changes (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Potential adverse and beneficial impacts may be associated with vegetation community changes. Some of the adverse impacts from planned vegetation treatments are disturbance to prairie dogs from people or equipment, the movement of small amounts of soil or vegetation into burrow entrances, the leveling of mounds, or the loss of forage in a colony. Changes also may occur to the vegetation community from a lack of, or suppression of, naturally ignited fires. Wildfires were important historically in maintaining open or grassy areas within the shrub-steppe ecosystem. Invasive plant species alter ecological processes by displacing native species, increasing the vulnerability of communities to more invaders, and reducing wildlife habitat quality (Masters and Sheley 2001). They can be particularly damaging in areas of low moisture, including shrub-steppe habitats, because they reduce water infiltration of the soil and possess deeper roots than native species, allowing them to use more water and reduce nutrient availability over time (DiTomaso 2000). Cheatgrass also can alter fire-return intervals and dramatically expand its range after fire (Masters and Sheley 2001; BLM 2011). Site-specific fire suppression, prescribed fire, and vegetation restoration activities can impact Utah prairie dogs or their habitat if such activities occur within occupied colonies. Damage to burrows may occasionally occur as a result of using heavy equipment. Smoke, fire, noise, or other fire-related disturbances may result in harassment, displacement, injury, or possible mortality of prairie dogs, or immediate post-project alteration of key habitat components (e.g., forage or vegetation cover). Furthermore, increased human presence related to fire and vegetation management activities may alter Utah prairie dog behavior, reducing the amount of time available for the individuals to forage and causing an unnecessary expenditure of energy in fleeing and alerting others (USFWS, 2012).

### **Recovery**

**Reclassification Criteria:**

Not available

**Delisting Criteria:**

1. At least 5,000 acres (ac) (2,023 hectare (ha)) of occupied habitat are protected in perpetuity in each RU (West Desert, Paunsaugunt, and Awapa Plateau). These occupied habitat criteria will be spatially distributed to provide sufficient connectivity and gene flow within each Recovery Unit (RU) (USFWS, 2012).
2. At least 2,000 adult animals (at least 1,000 counted adults in the spring counts) are present in each RU (West Desert, Paunsaugunt, and Awapa Plateau) within protected habitat for five consecutive years (USFWS, 2012).
3. Management strategies are in place to prevent and respond to threats from disease (USFWS, 2012).
4. Education, outreach, and public relations programs and state and/or local regulations are in place and are sufficient to minimize illegal take, manage legal lethal control post-delisting, and foster habitat management practices (USFWS, 2012).
5. Utah prairie dog-specific adaptive management strategies are in place on protected lands to improve suitable habitat in a manner that also will facilitate management responses to changing climatic conditions and other threat factors that are difficult to predict (USFWS, 2012).

**Recovery Actions:**

- Prioritize Utah prairie dog habitat for protection and management (USFWS, 2012).
- Conserve habitat on non-federal lands (USFWS, 2012).
- Manage and improve Utah prairie dog habitat on federal lands (USFWS, 2012).
- Develop and implement research priorities to improve our understanding of dispersal habitat (USFWS, 2012).
- Continue agency cooperation on Utah prairie dog surveys and annual population monitoring using existing protocols throughout the designated RUs. Consider other population monitoring techniques, such as occupancy modeling, as appropriate to improve understanding of range-wide Utah prairie dog distribution and trends (USFWS, 2012).
- Cooperatively expand Utah prairie dog surveys to unmapped but potential habitat to document the species' distribution (USFWS, 2012).
- Select and prioritize translocation sites across the range of Utah prairie dogs (USFWS, 2012).
- Implement translocations in accordance with Recommended Translocation Procedures to increase the number of Utah prairie dog colonies throughout the species' range (USFWS, 2012).
- Develop and implement research priorities to improve translocation efforts (USFWS, 2012).
- Develop and implement a plague prevention and response plan. This should include prioritizing focal areas and timeframes for preventative treatments (USFWS, 2012).
- Develop and implement a monitoring strategy and database for plague (USFWS, 2012).
- Develop and implement research priorities to minimize impacts from plague/disease (USFWS, 2012).
- Identify other diseases that may impact Utah prairie dogs (USFWS, 2012).
- Develop funding strategies to implement an outreach program (USFWS, 2012).
- Establish Utah prairie dog viewing sites and educational kiosks (USFWS, 2012).
- Work with State and local governments to provide regulatory and habitat protection for the species pre- and post- delisting (USFWS, 2012).

- Develop the capability and implement actions as needed to respond to natural disturbances (e.g., drought, fire) (USFWS, 2012).

***Conservation Measures and Best Management Practices:***

- Evaluate and update the occurrence and distribution data, maps, and survey efforts for the Utah prairie dog across its known range, as information becomes available (USFWS, 2012).
- Conserve sufficient acreages and distribution of occupied Utah prairie dog habitat on Federal, State, Tribal, and private lands (USFWS, 2012).
- Minimize impacts of diseases to Utah prairie dogs via research efforts, a plague prevention and response plan, and a monitoring strategy (USFWS, 2012).
- Develop the capability and implement actions as needed to respond to natural disturbances (e.g., drought, fire) (USFWS, 2012).
- Continue the translocation of Utah prairie dogs to suitable habitat using approved protocols (USFWS, 2012).
- Develop and implement a public outreach program that promotes a better understanding of and appreciation for the biological and habitat values of the Utah prairie dog as well as tolerance of the species (USFWS, 2012).
- Develop and implement research priorities to identify and evaluate threats and create tools to expand Utah prairie dog colonies where appropriate to assist with adaptive management and conservation of the species (USFWS, 2012).
- Incorporate monitoring into recovery actions to ensure efficacy of actions (USFWS, 2012).

**References**

NatureServe. 2015. NatureServe Central Databases. Arlington, Virginia, U.S.A.

U.S. Fish and Wildlife Service. 2015. Utah Prairie Dog (*Cynomys parvidens*) Status of the Species: June 2015. U.S. Fish and Wildlife Service, West Valley City, Utah. 15 pp.

U.S. Fish and Wildlife Service. 2012. Utah Prairie Dog (*Cynomys parvidens*) Revised Recovery Plan. U.S. Fish and Wildlife Service, Denver, CO. 169 pp.

## SPECIES ACCOUNT: *Dipodomys heermanni morroensis* (Morro Bay kangaroo rat)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; October 13, 1970 (35 FR 16047).

### **Physical Description**

The Morro Bay kangaroo rat is a small, nocturnal, burrowing rodent in the Heteromyidae family. This family is related to squirrels rather than to rats and mice, and thus belongs in the superfamily Sciuroidea (USFWS 1999). Like most other kangaroo rats, it has elongate hind legs and five-clawed toes on the hind foot, relatively small front legs, long tails, and large heads with external cheek pouches. The Morro Bay kangaroo rat is smaller and more darkly colored than the most similar subspecies of Heermann's kangaroo rats (*D. heermanni* ssp.) (USFWS 2000). Adult Morro Bay kangaroo rats exhibit an average weight of 65 grams (2.3 ounces); range from 26.9 to 32 centimeters (cm) (10.6 to 12.6 inches [in.]) in length (mean length of 29.3 cm or 11.5 in.); and are sexually dimorphic, with males being larger than females (Smithsonian 2015; USFWS 1999; USFWS 2011).

### **Taxonomy**

The Morro Bay kangaroo rat is the smallest of nine subspecies of Heermann's kangaroo rat (*D. heermanni* ssp.). The Morro Bay kangaroo rat is distinguished from other subspecies of Heermann's kangaroo rat by its darker brown dorsal coloration, incomplete or absent white hip stripe, and black stripe across the nose. It is similar in size and general appearance to two other species of kangaroo rats with ranges contiguous to that of Heermann's kangaroo rat, the agile kangaroo rat (*D. agilis*) and the narrow-faced kangaroo rat (*D. venustus*); and to two noncontinuous species, Stephens' kangaroo rat (*D. stephensi*) and the San Quintin kangaroo rat (*D. gravipes*). Mitochondrial DNA sequencing conducted in 2007 determined it to be genetically differentiated from all other subspecies of Heermann's kangaroo rat (*D. heermanni* ssp.) (USFWS 2011).

### **Historical Range**

Historically, the Morro Bay kangaroo rat occurred throughout the vicinity of Los Osos, San Luis Obispo County, and was reported in 1958 within a total area of 12.4 square kilometers (km<sup>2</sup>) (4.8 square miles [sq. mi.]), of which 5.7 km<sup>2</sup> (2.2 sq. mi.) were actually occupied by the animals (the rest comprising unsuitable habitat) (USFWS 1999; USFWS 2011).

### **Current Range**

The Morro Bay kangaroo rat subspecies is restricted to the vicinity of Los Osos, San Luis Obispo County. It occurs on old, stabilized sand dunes, corresponding generally to the distribution of Baywood fine sand soil series south and southeast of Morro Bay. Since 1971, the populations of Morro Bay kangaroo rats at five of six disjunct sites have disappeared. Although the Morro Bay kangaroo rat has not been captured in the wild since 1986, suitable habitat remains within the historic distribution of this species and isolated colonies may persist; Morro Palisades is the only site considered likely to still harbor this species. However, it is estimated that less than 1 percent of optimum habitat now exists within its geographic range (USFWS 1999; USFWS 2011).

**Distinct Population Segments Defined**

No

**Critical Habitat Designated**

Yes; 8/11/1977.

**Legal Description**

On August 11, 1977, the Director, U.S. Fish and Wildlife Service thereafter, the Director and the Service, respectively) hereby issues a rulemaking which determine Critical Habitat for the Morro Bay kangaroo rat (*Dipodomys heermanni morroensis*) pursuant to Section 7 of the Endangered Species Act of 1973 (42 FR 47840 - 47845). A Correction and Augmentation Final Rule was issued on September 22, 1977 (42 FR 47840-47845).

**Critical Habitat Designation**

California. An area of land, water, and airspace in San Luis Obispo County, with the following components (Mt. Diablo Meridian): T30S R10E S1/2 Sec. 14, those portions of Sec. 23 - 24 west of Pecho Valley Road.

**Primary Constituent Elements/Physical or Biological Features**

Not available

**Special Management Considerations or Protections**

Not available

***Life History*****Feeding Narrative**

Adult: Morro Bay kangaroo rats have a generalist feeding strategy. They feed on foliage, flowers, fruits, seeds, and insects. In captivity, Morro Bay kangaroo rats ate seeds of the following local plant species when offered as food: common sandaster (*Corethrogyne filaginifolia*), common deerweed (*Lotus scoparius*), cobwebby thistle (*Cirsium occidentale*), black sage (*Salvia mellifera*), California goldenbush (*Ericameria ericoides*), sealettuce (*Dudleya caespitosa*), chamisso bush lupine (*Lupinus chamissonis*), and yellow bush lupine (*Lupinus arboreus*). In addition, the captive individuals accepted stems and leaves of trefoil (*Lotus* sp.), dudleya (*Dudleya* sp.), lupine (*Lupinus* sp.), and brome (*Bromus* sp.); and also consumed ants, crickets, grasshoppers, and garden snails. Competition with other rodents, including burrowing rodents, is considered a threat to the Morro Bay kangaroo rat. During the summer, Morro Bay kangaroo rats first appear on the surface immediately after dusk and then periodically throughout the night until 1 to 2 hours before dawn. Their foraging behavior typically involves investigating the substrate and periodically stopping for 1 to 2 minutes while the front feet are shuffled through the sand. They also forage directly on foliage, flowers, or fruits. Occasionally they stand up on their hind legs, grab at low branches with their front feet, and vigorously shake the branches. Less frequently, they may climb and move through the overstory of branches as they forage 0.5 m (20 in.) above the ground. Food items are brought to the mouth, where they are either eaten or moved to the cheek pouches. Seed stored in the cheek pouches is either hoarded in the burrow or hidden in small surface-pit-caches (USFWS 2011).

**Reproduction Narrative**

Adult: Morro Bay kangaroo rats are solitary and are considered only slightly social. Each individual's exclusive burrow system is only shared during mating encounters or while rearing pups (USFWS 1999). The majority of breeding in the wild occurs in early to mid-spring. However, captive females exhibit estrus throughout the year and may produce litters in late fall and early winter, suggesting that the narrower seasonal pattern of breeding seen in wild populations is probably a consequence of fluctuations in exogenous (external) factors such as moisture, temperature, or food supply (USFWS 1999). After a gestation period of approximately 30 days, females may give birth to one to four young and can have one to two litters per year. Kangaroo rats are naked at birth; fine hairs start to appear when they are 3 days old, and their eyes open in about 2 weeks. Weaning begins soon after, and when they are about 40 days old they learn to dig, excavating small pits with their forefeet. When they are 20 weeks old they are full grown, with an adult's coat of fur (Smithsonian 2015). Morro Bay kangaroo rats reach sexual maturity at approximately 4 months. The lifespan of the Morro Bay kangaroo rat is probably 2 to 3 years in the wild (USFWS 2011). Population and community ecology of kangaroo rats and other rodent species in deserts is influenced by combinations of interacting biotic and abiotic factors, including climate, substrate, vegetation, productivity, food, competitors, and predators. There is every reason to think that Morro Bay kangaroo rat ecology is influenced by a similar set of factors. Proper dietary intake, access to food caches, housing temperatures near thermoneutrality, and social contact all contribute to improved body condition and reproductive activity in captive Morro Bay kangaroo rats (USFWS 1999).

**Geographic or Habitat Restraints or Barriers**

Adult: Dense vegetation; fire suppression

**Spatial Arrangements of the Population**

Adult: Clumped

**Environmental Specificity**

Adult: Narrow/specialist.

**Tolerance Ranges/Thresholds**

Adult: Low

**Site Fidelity**

Adult: High

**Dependency on Other Individuals or Species for Habitat**

Adult: Primarily solitary; however, tend to have delayed dispersal, which leads to temporary family groups and long-term occupancy of the same home ranges (Randall 1993).

**Habitat Narrative**

Adult: The Morro Bay kangaroo rat lives in burrow systems in early seral stages of the chaparral community, in sandy soils with slopes of less than 15 degrees. The early seral stages of chaparral community have low and sparse vegetation, widely scattered shrubs, and medium-textured sandy loam such as southern coastal scrub, coastal sage scrub, or coastal sand plains and stabilized dunes (NatureServe 2015). Such habitats are generally characterized by somewhat lower plant species diversity; scattered areas of bare ground; increased coverage by deerweed (*Lotus scoparius*), silverweed (*Argentina anserina*), and buckbrush (*Ceanothus cuneatus*);

reduced coverage by yarrow (*Achillea millefolium*), California aster (*Corethrogyne filaginifolia*), and dudleya (*Dudleya* sp.); and moderately sparse dispersions of California sage (*Artemisia californica*), black sage (*Salvia mellifera*), mock heather (*Ericamaria ericoides*), and bush lupine (*Lupinus arboreus*) (USFWS 1999). Morro Bay kangaroo rats do not occupy dense vegetation, because it lacks their food plants, and likely inhibits their movement. As the young coastal dune scrub matures, the vegetation becomes taller and denser, which inhibits germination of annual plants. The changing habitat becomes less favorable to Morro Bay kangaroo rats as succession progresses (USFWS 2011). The Morro Bay kangaroo rat is primarily solitary; however, they tend to have delayed dispersal, which leads to temporary family groups and long-term occupancy of the same home ranges (Randall 1993).

***Dispersal/Migration*****Motility/Mobility**

Adult: Low

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Low

**Immigration/Emigration**

Adult: No

**Dependency on Other Individuals or Species for Dispersal**

Adult: No

**Dispersal/Migration Narrative**

Adult: Morro Bay kangaroo rats have low mobility and are nonmigratory. They are largely solitary and territorial; home range data collected from 40 animals on three Morro Palisades plots showed a mean home range of 0.23 hectare (ha) (0.57 acre [ac.]), and was not significantly different between males and females. Additionally, dispersal and movement of Morro Bay kangaroo rats is low. One study found that 91 percent of the maximum distances moved by 29 individuals were 50 m (164 ft.) or less. Additionally, once an animal establishes residency, it is far more likely to remain than to relocate. Juveniles, on average, tend to have a shorter mean distance moved (USFWS 1999).

**Additional Life History Information**

Adult: Morro Bay kangaroo rats remain fairly close to their main burrows and only rarely move to different localities. One study found 91 percent of the maximum distances moved by 29 individuals were 50 m (164 ft.) or less (USFWS 1999). Additionally, once an animal establishes residency, it is far more likely to remain than to relocate. Juveniles, on average, tend to have a shorter mean distance moved (USFWS 1999).

***Population Information and Trends*****Population Trends:**

Declining

**Species Trends:**

Declining

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Number of Populations:**

Unknown and possibly extinct (USFWS 2011).

**Population Size:**

Unknown and possibly extinct (USFWS 2011).

**Resistance to Disease:**

Moderate

**Adaptability:**

Low

**Additional Population-level Information:**

The low persistence rates and lack of reproduction observed when captive-born and captive-maintained Morro Bay kangaroo rats were experimentally released to the wild differs sharply from the results of translocation experiments with other kangaroo rat species. Other kangaroo rat species, both wild-caught and captive born, will breed and produce young in captivity (USFWS 1999).

**Population Narrative:**

The population and occupied habitat declined from approximately 8,000 individuals on 6.5 km<sup>2</sup> (2.5 sq. mi.) in 1957 to 50 individuals on 12.6 ha (31.1 ac.) in 1986. The Morro Bay kangaroo rat has not been observed in the wild since 1986. The U.S. Fish and Wildlife Service (USFWS) and the California Department of Fish and Wildlife consider this species to be possibly extinct. However, pockets of suitable habitat remain throughout the Los Osos area, particularly on private properties. Based on recent potential signs of Morro Bay kangaroo rat activity at the Hazard, Pecho, and Junior High/Santa Ysabel areas, some isolated colonies may still persist in pockets of suitable habitat. From 2008 to 2011, the USFWS, California Polytechnic State University, and California Department of Parks and Recreation have been conducting a search for the Morro Bay kangaroo rat. The search includes habitat assessments, visual surveys, and trapping throughout the known geographic range, and also in adjacent and nearby areas with sandy soil. In addition, the USFWS is considering habitat restoration (prescribed burns) at several places where potential signs of Morro Bay kangaroo rats have been observed recently on state lands, with the intent to present any persisting colonies with opportunities to expand

into optimum habitat (USFWS 2011). The low persistence rates and lack of reproduction observed when captive-born and captive-maintained Morro Bay kangaroo rats were experimentally released to the wild differs sharply from the results of translocation experiments with other kangaroo rat species. Other kangaroo rat species, both wild-caught and captive born, will breed and produce young in captivity (USFWS 1999).

### ***Threats and Stressors***

**Stressor:** Habitat loss

**Exposure:** Urban development in the vicinity of Los Osos, and vegetation changes from fire suppression.

**Response:** Loss of habitat, foraging resources.

**Consequence:** Fragmentation of habitat, extirpation.

**Narrative:** Since at least the early 1970s, the habitat of the Morro Bay kangaroo rat has become increasingly fragmented by development, resulting in the loss of habitat and habitat connectivity across the landscape. Additionally, fire control in the Los Osos area has contributed to succession of a more mature plant community, with dense coastal scrub and chaparral, which excludes Morro Bay kangaroo rats. It is estimated that less than 1 percent of the optimum habitat now exists in the geographic range of the Morro Bay kangaroo rat (USFWS 2011).

**Stressor:** Invasive plant species

**Exposure:** Invasive species outcompete and displace native vegetation.

**Response:** Elimination of open spaces and competition with native vegetation.

**Consequence:** Loss of habitat.

**Narrative:** Perennial veldtgrass (*Ehrharta calycina*) and other invasive species can outcompete and displace native vegetation. This is of particular concern for the Morro Bay kangaroo rat, because it relies on early seral stages of native dune mat with a relatively low vegetation cover. Dense vegetation, native and nonnative, eliminates the sparsely remaining habitat for this species (USFWS 2011).

**Stressor:** Competition with other burrowing rodents

**Exposure:** Alterations in vegetation may alter the small mammal community.

**Response:** Competition with other burrowing rodent species.

**Consequence:** Other rodent species may disturb or destroy the burrows.

**Narrative:** As the more open vegetation becomes denser due to fire control, other small mammals that might be absent at low vegetation densities may now be present. This likely puts Morro Bay kangaroo rats into competition with other species of rodents, including burrowing rodents, that may in turn disturb or destroy the burrows of Morro Bay kangaroo rats (USFWS 2011).

**Stressor:** Extreme weather events

**Exposure:** Climate change.

**Response:** Warmer air temperatures, more intense precipitation events, and increased summer continental drying.

**Consequence:** Vulnerable to extinction.

**Narrative:** Current climate change projections indicate that the northern hemisphere may experience warmer air temperatures, more intense precipitation events, and increased summer

continental drying. Limited-range species, such as the Morro Bay kangaroo rat, may be more vulnerable to extinction due to these changing conditions (USFWS 2011).

### **Recovery**

#### **Reclassification Criteria:**

Because a Morro Bay kangaroo rat was last captured in 1986 and the last captive individual died in 1993, the draft revised recovery plan has not been finalized. As of 2011, the USFWS recognized the Morro Bay kangaroo rat as possibly extinct. The following reclassification criteria are based on the draft revised recovery plan:

Based purely on genetic considerations, the Morro Bay kangaroo rat may be reclassified as threatened when an effective genetic population size of 500 has been achieved, which translates to an actual census size of about 2,000 individuals. The subspecies must have a 95 percent probability of persisting for at least 100 years. This population size must be sustained with a mean at that level for 10 consecutive years, with adequate geographic distribution (USFWS 1999).

Assuming a mean density of 10 animals per ha (four animals per ac.), approximately 200 ha (500 ac.) of functional habitat will be required for status improvement. If habitat is not managed to sustain a mean density of 10 animals per ha (four animals per ac.), more land will be required. Any change in the protected status of the Morro Bay kangaroo rat should be based on the status of the subspecies in the wild (USFWS 1999).

#### **Delisting Criteria:**

Because a Morro Bay kangaroo rat was last captured in 1986 and the last captive individual died in 1993, the draft revised recovery plan has not been finalized. As of 2011, the USFWS recognized the Morro Bay kangaroo rat as possibly extinct. Based on the Morro Bay Kangaroo Rat (*Dipodomys heermanni morroensis*) Draft Revised Recovery Plan, delisting is not likely because the limited amount of remaining historic habitat is probably insufficient to ever remove the threat of endangerment. Therefore, there are no delisting criteria available (USFWS 1999).

#### **Recovery Actions:**

- Because a Morro Bay kangaroo rat was last captured in 1986 and the last captive individual died in 1993, the draft revised recovery plan has not been finalized. As of 2011, the USFWS recognized the Morro Bay kangaroo rat as possibly extinct. However, because potential signs have been observed recently by an expert on the species, the USFWS believes that some isolated colonies may still persist in pockets of suitable habitat. Therefore, the goal at this point in time is to locate any colonies that may still persist (USFWS 2011). The following list contains recovery actions based on the draft recovery plan and recommended actions from the 5-year review:
- Remove as many as 100 Morro Bay kangaroo rats from the wild and breed them in captivity using techniques developed with a surrogate, the Lompoc kangaroo rat (*Dipodomys heermanni arenae*) (USFWS 1999).
- Identify and coordinate interagency activities to secure, manage, and improve habitat for all available areas in historic habitat (USFWS 1999).
- Reintroduce Morro Bay kangaroo rats to the wild in restored habitat, using techniques developed with the Lompoc kangaroo rat (*Dipodomys heermanni arenae*) (USFWS 1999).

- Revise the Morro Bay kangaroo rat recovery plan based on population viability analyses (USFWS 1999).
- Conduct public outreach and fundraising efforts (USFWS 1999).
- Searches for the Morro Bay kangaroo rat should continue on state lands, in particular at Hazard, Pecho, Junior High/Santa Ysabel, and Bayview. However, rather than using the traditional transect method employed in almost all previous trapping efforts, the USFWS recommends that grid trapping and/or saturation trapping be used where potential signs are observed. It is also recommended that bait stations with cameras, cameras at potential burrows, and sniffer dogs be used as additional detection methods. These methods may be more effective at detecting a species possibly persisting in isolated colonies at low density (USFWS 2011).
- Restoring the existing, mature habitat to an early stage may attract Morro Bay kangaroo rats from any nearby persisting colonies. Therefore, the USFWS recommends that habitat restoration be conducted on state lands. In particular, prescribed burns should be conducted where recent potential signs were observed by a species expert at Pecho South, Hazard South, Bayview, and Junior High/Santa Ysabel. Any prescribed burn should include a monitoring program with pre-burn/post-burn assessments (USFWS 2011).
- Morro Bay kangaroo rats were last captured on the main Buckskin property in 1985, on APN 067-011-041 in 1971, and on a property adjacent to APN 074-229-024 in 1957. The USFWS recommends that efforts continue for gaining permission to conduct surveys on the following private properties: the main Buckskin property (26.3 ha [65 ac.]), Assessor's Parcel Number (APN) 074-222-013; APN 067-011-041 (16.0 ha [40 ac.]), immediately northwest of the main Buckskin property; APN 067-012-018 (5.7 ha [14 ac.]), Junior High/Santa Ysabel area, immediately west of APN 067-011-041; and APN 074-229-024 (21.9 ha [54 ac.]), Baywood Park area. The current status of the species on these four properties cannot be determined until surveys are conducted (USFWS 2011).
- The USFWS recommends that the search area be expanded further eastward toward Edna and southward toward Pismo Beach. This is in consideration of an individual Morro Bay kangaroo rat recorded near the eastern end of Los Osos Valley Road (35.2564 degrees north, 120.6946 degrees west) in the southwestern part of the city of San Luis Obispo, approximately 12.5 kilometers (7.7 miles) beyond the known geographic range. The origin of this specimen is questioned (USFWS 2011).

***Conservation Measures and Best Management Practices:***

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***Additional Threshold Information:***

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**References**

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## SPECIES ACCOUNT: *Dipodomys ingens* (Giant kangaroo rat)

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### *Species Taxonomic and Listing Information*

**Commonly-used Acronym:** GKR

**Listing Status:** Endangered; January 5, 1987 (52 FR 283).

### **Physical Description**

The giant kangaroo rat is adapted for bipedal locomotion (two-footed hopping). The hind limbs are large compared to the forelimbs; the neck is short; and the head is large and flattened. The tail is longer than the combined head and body length; it has a dorsal crest of long hairs toward the end, and terminates in a large tuft. Large, fur-lined cheek pouches open on each side of the mouth. The pouches extend as deep invaginated pockets of skin folded inward along the sides of the head. Adult males weigh approximately 157 grams (5.54 ounces), and adult females weigh approximately 151.4 grams (5.34 ounces) (USFWS 1998). The total body length is 12.2 to 13.7 inches (in.) (311 to 348 millimeters), the tail length is 6.2 to 7.8 in. (157 to 198 millimeters), and the hind foot length is 1.8 to 2.2 in. (46 to 55 millimeters) (52 FR 283).

### **Taxonomy**

Giant kangaroo rats are distinguished from two coexisting species—San Joaquin kangaroo rat (*D. nitratoides*) and Heermann's kangaroo rat (*D. heermanni*)—by the size and number of toes on the hind foot. The hind feet of adult giant kangaroo rats each have five toes and are longer than 47 millimeters (1.85 in.). The giant kangaroo rat is the largest of more than 20 species in the genus (USFWS 1998).

### **Historical Range**

Up until the 1950s, colonies of giant kangaroo rats were spread over hundreds of thousands of hectares/acres of continuous habitat in the western San Joaquin Valley, Carrizo Plain, and Cuyama Valley. The historical distribution of giant kangaroo rats encompassed a narrow band of gently sloping ground along the western edge of the San Joaquin Valley, California. Colonies of giant kangaroo rats were found from the base of the Tehachapi Mountains in the south; to 10 miles south of Los Baños, Merced County, in the north; the Carrizo Plain and San Juan Creek watershed west of the Temblor Range to the west; and the floor of the San Joaquin Valley to the east. This area encompasses an estimated 631,724 hectares (ha) (1,561,017 acres [ac.]) of suitable habitat (USFWS 2010).

### **Current Range**

Currently, the giant kangaroo rat is found on less than 5 percent of its historic range, which is fragmented into six major geographic units: (1) the Ciervo-Panoche Region in western Fresno and eastern San Benito counties; (2) Kettleman Hills in southwestern Kings County; (3) San Juan Creek Valley in eastern San Luis Obispo County; (4) the Lokern area, Elk Hills, previously known as the National Petroleum Reserve Number One (NPR 1), which includes Buena Vista and McKittrick valleys, National Petroleum Reserve Number Two (NPR 2), Taft, and Maricopa in western Kern County; (5) the Carrizo Plain in eastern San Luis Obispo County; and (6) the Cuyama Valley along the eastern Santa Barbara-San Luis Obispo County line. These major units are fragmented into more than 100 smaller populations, many of which are isolated by several kilometers/miles of barriers such as steep terrain with plant communities unsuitable as habitat;

or agricultural, industrial, or urban land without habitat for this species (USFWS 1998; USFWS 2010). The Recovery Plan for Uplands Species of the San Joaquin Valley, California (1998) estimated total occupied habitat over the range of the giant kangaroo rat at 11,109 ha (27,450 ac.), with 2,783 ha (6,877 ac.) occupied on the Carrizo Plain. These values are estimates from range-wide surveys conducted between 1979 and 1987. As of 2009, increased range-wide surveys conducted by the California Department of Fish and Wildlife and the acquisition and restoration of natural habitat (including 16,187 ha [40,000 ac.] of occupied habitat in the Carrizo Plain National Monument) have significantly increased the amount habitat for the giant kangaroo rat, to 31,565 ha (78,000 ac.) in the Carrizo Plain (USFWS 2010).

**Distinct Population Segments Defined**

No

**Critical Habitat Designated**

No;

***Life History*****Feeding Narrative**

Adult: Giant kangaroo rats feed on seeds, especially those of peppergrass (*Lepidium nitidum*), evening primrose (*Oenothera* sp.), red brome (*Bromus rubens*), and redstem stork's bill (*Erodium cicutarium*). They also eat some green herbaceous vegetation, and occasionally insects. Giant kangaroo rats are nocturnal, foraging on the surface from around sunset to near sunrise, though most activity takes place in the first 2 hours after dark. Growth rates are variable, based on population density and available resources. Little direct evidence exists on aggression by giant kangaroo rats, but they seem to be much more aggressive than the two co-occurring species, and frequently exclude all other nocturnal rodents from areas where they occur. They temporarily bury seeds in the ground before storing them in the burrow. Giant kangaroo rats cut the ripening heads of grasses and forbs and cure them in small surface pits on the area over their burrow system. They also gather individual seeds scattered over the ground's surface and mixed in the upper layer of soil. Surface pits are uniform in diameter and depth (about 2.5 cm, 1 inch), placed vertically in firm soil, and filled with seed pods. Before being moved underground, the seeds—including redstem stork's bill and peppergrass—are sun-dried, which prevents molding (NatureServe 2015, USFWS 1998).

**Reproduction Narrative**

Adult: Giant kangaroo rats can begin to breed the year of their birth when there is sufficient food and space. They have adaptive reproductive strategies that are directly correlated with the rate of plant productivity and the population density. During times of relatively high population density, females have a short winter reproductive season with only one litter produced, and there is no breeding by young-of-the-year. This is also true both in years of high plant productivity and drought. In contrast, populations at low densities continue to breed into summer during drought (USFWS 1998). At the Soda Lake colony, juvenile females had their first litters at an estimated mean age of 5 months. Gestation lasts between 30 and 35 days. The number of reproductive events is variable. Some females may have two to three litters per year, with a mean litter size and embryo count of 3.75; however, this is considered to be a value higher than the number born, with observed live young per mother being primarily one or two young, rarely three. Most years, females are reproductive between December and March or

April, but in colonies with low densities, reproduction is extended into August or September. Additionally, there are indications that mating strategies are flexible and may be responding to the age of males, proximity of females, and changes in sex ratios. Parental care is high; young rats remain in their natal burrows until opportunity arises or they are finally driven off by the mother or one of the siblings, usually about 11 to 12 weeks after birth (USFWS 1998).

**Geographic or Habitat Restraints or Barriers**

Adult: Giant kangaroo rats are absent from areas continuously in dry-land cultivation and from irrigated fields, but may recolonize fallow dry-land grain fields if there are colonies on uncultivated land nearby (NatureServe 2015).

**Spatial Arrangements of the Population**

Adult: Clumped in remaining, fragmented habitat.

**Environmental Specificity**

Adult: Narrow; specialist.

**Tolerance Ranges/Thresholds**

Adult: Moderate

**Site Fidelity**

Adult: Moderate

**Dependency on Other Individuals or Species for Habitat**

Adult: Giant kangaroo rats live in colonies and occupy burrows with multiple individuals that appear to be family groups of females and offspring of different ages (USFWS 1998).

**Habitat Narrative**

Adult: At the time of listing, the giant kangaroo rat was believed to inhabit annual grassland communities with few or no shrubs, and sandy-loam soils on gentle slopes (approximately 10 percent); and areas receiving 6 to 7 in. of rain per year but free from flooding. Currently, the giant kangaroo rat inhabits areas of both annual grasslands and shrub communities with various soil types and slopes up to 22 percent. This broader concept of habitat suggests that current populations are found on suboptimal lands, now that the optimal grassland habitats of historic populations are under cultivation (USFWS 2010). Currently, the giant kangaroo rat habitat includes gently sloping and level piedmont plains and areas that formerly supported saltbush (*Atriplex* sp.) and perennial grasses; habitat is now dominated by introduced annuals, with many shrubs in some areas. The species occupies areas of sparse vegetative cover and well-drained soils, with a slope of generally less than 9 percent, and often with heavy grazing by cattle and sheep (NatureServe 2015). They prefer semi-arid slopes at the head of draws in barren shrubless areas, with loose, easily diggable, sandy loam soils. Giant kangaroo rats are absent from areas that are continuously in dry-land cultivation and from irrigated fields, but may recolonize fallow dry-land grain fields if there are colonies on uncultivated land nearby. Giant kangaroo rats live in colonies and occupy burrows with multiple individuals that appear to be family groups of females and offspring of different ages (USFWS 1998). The following are the preferred habitats of the giant kangaroo rat, listed in order of decreasing favorability: 1) annual grassland association in areas with less than 12 to 15 cm (5 to 6 in.) annual rain, and level to gently sloping ground; 2) alkali desert scrub association in areas with less than 12-15 cm (5-6 in.) annual rain,

sandy loam soils, and level to gently sloping ground; 3) friable soils of sand, loam, clay loam, or gravel in areas with the above characteristics; and 4) slopes of 10 to 15 degrees with the above characteristics and located near colonies in more favorable habitats (NatureServe 2015).

***Dispersal/Migration*****Motility/Mobility**

Adult: Moderate

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Moderate

**Immigration/Emigration**

Adult: Immigrates/emigrates

**Dependency on Other Individuals or Species for Dispersal**

Adult: Dispersal is dependent on population density. The major time for dispersal of giant kangaroo rats seems to be following maturation of young, about 11 to 12 weeks after birth. However, in years of high density when most or all burrow systems are occupied, most young appear to remain in their natal burrows until the opportunity to disperse arises or they finally are driven off by the mother or one of the siblings. Under these circumstances, death or dispersal of the resident does not leave a burrow system vacant for long (USFWS 1998).

**Dispersal/Migration Narrative**

Adult: Dispersal is dependent on population density. The major time for dispersal of giant kangaroo rats seems to be following maturation of young, about 11 to 12 weeks after birth. However, in years of high density when most or all burrow systems are occupied, most young appear to remain in their natal burrows until the opportunity to disperse arises or they finally are driven off by the mother or one of the siblings. Under these circumstances, death or dispersal of the resident does not leave a burrow system vacant for long. Limited data suggest that effective dispersal may extend over several kilometers (a couple of miles), and that individuals can disperse through highly inhospitable habitat (USFWS 1998).

**Additional Life History Information**

Adult: Limited data suggest that effective dispersal may extend over several kilometers (a couple of miles), and that individuals can disperse through highly inhospitable habitat (USFWS 1998).

***Population Information and Trends*****Population Trends:**

Stable in the southern range, unknown in the northern range (USFWS 2010).

**Species Trends:**

Stable

**Resiliency:**

Moderate

**Representation:**

Moderate

**Redundancy:**

Moderate

**Population Growth Rate:**

Declining

**Number of Populations:**

There are more than 100 relatively distinct populations, but most of these represent fragments of a formerly more continuous distribution (NatureServe 2015).

**Population Size:**

Adult population size varies substantially with drought and plant productivity, but ranges between 10,000 to more than 1,000,000 individuals (NatureServe 2015). Based on burrow and food-cache counts, and on capture-mark/recapture methods, the estimated subpopulations of the giant kangaroo rat in eastern San Luis Obispo County is about 21,800 in the Carrizo Plain: 20,000 in the Elkhorn Plain portion of the Carrizo Plain, 500 at Painted Rock in the Carrizo Plain, and 1,300 in the translocated population at Soda Lake in the Carrizo Plain. No estimates were given for the giant kangaroo rat subpopulations in western Kern County (USFWS 2010).

**Resistance to Disease:**

Unknown; however, the colonial living structure of the giant kangaroo rat makes them potentially susceptible to disease epidemics. Fungal disease is evident in giant kangaroo rat populations, but its potential effects on mortality or recruitment are unknown. In instances of high soil moisture (as may be produced by abnormally wet weather conditions or dense grass cover), fatal respiratory issues may develop (perhaps from the development of pathogenic toxic mold growth on seed caches). Low genetic diversity among smaller satellite populations increases their risk for elimination by random environmental events such as disease emergence (USFWS 2010).

**Adaptability:**

Moderate

**Additional Population-level Information:**

The population is currently fragmented into six major geographic units: 1) the Panoche region in western Fresno and eastern San Benito counties; 2) Kettleman Hills in Kings County; 3) San Juan Creek Valley in San Luis Obispo County; 4) western Kern County in the area of the Lokern, Elk Hills, and other uplands around McKittrick, Taft, and Maricopa; 5) Carrizo Plain in eastern San Luis Obispo County; and 6) Cuyama Valley in Santa Barbara and San Luis Obispo counties (USFWS 1998). These major units are fragmented into more than 100 smaller populations, many of which are isolated by several kilometers/miles of barriers, such as steep terrain with plant communities unsuitable as habitat; or agricultural, industrial, or urban land without habitat for this species (USFWS 1998). No estimates were given for the giant kangaroo rat subpopulations in

western Kern County. In 1992 and 1993, the population of the giant kangaroo rats in the northern range was estimated to be 37,125 over an area of 1,883 ha (20 per ha) (4,653 ac. [8.0 per ac.]). In 2005, the estimated population of the giant kangaroo rats in the northern range was about 12,375, based on burrow and food-cache counts, and on capture-mark/recapture methods. The subpopulations of the giant kangaroo rat within the northern range are estimated to be about 80 in the Ciervo Hills, 1,194 in Tumey Hills, 5,480 in Monocline Ridge, and 5,621 in the Panoche Valley (USFWS 2010). The populations of giant kangaroo rats fluctuate widely in response to inter-annual variations in precipitation and vegetation structure. Long-term population studies in the southern range on the Carrizo Plain show the species' status on protected lands to be stable. The species' status within the northern range and the satellite populations is unknown, because no long-term studies have been conducted there (USFWS 2010).

**Population Narrative:**

The population is currently fragmented into six major geographic units: 1) the Panoche region in western Fresno and eastern San Benito counties; 2) Kettleman Hills in Kings County; 3) San Juan Creek Valley in San Luis Obispo County; 4) western Kern County in the area of the Lokern, Elk Hills, and other uplands around McKittrick, Taft, and Maricopa; 5) Carrizo Plain in eastern San Luis Obispo County; and 6) Cuyama Valley in Santa Barbara and San Luis Obispo counties (USFWS 1998). These major units are fragmented into more than 100 smaller populations, many of which are isolated by several miles of barriers, such as steep terrain with plant communities unsuitable as habitat; or agricultural, industrial, or urban land without habitat for this species (USFWS 1998). The populations of giant kangaroo rats fluctuate widely in response to inter-annual variations in precipitation and vegetation structure, but range between 10,000 to more than 1,000,000 individuals (NatureServe 2015). Long-term population studies in the southern range on the Carrizo Plain show the species' status on protected lands to be stable. There are more than 100 relatively distinct populations, but most of these represent fragments of a formerly more continuous distribution (NatureServe 2015). Based on burrow and food-cache counts, and on capture-mark/recapture methods, a 2005 study estimated the subpopulations of the giant kangaroo rat within eastern San Luis Obispo County to be about 21,800, divided into three subpopulations: 20,000 in the Elkhorn Plain portion of the Carrizo Plain, 500 at Painted Rock in the Carrizo Plain, and 1,300 in the translocated population at Soda Lake in the Carrizo Plain. The species' status within the northern range and the satellite populations is unknown, because no long-term studies have been conducted there (USFWS 2010). No estimates were given for the giant kangaroo rat subpopulations in western Kern County. In 1992 and 1993, the population of the giant kangaroo rats in the northern range was estimated to be 37,125 over an area of 4,653 ac. (8.0 per ac.). In 2005, the estimated population of the giant kangaroo rats in the northern range was about 12,375, based on burrow and food-cache counts, and on capture-mark/recapture methods. The subpopulations of the giant kangaroo rat within the northern range are estimated to be about 80 in the Ciervo Hills, 1,194 in Tumey Hills, 5,480 in Monocline Ridge, and 5,621 in the Panoche Valley (USFWS 2010). Resistance to disease is unknown; however, the colonial living structure of the giant kangaroo rat makes them potentially susceptible to disease epidemics (USFWS 2010). Fungal disease is evident in giant kangaroo rat populations, but its potential effects on mortality or recruitment are unknown. In instances of high soil moisture (as may be produced by abnormally wet weather conditions or dense grass cover), fatal respiratory issues may develop (perhaps from the development of pathogenic toxic mold growth on seed caches). Low genetic diversity among smaller satellite populations

increases their risk for elimination by random environmental events such as disease emergence (USFWS 2010).

### ***Threats and Stressors***

**Stressor:** Habitat loss and degradation

**Exposure:** Broad-scale land conversion of natural habitat. Current threats continue to fragment habitat and diminish habitat quality (USFWS 2010).

**Response:** Giant kangaroo rat populations currently occupy a fraction of their historical range.

**Consequence:** The huge colonies described in historical literature no longer exist, and current populations exist in increasingly fragmented locations.

**Narrative:** Rapid habitat loss occurred between 1970 and 1979 when the natural communities on the western floor and gentle western slopes of the Tulare Basin were developed for irrigated agriculture (USFWS 1998). Approximately 98 percent of giant kangaroo rat habitat was lost before 1987 (USFWS 2010). This resulted in the loss of the huge colonies that were historically reported, and the fragmentation of the remaining habitat. This rapid habitat loss was the main reason for the listing of the giant kangaroo rat as endangered (USFWS 1998). Current threats to habitat include oil and gas exploration and extraction activities; the new development threat of large solar power plants; and urban and residential development (USFWS 2010).

**Stressor:** Overgrazing or cessation of grazing

**Exposure:** Grazing occurs over the entire range of the giant kangaroo rat.

**Response:** Diminishes habitat quality.

**Consequence:** Reduction in populations.

**Narrative:** Although habitat management with appropriate applications of grazing or fire fuels reduction measures benefit the giant kangaroo rat, there are documented studies of the negative effects of overgrazing on habitat quality. Competition occurs between the cattle and the giant kangaroo rat. Additionally, there is a potential for collapse of the burrows (USFWS 2010).

**Stressor:** Rodenticide

**Exposure:** During the 1960s through the early 1980s, rodenticides were often broadcast over large areas by airplane.

**Response:** Several of the rodenticides that were employed are detrimental to the existence of giant kangaroo rats.

**Consequence:** Extirpation of populations in historic habitat.

**Narrative:** Several areas that show characteristic features of giant kangaroo rat precincts are currently unoccupied. It is believed that populations in these areas may have been eliminated by use of rodenticides (USFWS 2010).

**Stressor:** Climate change

**Exposure:** Future climate change patterns may include drought and changes in rainfall patterns.

**Response:** Changes in the vegetative communities of giant kangaroo rat habitat.

**Consequence:** May have impacts on population size and habitat quality.

**Narrative:** The population trend of the giant kangaroo rat is highly correlated with inter-annual variations in precipitation. Years of successive drought lead to dramatic declines in the numbers of giant kangaroo rats. Additionally, years of above-normal precipitation also result in significant declines in giant kangaroo rat populations, particularly in areas that are not grazed. Changes in precipitation patterns due to climate change may have significant impacts on the giant kangaroo

rat (USFWS 2010). Changes in annual rainfall totals are the major natural ecosystem process occurring in giant kangaroo rat habitat. This process was not specifically identified in the listing rule, but in the Recovery Plan for Upland Species of the San Joaquin Valley, changes in weather patterns were linked to expansion and declines in giant kangaroo rat populations (USFWS 1998). Changes in annual rainfall can affect forage availability, the development of pathogenic toxic molds, and the availability of fuels for fire (USFWS 2010).

**Stressor:** Disease

**Exposure:** Respiratory problems, genital fungus or disease.

**Response:**

**Consequence:** Potential effects on giant kangaroo rat mortality or recruitment (USFWS 2010).

**Narrative:** Abnormally wet periods may cause some kangaroo rats to develop fatal respiratory problems, as was seen in captive Tipton kangaroo rats during an abnormally rainy February in 1995. During the 2008 annual survey on giant kangaroo rats in grazed and ungrazed plots on the Carrizo Plain, researchers discovered a genital fungus or disease on 16 percent of individual giant kangaroo rats examined (192 out of 1,210 individuals). The infection rates for juveniles was the same as that for adults, but females were infected at a higher rate than males (females 20 percent, males 12 percent). However, it is unknown if this infectious agent has potential effects on giant kangaroo rat mortality or recruitment (USFWS 2010).

## ***Recovery***

### **Reclassification Criteria:**

Reclassification to threatened status will be evaluated when the species is protected in specified recovery areas from incompatible uses; management plans have been approved and implemented for recovery areas that include survival of the species as an objective; and population monitoring indicates that the species is stable. Downlisting criteria include:

Secure and protect specified recovery areas from incompatible uses. A) All occupied lands in Carrizo Plain Natural Area and Ciervo-Panoche Natural Area B) Western Kern County areas: 1. 90 percent of the existing natural land in the Lokern area of western Kern County (bounded on the east by the California Aqueduct, on the south by Occidental of Elk Hills, on the west by State Highway 33, and on the north by Lokern Road), and 2. Naval Petroleum Reserves in California a. 90 percent of the natural land in Elk Hills (Naval Petroleum Reserve No. 1 [NPR 1]) in western Kern County, and b. 80 percent of the natural land in Naval Petroleum Reserve No. 2 (NPR 2) in western Kern County, including all in the Buena Vista/McKittrick Valley between Elk Hills Road on the southeast and State Highway 33 on the northwest, and 3. 80 percent of other occupied habitat in western Kern County.

Approve and implement management plans for all protected areas (including the Carrizo Plain Natural Area) identified as important to the continued survival of the giant kangaroo rat as an objective.

Population monitoring shows during a 5-year period no greater than a 20 percent change in population size: A) During years without drought, or B) When annual precipitation is greater than 35 percent above average.

### **Delisting Criteria:**

Delisting will be considered when, in addition to the criteria for downlisting, all of the following conditions have been met:

100 percent of occupied habitat on public lands in the Cuyama Valley, San Juan Creek Valley, and Kettleman Hills are protected, and

Approve and implement management plans for all protected areas public lands in Cuyama Valley and Kettleman Hills recovery areas identified as important to the continued survival of the giant kangaroo rat as an objective.

Populations are stable or increasing in the Carrizo, Panoche, and western Kern County metapopulations through one precipitation cycle.

**Recovery Actions:**

- 1. Of highest priority for habitat protection is proper land use and management on publicly owned and conservation lands in the Carrizo Plain Natural Area, Naval Petroleum Reserves in California, Lokern Natural Area, and Ciervo- Panoche Natural Area. Where populations of giant kangaroo rats are associated, listed species appear to be robust, land use should not be changed when ownership or conservation status of parcels changes unless there are compelling reasons to do so. For land already in public and conservation ownership, historical uses that maintained habitat for giant kangaroo rats, such as livestock grazing, should be reestablished where appropriate (USFWS 1998).
- 2. Of equal priority is supporting research on habitat management and restoration, focusing on effects of livestock grazing on habitat quality, and habitat restoration on retired farmland, especially abandoned dryland farms (USFWS 1998).
- 3. Second in priority for habitat protection is the protection of additional land supporting key populations by acquisition of title, conservation easement, or other mechanisms. Areas to be protected are prioritized, as follows:
  - a. (1) Land in the Lokern Area of western Kern County. The goal is to protect 90 percent of the existing natural land bounded on the east by natural lands just east of the California Aqueduct, on the south by Occidental of Elk Hills, on the west by State Highway 33, and on the north by Lokern Road;
  - a. (2) Land in the Naval Petroleum Reserves in California of western Kern County. The goal is to maintain in a natural state (i.e., grassland and saltbush scrub communities) 90 percent of the existing natural land in Occidental of Elk Hills, and 80 percent of the natural land in Naval Petroleum Reserve in California No. 2, including all in the Buena Vista/McKittrick Valley between Elk Hills Road on the southeast and State Highway 33 on the northwest;
  - b. Existing natural land providing habitat for giant kangaroo rats in western Fresno and eastern San Benito counties. The goal is to protect all existing natural land on the Silver Creek Ranch, and existing habitat for this species along the eastern bases of Monocline Ridge and the Tumey Hills, between Arroyo Ciervo on the south and Panoche Creek on the north;
  - c. Acquire and restore habitat on periodically farmed land with no or Class-3 irrigation water rights immediately east of occupied natural habitat along the strip described in (3.b., above), and west of Interstate Highway 5;
  - d. Other natural land occupied by giant kangaroo rats in western Kern County. The goal is to protect 80 percent of existing habitat for giant kangaroo rats;

- e. Land occupied by giant kangaroo rats in the Cuyama Valley, Santa Barbara County;
- f. Land occupied by giant kangaroo rats in the Kettleman Hills, Kings County;
- f. Land occupied by giant kangaroo rats in the Kettleman Hills, Kings County;
- No formal guidelines containing conservation measures have been developed for this species. The USFWS (U.S. Fish and Wildlife Service) Giant Kangaroo Rat 5-Year Review (2010) provides a number of recommendations for actions over the next 5 years.
- Locations that should be targeted for protection: a) Dispersal corridors within the northern range along Panoche Creek and Silver Creek in western Fresno County b) The Panoche Valley in eastern San Benito County as an important source of regional expansion within the northern range c) Buena Vista Valley in western Kern County d) Co-locate the conservation lands acquired for San Joaquin kit fox and blunt-nosed leopard lizard with giant kangaroo rat habitat (USFWS 2010).
- Kern County – completion of Habitat Conservation Plans (HCPs) and issuance of incidental take permits: a) Draft Kern County Valley Floor HCP b) Draft Chevron Lokern HCP c) Draft Occidental Petroleum of Elk Hills HCP d) Encourage Crimson Resource Management to start an HCP or Section 7 formal consultation to protect lands in Buena Vista Valley, NPR 2, and Buena Vista Hills (USFWS 2010).
- Approval and implementation of habitat management plans: a) Establishment of the 44,000-ac. Lokern Natural Area in western Kern County b) Include in all habitat management plans (including the Carrizo Plain National Monument) the flexibility to alter the dates and stocking rates of livestock to respond to annual plant production to prevent the dominance of exotic grasses in giant kangaroo rat habitat (USFWS 2010).
- Future research and monitoring: a) Continued long-term monitoring in western Kern County and Carrizo Plain b) Begin long-term monitoring of populations within the Ciervo-Panoche area of western Fresno and eastern San Benito counties c) Census and monitor giant kangaroo rats in the satellite populations in the Cuyama Valley (eastern San Luis Obispo and eastern Santa Barbara counties), San Juan Creek Valley (eastern San Luis Obispo County), and Kettleman Hills (southwestern Kings County) (USFWS 2010)

***Additional Threshold Information:***

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**References**

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## SPECIES ACCOUNT: *Dipodomys merriami parvus* (San Bernardino Merriam's kangaroo rat)

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### *Species Taxonomic and Listing Information*

**Commonly-used Acronym:** SBKR

**Listing Status:** Endangered; 01/27/1998; California/Nevada Region (R8) (USFWS, 2016)

### **Physical Description**

The San Bernardino kangaroo rat (*Dipodomys merriami parvus*; SBKR) is a small, dark-colored kangaroo rat. In coastal southern California, *D. merriami* is the only kangaroo rat with four toes on each of its hind feet. It has a body length of about 95 millimeters (mm) (3.7 inches [in.]) and a total length of 230 to 235 mm (9.0 to 9.3 in.). The hind foot measures less than 36 mm (1.4 in.) in length. The body color is pale yellow, with a heavy overwash of dusky brown. The tail stripes are medium to dark brown, and the foot pads and tail hairs are dark brown. The flanks and cheeks of the subspecies are dusky. The flanks and cheeks of the subspecies are dusky. The San Bernardino kangaroo rat, endemic to southern California, is one of the most highly differentiated subspecies of Merriam's kangaroo rat (65 FR 77178; 67 FR 19812).

### **Taxonomy**

The San Bernardino kangaroo rat is one of 19 recognized subspecies of Merriam's kangaroo rat (*D. merriami*), a widespread species distributed throughout arid regions of the western United States and northwestern Mexico. In coastal southern California, *D. merriami* is the only kangaroo rat with four toes on both of its hind feet. Additionally, it is considerably darker and much smaller than either of the other two subspecies of Merriam's kangaroo rat in southern California—Merriam's kangaroo rat (*D. merriami merriami*) and Earthquake Merriam's kangaroo rat (*D. merriami collinus*). It is also one of the most highly differentiated subspecies of Merriam's kangaroo rat, likely due to its apparent isolation from other members of *D. merriami* (63 FR 51005).

### **Historical Range**

The historical range of the San Bernardino kangaroo rat extends from the San Bernardino Valley in San Bernardino County to the Menifee Valley in Riverside County. Prior to 1960, this subspecies was known from more than 25 localities within this range. From the early 1880s to the early 1930s, it was a common resident of the San Bernardino and San Jacinto valleys of southern California (67 FR 19812). Based on aerial photography, museum records, field surveys, and literature, the historical range is thought to have encompassed roughly 11,331 hectares (ha) (28,000 acres [ac.]) of alluvial floodplain habitat. This historical range of the San Bernardino kangaroo rat was thought to include the extensive alluvial fan terraces at the bases of the San Gabriel, San Bernardino, and San Jacinto mountain ranges in San Bernardino and Riverside counties, California. The northern extent of this subspecies range was likely the Cajon Pass in San Bernardino County, and the southernmost extent was in Menifee in Riverside County (USFWS 2009).

### **Current Range**

The majority of the remaining San Bernardino kangaroo rat populations are primarily found in three areas: the Santa Ana Wash, the San Jacinto Wash, and Lytle Creek and Cajon Wash. Other smaller populations of the San Bernardino kangaroo rat are documented in washes and hills in the areas surrounding the three main population centers (67 FR 19812). The current range of the species encompasses at least 4,328 ha (10,696 ac.). Although this area does not encompass all habitat occupied by or suitable for the San Bernardino kangaroo rat, we believe that they do represent much of the remaining occupied habitat (USFWS 2009).

**Distinct Population Segments Defined**

No

**Critical Habitat Designated**

Yes; 10/17/2008.

**Legal Description**

On October 17, 2008, the U.S. Fish and Wildlife Service (Service), designated final revised critical habitat for the San Bernardino kangaroo rat (*Dipodomys merriami parvus*) under the Endangered Species Act of 1973, as amended (Act). Approximately 7,779 acres (ac) (3,148 hectares (ha)) of habitat in San Bernardino and Riverside Counties, California, were designated as critical habitat for the San Bernardino kangaroo rat. The final revised designation constitutes a reduction of approximately 25,516 ac (10,326 ha) from the 2002 designation of critical habitat for the San Bernardino kangaroo rat.

**Critical Habitat Designation**

Approximately 7,779 ac (3,148 ha) of land are designated as critical habitat for the San Bernardino kangaroo rat in five units.

Unit 1: Santa Ana River Wash Unit 1 consists of approximately 3,258 ac (1,318 ha) and is located in San Bernardino County. This unit includes the Santa Ana River and portions of City, Plunge, and Mill Creeks. The area includes lands within the cities of San Bernardino, Redlands, and Highland. Although Seven Oaks Dam (northeast of Unit 1) impedes sediment transport and reduces the magnitude, frequency, and extent of flood events from the Santa Ana River, the system still retains partial fluvial dynamics because Mill Creek is not impeded by a dam or debris basin. This critical habitat unit was occupied at the time of listing, is currently occupied, and contains all of the features essential to the conservation of the San Bernardino kangaroo rat. Additionally, this unit contains the highest densities of San Bernardino kangaroo rats in the Santa Ana wash. The physical and biological features contained within this unit may require special management considerations or protection to minimize impacts associated with flood control operations, water conservation projects, sand and gravel mining, and urban development. Approximately 751 ac (304 ha) of revised proposed critical habitat Unit 1 occurred within the WSPA, a section of the floodplain downstream of Seven Oaks Dam that was preserved by the flood control districts of Orange, Riverside, and San Bernardino Counties. The WSPA was established in 1988 by the ACOE to minimize the effects of Seven Oaks Dam on the federally endangered plant, *Eriastrum densifolium* ssp. *sanctorum* (Santa Ana River woolly-star). This area of alluvial fan scrub in the wash near the low-flow channel of the river was identified for preservation because these sections of the wash were thought to have the highest potential to maintain the hydrology necessary for the periodic regeneration of early phases of alluvial fan sage scrub. A 1993 Management Plan for the Santa Ana River WSPA has been completed, and a draft MSHMP for

WSPA lands, which includes protection for the San Bernardino kangaroo rat, is to be completed as an additional conservation measure pursuant to our December 19, 2002, biological opinion on operations for Seven Oaks Dam (Service 2002b, p. 8). As a result of our partnership and development of approved management plans, we excluded the approximately 751 ac (304 ha) of WSPA lands from the final revised critical habitat designation (see “Exclusions Under Section 4(b)(2) of the Act” section for a detailed discussion). In 1994, the BLM designated three parcels in the Santa Ana River, a total of approximately 760 ac (308 ha), as an ACEC. One parcel is located south of the Seven Oaks borrow pit, another is farther west and south of Plunge Creek, and the third is located farther west between two large mining pits. The primary goal of this ACEC designation is to protect and enhance the habitat of federally listed plant species occurring in the area while providing for the administration of valid existing water conservation rights. Although the establishment of this ACEC is important in regard to conservation of sensitive species and vegetation communities in this area, the administration of existing water conservation rights conflicts with the BLM’s ability to manage their lands for the San Bernardino kangaroo rat. Existing rights include a withdrawal of Federal lands for water conservation through an act of Congress on February 20, 1909 (Public Law 248, 60th Cong., 2nd sess.). The entire ACEC is included in this withdrawn land and may be used for water conservation measures, such as the construction of percolation basins. Although the BLM is coordinating with the Service to conserve San Bernardino kangaroo rat habitat, at this time we do not consider these lands to be managed for the benefit of the San Bernardino kangaroo rat or its PCEs, and we are not excluding these lands from the final revised critical habitat designation. We are currently coordinating with the BLM, ACOE, San Bernardino Valley Conservation District, Cemex Construction Materials, Robertson’s Ready Mix, and other local interests on a proposed exchange of Federal and private lands and the development of the Upper Santa Ana River Habitat Conservation Plan (USAR HCP, also known as “Plan B”). The goal of the USAR HCP is to consolidate a large block of alluvial fan scrub occupied by three federally endangered species (the San Bernardino kangaroo rat, *Eriastrum densifolium* ssp. *sanctorum*, and *Dodecahema leptoceras* (slender-horned spineflower)) and one federally threatened species (the coastal California gnatcatcher (*Polioptila californica californica*)). The area under consideration includes the majority of the Santa Ana wash from just downstream of the confluence of Mill Creek with the Santa Ana River to Alabama Street. While the goal of this effort is to benefit the San Bernardino kangaroo rat through the establishment of preserve lands that will be managed for this subspecies and other listed species, we are still in the development phase of this HCP, and we are not excluding lands within the proposed Santa Ana River Wash Conservation Area from the final revised critical habitat designation. Approximately 267 ac (108 ha) of occupied habitat in the Santa Ana River wash is set aside for conservation in perpetuity by the U.S. Air Force as part of on-base site remediation efforts at the former Norton Air Force Base in San Bernardino, California. These areas are managed specifically for the San Bernardino kangaroo rat and *Eriastrum densifolium* spp. *sanctorum* pursuant to the Former Norton Air Force Base CMP completed in March 2002. We excluded these 267 ac (109 ha) from the final revised critical habitat designation based on benefits provided to San Bernardino kangaroo rat habitat through our partnership and the approved CMP.

Unit 2: Lytle/Cajon Creek Wash Unit 2 encompasses approximately 3,421 ac (1,384 ha) in San Bernardino County and includes the northern extent of this subspecies’ remaining distribution. This unit contains habitat along and between Lytle and Cajon Creeks from the Interstate 15 Bridge in Lytle Creek and the Kenwood Avenue/ Cajon Boulevard junction in Cajon Creek, downstream to Highland Avenue. Unit 2 was occupied at the time of listing, is currently occupied,

and contains all of the features essential to the conservation of the San Bernardino kangaroo rat. This unit includes some of the last remaining alluvial fans, floodplain terraces, historical braided river channels, and associated alluvial sage scrub and upland vegetation that provides habitat for the San Bernardino kangaroo rat in the Lytle/Cajon Creek wash. This unit also contains the highest densities of San Bernardino kangaroo rat in the Lytle/Cajon wash. The physical and biological features within this unit may require special management considerations or protection to minimize impacts associated with flood control operations, water conservation projects, sand and gravel mining, and urban development. The hydro-geomorphological processes that apparently rejuvenate and maintain the dynamic mosaic of alluvial fan sage scrub are still largely intact in Lytle and Cajon Creeks (i.e., stream flows are not impeded by dams or debris basins), and the remaining habitat allows dispersal between these two drainages, which is important for genetic exchange between populations (67 FR 19812, April 23, 2002). This unit is adjacent to large tracts of undeveloped land and contains upland areas occupied by the subspecies (PCEs 1, 2, and 3). Several areas that were proposed in Unit 2 will be or are protected and managed to some extent for the San Bernardino kangaroo rat. The Cajon Creek Habitat Conservation Management Area (HCMA) includes approximately 1,265 ac (512 ha) to offset approximately 2,270 ac (919 ha) of sand and gravel mining proposed within and adjacent to Cajon Creek. Of the 1,265 ac (512 ha) Cajon Creek HCMA, approximately 567 ac (229 ha) is the Cajon Creek Conservation Bank established to help conserve populations of 24 species associated with alluvial fan scrub, including the San Bernardino kangaroo rat. Furthermore, the remaining 698 ac (282 ha) are set aside as permanent conservation lands. These conservation lands will be managed in perpetuity for alluvial fan scrub habitat and associated listed species (including the San Bernardino kangaroo rat) pursuant to the HEMP (M. Blane and Associates 1996) and associated Memorandum of Understanding and Implementation Agreement for the Cajon Creek Habitat Management Area (MOU) (CalMat Company 1996). We excluded 1,265 ac (512 ha) of HCMA lands from the final revised critical habitat designation based on our partnership and benefits provided by the HEMP and MOU (see "Exclusions Under Section 4(b)(2) of the Act" for a detailed discussion). In 2003, the Service issued a biological opinion for the Lytle Creek North Master Planned Community, which falls within the boundary of existing San Bernardino kangaroo rat habitat (Service 2003a, FWS-SB- 1640.11). The project includes an approximately 677 ac (274 ha) master planned community with over 2,400 residential units. Construction activities are proposed to be phased over an estimated 5 to 10 years. As an off-site measure for this project, the Lytle Creek Development Company will dedicate approximately 213 ac (86 ha) of largely undeveloped habitat within Lytle Creek (Unit 2) as a conservation area for the San Bernardino kangaroo rat. Habitat that provides primary foraging, sheltering, and breeding habitat for the San Bernardino kangaroo rat within this area will be conserved and managed in perpetuity (Service 2003a, p. 45). Forty acres (16 ha) of this area is upland island habitat that lies within the floodplain and will receive additional management through restoration or enhancement for the benefit of the San Bernardino kangaroo rat (Service 2003a, p. 42). A long-term management plan will be completed at the end of an initial management period allowing for lessons learned during that time to be incorporated into the long-term management plan. However, to date, no conservation easements or endowments have been secured for the lands proposed as conservation areas, nor has the long-term management plan been completed, and we are not excluding the 213 ac (86 ha) of proposed future conservation lands that will be established as a result of this project from the final revised critical habitat designation. On June 15, 1999, we issued our biological opinion on the construction and extension of the north levee at Sunwest Materials' (now CEMEX) Lytle Creek Quarry (Service 1999, 1-6-99-F- 42). The armored, engineered levee (over 10,000 feet (3,048 meters) in length) protects mining operations from

flooding and replaces a shorter, earthen embankment (Service 1999, p. 3). As a conservation measure for this project, Sunwest Materials delivered to the California Department of Fish and Game a conservation easement deed to approximately 26 ac (11 ha) delineated as Conservation Area 1 to protect biological resources in perpetuity (Service 1999, p. 7). Additionally, Sunwest Materials is to record a biological resource deed restriction on approximately 12 ac (5 ha) of land to permanently preclude activities that would interfere with habitat value (Service 1999, p. 8). However, a management plan benefiting the San Bernardino kangaroo rat is not yet developed for these lands, and we are not excluding these 38 ac (16 ha) from the final revised critical habitat designation.

**Unit 3: San Jacinto River Wash** Unit 3 encompasses approximately 506 ac (205 ha) in Riverside County and includes areas along the San Jacinto River in the vicinity of San Jacinto, Hemet, and Valle Vista. This unit encompasses the San Jacinto River wash from the Blackburn Road/Lake Hemet Main Canal area, downstream to the East Main Street Bridge. This unit includes all of the features essential to the conservation of the San Bernardino kangaroo rat, was occupied at the time of listing, and is currently occupied. Additionally, this unit contains one of only three large extant core populations of the San Bernardino kangaroo rat and is the only core population in Riverside County. Historically, the San Bernardino kangaroo rat occurred along the San Jacinto River from the upper reach of habitat in the river downstream past State Route 79. The physical and biological features within this unit may require special management considerations or protection to minimize impacts associated with flood control operations, channelization, water conservation projects (groundwater recharge ponds), off-road vehicle activity, and urban development. Lands within Unit 3 are adjacent to lands of the Soboba Band of Luisen~o Indians Reservation, which were included in the 2002 final critical habitat designation (see 50 CFR 17.95(a); 67 FR 19812, April 23, 2002). We are not designating these Tribal lands as critical habitat for the San Bernardino kangaroo rat in this final revised critical habitat designation (see "Government-to-Government Relationship with Tribes" section for a detailed discussion). All private lands proposed as critical habitat in the San Jacinto River wash fall within the boundaries of the Western Riverside County MSHCP. We excluded private lands under the jurisdiction of permittees to the MSHCP and all lands owned and managed by permittees to the MSHCP within this area (263 ac (106 ha)) based on our partnership and the benefits provided to the San Bernardino kangaroo rat by the Western Riverside County MSHCP. We are also excluding 39 ac (16 ha) of land owned by the Eastern Municipal Water District related to The Soboba Band of Luisen~o Indians Settlement Act and implementation of its associated settlement agreement.

**Unit 4: Cable Creek Wash** Unit 4 consists of approximately 483 ac (195 ha) and is located in San Bernardino County. This unit encompasses the Cable Creek alluvial floodplain from the mouth of Cable Canyon to I-215 where the creek becomes channelized. Because Cable Creek is not impeded by a dam or debris basin, the fluvial dynamics necessary to maintain the PCEs of San Bernardino kangaroo rat habitat remain in this unchannelized portion of Cable Creek. This critical habitat unit was occupied at the time of listing, is currently occupied, and contains all of the features essential to the conservation of the San Bernardino kangaroo rat. Additionally, this unit contains a likely self-sustaining population of San Bernardino kangaroo rats that may be important for the long-term conservation of the subspecies. This unit is demographically isolated from the core population of the subspecies in the Lytle/Cajon wash (Unit 2). A stochastic event causing dramatic population decline or local extirpation in Unit 2 may have little effect on Unit 4. In such a case, the population in Unit 4 could serve as a source of individuals for repopulating Unit 2. The physical and biological features contained within this unit may require special

management considerations or protection to minimize impacts associated with flood control operations, water conservation projects, sand and gravel mining, and urban development.

Unit 5: Bautista Creek Unit 5 consists of approximately 111 ac (45 ha) and is located in Riverside County. This unit includes occupied habitat from the unchannelized reach of Bautista Creek (i.e., from the existing instream mining operation to upstream areas where the grade of the creek precludes the formation of alluvial terraces or braids). This unit represents the southernmost extent of the San Bernardino kangaroo rat's current range. The wash system in upper Bautista Creek retains fluvial dynamics because it is not impeded by a dam, debris basin, or concrete channelization. This critical habitat unit was occupied at the time of listing, is currently occupied, and contains all of the features essential to the conservation of the San Bernardino kangaroo rat. Historically, the subspecies occurred upstream of the Bautista flood control basin until the topography of the canyon becomes too steep. This unit contains agricultural areas that could be occupied at low densities by this subspecies (PCE 3). Additionally, this unit contains a likely self-sustaining population of San Bernardino kangaroo rats that may be important for the long-term conservation of the subspecies. This unit is demographically isolated from the core population of the subspecies in the San Jacinto wash (Unit 3) by a concrete-lined channel. This channel directs flows from upper Bautista Creek downstream to the San Jacinto River. Given the current status of the San Bernardino kangaroo rat and ongoing threats to its habitat, it is important for the conservation of the San Bernardino kangaroo rat that natural fluvial processes in occupied habitat are maintained. A stochastic event could cause a dramatic population decline or local extirpation in either Units 3 or 5. In such a case, through relocation for the purposes of recovery, the population in Unit 5 could serve as a source of individuals for repopulating Unit 3, and vice versa. The physical and biological features contained within this unit may require special management considerations or protection to minimize impacts associated with agricultural activities, sand and gravel mining, and urban development. All private lands proposed as critical habitat in Bautista Creek fall within the boundaries of the Western Riverside County MSHCP. We excluded private lands under the jurisdiction of permittees to the MSHCP and all lands owned and managed by permittees to the MSHCP within this area (332 ac) based on our partnership and the benefits provided to the San Bernardino kangaroo rat by the Western Riverside County MSHCP.

#### **Primary Constituent Elements/Physical or Biological Features**

Critical habitat units are designated for San Bernardino and Riverside Counties, California. The PCEs of critical habitat for the San Bernardino kangaroo rat are the habitat components that provide:

- (i) Alluvial fans, washes, and associated floodplain areas containing soils consisting predominately of sand, loamy sand, sandy loam, and loam, which provide burrowing habitat necessary for sheltering and rearing offspring, storing food in surface caches, and movement between occupied patches;
- (ii) Upland areas adjacent to alluvial fans, washes, and associated floodplain areas containing alluvial sage scrub habitat and associated vegetation, such as coastal sage scrub and chamise chaparral, with up to approximately 50 percent canopy cover providing protection from predators, while leaving bare ground and open areas necessary for foraging and movement of this subspecies; and

(iii) Upland areas adjacent to alluvial fans, washes, and associated floodplain areas, which may include marginal habitat such as alluvial sage scrub with greater than 50 percent canopy cover with patches of suitable soils that support individuals for re-population of wash areas following flood events. These areas may include agricultural lands, areas of inactive aggregate mining activities, and urban/wildland interfaces.

### **Special Management Considerations or Protections**

Critical habitat does not include manmade structures (such as buildings, aqueducts, airports, roads, other paved areas, and the land on which such structures are located) existing on the effective date of this rule and not containing one or more of the PCEs.

Special management considerations or protection may be required to minimize effects of mining activities on alluvial sage scrub habitat and the natural hydrological processes that maintain proper alluvial sage scrub conditions for the San Bernardino kangaroo rat.

Special management considerations or protection may be required to minimize effects of flood control and water conservation activities on alluvial sage scrub habitat and the natural hydrological processes that maintain proper alluvial sage scrub conditions for the San Bernardino kangaroo rat.

Special management considerations or protection may be required to minimize the impacts of development within the alluvial wash and adjacent upland areas. Areas of the alluvial washes and floodplains adjacent to development may require exclusionary fencing and signage to minimize human and domestic animal disturbance of San Bernardino kangaroo rat habitat. Because this subspecies is active at night, lights from adjacent developed areas should be minimized and directed away from San Bernardino kangaroo rat habitat.

Special management considerations or protection may be required to minimize effects of agricultural activities on alluvial sage scrub habitat.

Special management considerations or protection, such as exclusionary fencing, additional enforcement, and signage placed around areas of the wash, may be needed to minimize impacts from unauthorized off-road vehicle use.

### ***Life History***

#### **Feeding Narrative**

Adult: Typical of kangaroo rats, Merriam's kangaroo rats are primarily nocturnal, granivorous, and often store large quantities of seeds in surface caches. Seed caching may enable them to endure temporary shortages of food. Water is obtained metabolically from moisture in food. Although seeds are the primary food source, green vegetation (primarily filaree [*Erodium* sp.]) and insects appear to be important seasonal food and water sources for the San Bernardino kangaroo rat (USFWS 2009; Zeiner et al. 1990). Insects, when available, have been documented to constitute as much as 50 percent of a kangaroo rat's diet (67 FR 19812). Because the distribution of the San Bernardino kangaroo rat appears to be driven by soils type (sandy loam substrains) that are characteristic of alluvial fans and floodplains, the hydrologic regime in the alluvial fans supporting the subspecies is of critical importance. Maintaining habitat connectivity between upland terrace habitat and the channel for the movement of animals between upland

and instream habitat is critical to support animals in both locations (USFWS 2009). The San Bernardino kangaroo rat is active year-round; and the species has a rapid growth rate, because females are capable of breeding shortly after weaning (as soon as 2 to 3 months of age) (Ceballos 2014; Zeiner et al. 1990).

### **Reproduction Narrative**

Adult: Although reproductive activities peak in June and July, the San Bernardino kangaroo rat appears to have a prolonged breeding season. Pregnant or lactating females have been captured between January and November, while males in reproductive condition have been captured between January and August (USFWS 2009). This subspecies may breed several times a year, typically once to twice, with the ability to confine reproduction to periods of favorable conditions (Ceballos 2014; Zeiner et al. 1990). The San Bernardino kangaroo rat has a gestation period of 33 days, and typically birth between two and three young per litter (USFWS 2009). Weaning occurs 24 to 33 days after birth, and females are capable of breeding shortly after weaning. Merriam's kangaroo rat (*D. merriami*) exhibit an average lifespan of 3.5 years (Ceballos 2014). Kangaroo rat populations typically exhibit large fluctuations in density in response to temporal variability in plant productivity. Reproduction appears to be timed to coincide with high food-availability, but the rate of population growth is limited by the relatively small size of litters and long intervals between litters (USFWS 2009).

### **Geographic or Habitat Restraints or Barriers**

Adult: Habitat fragmentation due to development and related activities in the San Bernardino and San Jacinto valleys act as a habitat restraint (67 FR 19812).

### **Spatial Arrangements of the Population**

Adult: Clumped

### **Environmental Specificity**

Adult: Narrow/specialist

### **Tolerance Ranges/Thresholds**

Adult: Moderate

### **Site Fidelity**

Adult: High

### **Dependency on Other Individuals or Species for Habitat**

Adult: No

### **Habitat Narrative**

Adult: San Bernardino kangaroo rats are typically found on alluvial fans (relatively flat or gently sloping masses of loose rock, gravel, and sand deposited by a stream as it flows into a valley or upon a plain); floodplains; along washes; in adjacent upland areas containing appropriate physical and vegetative characteristics; and in areas with historic braided channels. These areas consist of sand, loam, sandy loam, or gravelly soils that are associated with alluvial processes. San Bernardino kangaroo rat also occupy areas where winds contribute to the deposition of sandy soils. The soils deposited by alluvial or wind-driven processes typically support alluvial sage scrub and chaparral vegetation, and allow kangaroo rats to dig simple, shallow burrow

systems. The burrow systems of adults are often clustered in a given area, and adults actively defend small core areas near their burrows (USFWS 2009). Alluvial sage scrub is a relatively open vegetation type that is adapted to periodic flooding and erosion, and is composed of an assortment of drought-deciduous shrubs and larger evergreen woody shrubs characteristic of both coastal sage scrub and chaparral communities (73 FR 61936). Habitat fragmentation due to development and related activities in the San Bernardino and San Jacinto valleys act as a habitat restraint (67 FR 19812).

***Dispersal/Migration*****Motility/Mobility**

Adult: Moderate

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Moderate; in a mark-recapture study conducted between 1978 and 1984, it was found that 75 percent of adult male and 59 percent of adult female Merriam's kangaroo rats dispersed more than 60 meters (m) (197 feet [ft.]) from their initial capture sites, and that some individuals of both sexes moved more than 240 m (787 ft.) before they were no longer found by researchers. In a similar study, male San Bernardino kangaroo rats of reproductive age routinely moved as much as 40 m (131 ft.)—a conservative estimate based on distance between locations where they were trapped, which is not necessarily the total distance an animal may have actually travelled. Long-range dispersal and population expansion by the San Bernardino kangaroo rat is likely hampered by the presence of other rodents (USFWS 2009).

**Immigration/Emigration**

Adult: No

**Dependency on Other Individuals or Species for Dispersal**

Adult: No

**Dispersal/Migration Narrative**

Adult: The San Bernardino kangaroo rat has moderate motility and is nonmigratory. Habitat connectivity and sufficient food resources are needed for dispersal. Little is known about home range size, dispersal distances, or other spatial requirements of the San Bernardino kangaroo rat. In a mark-recapture study conducted between 1978 and 1984, it was found that 75 percent of adult male and 59 percent of adult female Merriam's kangaroo rats dispersed more than 60 m (197 ft.) from their initial capture sites, and that some individuals of both sexes moved more than 240 m (787 ft.) before they were no longer found by researchers. In a similar study, male San Bernardino kangaroo rats of reproductive age routinely moved as much as 40 m (131 ft.)—a conservative estimate based on distance between locations where they were trapped, which is not necessarily the total distance an animal may have actually travelled. Long-range dispersal and population expansion by the San Bernardino kangaroo rat is likely hampered by the presence of other rodents (USFWS 2009). Home ranges for the Merriam's kangaroo rat in the Palm Springs, California, area averaged 0.33 ha (0.82 ac.) for males and 0.31 ha (0.77 ac.) for females. Much larger home ranges have been reported for Merriam's kangaroo rats in New

Mexico, where home ranges averaged 1.7 ha (4.1 ac.) for males and 1.6 ha (3.9 ac.) for females. Space requirements for the San Bernardino kangaroo rat likely vary according to season, age and sex of animal, food availability, and other factors. Although outlying areas of their home ranges may overlap, *Dipodomys* adults actively defend small core areas near their burrows (73 FR 61936). Juveniles exhibit high fidelity to natal areas (Ceballos 2014).

**Additional Life History Information**

Adult: Little is known about home range size, dispersal distances, or other spatial requirements of the San Bernardino kangaroo rat. However, home ranges for the Merriam's kangaroo rat in the Palm Springs, California, area averaged 0.82 ac. (0.33 ha) for males and 0.77 ac. (0.31 ha) for females. Much larger home ranges have been reported for Merriam's kangaroo rats in New Mexico, where home ranges averaged 4.1 ac. (1.7 ha) for males and 3.9 ac. (1.6 ha) for females. Space requirements for the San Bernardino kangaroo rat likely vary according to season, age and sex of animal, food availability, and other factors. Although outlying areas of their home ranges may overlap, *Dipodomys* adults actively defend small core areas near their burrows (73 FR 61936).

***Population Information and Trends*****Population Trends:**

Decreasing; population has declined dramatically. Historic range has been reduced by about 96 percent; reduced from 25 historic locations to 7 (8 identified by others) currently occupied sites. Habitat loss and fragmentation is continuing (NatureServe 2015).

**Species Trends:**

Decreasing (NatureServe 2015)

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Number of Populations:**

Occurs at eight localities. Four localities (Santa Ana River, Lytle and Cajon washes, and Etiwanda Creek) contain moderately large populations and four (Badlands, Bautista Canyon, San Timoteo Creek, and San Jacinto River near Hemet) have small populations in fragmented and isolated habitat patches (Bolster 1998).

**Population Size:**

Unknown; the highest densities occur in the Santa Ana River Wash and Cajon Wash (NatureServe 2015). Actual densities not known; female home ranges (which overlap only slightly) for this species in other regions range from 0.31 to 1.6 ha (0.76 to 3.95 ac.) (NatureServe 2015). In 1997, San Bernardino kangaroo rat occupied about 6,576 ha (16,250 ac.) of habitat (NatureServe 2015). Population size is highly variable, and is correlated with percent

vegetation cover and vegetation type as well as variations in substrate (percent sand, gravel, and cobble) (USFWS 2009).

**Resistance to Disease:**

Moderate

**Adaptability:**

Low

**Additional Population-level Information:**

Population density studies have documented substantial annual variation. There appear to be several reasons for these greatly disparate values, including: 1) a low population density observed in an area at one point in time does not mean that the area is occupied at the same low density in any other month, season, or year; 2) a low population density is not an indicator of low habitat quality or low overall value of the land for the conservation of the species; 3) an abundance of San Bernardino kangaroo rat can decrease rapidly; and 4) one or more factors (e.g., food availability, fecundity, disease, predation, genetics, and environment) are strongly influencing the species' population dynamics in one or more areas (67 FR 19812).

**Population Narrative:**

Populations of San Bernardino kangaroo rats are decreasing and have declined dramatically. The historic range of the species has been reduced by about 96 percent; reduced from 25 historic locations to 7 (8 identified by others) currently occupied sites. Habitat loss and fragmentation is continuing (NatureServe 2015). The San Bernardino kangaroo rat occurs at eight localities. Four localities (Santa Ana River, Lytle and Cajon washes, and Etiwanda Creek) contain moderately large populations and four (Badlands, Bautista Canyon, San Timoteo Creek, and San Jacinto River near Hemet) have small populations in fragmented and isolated habitat patches (Bolster 1998). Population size is highly variable, and is correlated with percent vegetation cover and vegetation type as well as variations in substrate (percent sand, gravel, and cobble) (USFWS 2009). Population density studies have documented substantial annual variation. There appear to be several reasons for these greatly disparate values, including: 1) a low population density observed in an area at one point in time does not mean that the area is occupied at the same low density in any other month, season, or year; 2) a low population density is not an indicator of low habitat quality or low overall value of the land for the conservation of the species; 3) an abundance of San Bernardino kangaroo rats can decrease rapidly; and 4) one or more factors (e.g., food availability, fecundity, disease, predation, genetics, and environment) are strongly influencing the species' population dynamics in one or more areas (67 FR 19812).

**Threats and Stressors**

**Stressor:** Habitat fragmentation and loss

**Exposure:** Flood control structures and operations, aggregate mining, agricultural activities, urban and industrial development, and off-highway vehicles.

**Response:** Habitat loss, degradation, and fragmentation, and vulnerability to catastrophic events such as flooding.

**Consequence:** Mortality, population decline.

**Narrative:** Development in the floodplain habitat continues to increase with population growth. As a result of the combined pressures of agriculture, flood control structures, aggregate mining,

and urbanization, it is anticipated that there will be new construction of roads, expansion of existing roads and bridges, and additional construction of pipelines, reservoirs, and pumping stations, primarily in the Santa Ana and San Jacinto River floodplains. This will result in an overall reduction in the amount of available habitat to the San Bernardino kangaroo rat. Because the limited amount of remaining habitat is in the floodplain and in close proximity to active channels, with little upland refugia, the subspecies remains vulnerable to catastrophic events such as flooding. Additionally, the highly fragmented landscape, reduced habitat patch size, and isolation will exacerbate the effect of habitat loss on a species' persistence (USFWS 2009).

**Stressor:** Inadequacy of existing regulatory mechanisms.

**Exposure:** The majority of known populations of the San Bernardino kangaroo rat occur on nonfederal land.

**Response:** Inadequate state and federal protections on private land.

**Consequence:** Mortality, population decline.

**Narrative:** Although state and other federal regulations may provide some discretionary conservation benefit to the San Bernardino kangaroo rat, the Federal Endangered Species Act (ESA) is the primary regulatory mechanism mandating San Bernardino kangaroo rat conservation and ensuring that the San Bernardino kangaroo rat is addressed during planning efforts that may impact the subspecies or its habitat. Section 10 of the ESA is the primary federal process for addressing both the economic development needs and the conservation needs of the species on private lands in Riverside County. Management and coordination with federal, state, and local government agencies and mining operations will be needed to protect San Bernardino kangaroo rat from further habitat fragmentation and loss (USFWS 2009).

**Stressor:** Small population size

**Exposure:** Remaining populations of the San Bernardino kangaroo rat continue to be at risk due to their small size.

**Response:** Susceptible to inbreeding, loss of genetic variation, and stochastic events.

**Consequence:** Potential extirpation of populations.

**Narrative:** Small populations have a higher probability of extinction than larger populations, because their low abundance renders them susceptible to inbreeding; the loss of genetic variation; demographic problems like skewed variability in age and sex ratios; and stochastic events such as floods, droughts, or disease epidemics (USFWS 2009).

**Stressor:** Predation

**Exposure:** Fragmentation of habitat and urbanization.

**Response:** Could promote higher levels of predation by urban-associated animals, particularly domestic cats.

**Consequence:** Could threaten remnant populations of San Bernardino kangaroo rat.

**Narrative:** Fragmentation of habitat and urbanization likely promotes higher levels of predation by urban-associated animals, specifically domestic cats. As the interface between occupied habitat to developed areas is increased, predation could become a larger threat to the San Bernardino kangaroo rat (USFWS 2009).

**Stressor:** Isolation

**Exposure:** Habitat fragmentation of the San Bernardino kangaroo rat's historical range.

**Response:** Remaining blocks of isolated habitat may now function independently of each other.

**Consequence:** Higher susceptibility to extirpation by accidental or natural catastrophes.

**Narrative:** Altered fluvial processes, urbanization, and land conversion have fragmented the historical range of the San Bernardino kangaroo rat in such a way that remaining extant populations are isolated from and function independently of one another. Therefore, the extirpation of remnant populations during local catastrophes will continue to become more probable with increased habitat fragmentation and isolation (USFWS 2009).

### ***Recovery***

#### **Reclassification Criteria:**

Reclassification criteria for this species have not been developed.

#### **Delisting Criteria:**

Delisting criteria for this species have not been developed.

#### **Recovery Actions:**

- Need to develop an approved Recovery Plan containing recovery actions.
- Work with partners and identify opportunities through USFWS' Partners for Fish and Wildlife Program to seek habitat management, restoration, and enhancement opportunities for San Bernardino kangaroo rat. A goal of habitat restoration projects and management actions should be to determine more specific habitat requirements for this species (USFWS 2009).
- Work with partners to protect additional San Bernardino kangaroo rat habitat, including upland refugia habitat to support San Bernardino kangaroo rat during flood events. Occupied floodplains and adjacent upland habitat should be conserved to ensure protection of populations large enough to remain viable in the long term (USFWS 2009).
- Monitor San Bernardino kangaroo rat populations throughout known and potentially occupied sites to track their recovery. Systematic sampling efforts for a minimum of 5 years at each occupied site would provide basic data to estimate occupancy and relative abundance through time. Standard survey protocols for San Bernardino kangaroo rat population abundance or density trends need to be developed (USFWS 2009).
- Develop a recovery plan for the San Bernardino kangaroo rat (USFWS 2009).

#### ***Conservation Measures and Best Management Practices:***

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#### ***Additional Threshold Information:***

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## SPECIES ACCOUNT: *Dipodomys nitratoides exilis* (Fresno kangaroo rat)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; January 30, 1985 (50 FR 4222).

#### **Physical Description**

The San Joaquin kangaroo rat is similar in general appearance to the other 20 species of kangaroo rats, but is smaller and differs substantially in several ways. The Fresno kangaroo rat is adapted for survival in an arid environment. Its adaptations for bipedal locomotion include elongated hind limbs, a long tufted tail for balance, a shortened neck, and a large head compared to typical rodents. The skull is flattened from top to bottom, with enlarged auditory bullae (bony capsules containing the middle and inner ears). Other characteristics include large eyes placed near the top of the head and small, rounded ears. Forelimbs are comparatively short, with stout claws that facilitate digging burrows. Its total length averages about 231 millimeters (mm) (9.09 inches [in.]) for males and 225 mm (8.86 in.) for females. The hind foot usually is less than 36 mm (1.42 in.) in length. The fur is dark yellowish-buff dorsally and white ventrally. A white stripe extends across the hips, continuing for the length of the prominently tufted tail. The base of the tail is circumscribed by white. Dorsal and ventral sides of the tail are blackish. Dark whisker patches on each side of the nose are connected by a black band of fur (USFWS 1998).

#### **Taxonomy**

The San Joaquin kangaroo rat can be distinguished from other kangaroo rats within its geographic range by the presence of four toes on the hind foot; the other species found in the same area have five toes. The Fresno kangaroo rat is the smallest of three subspecies of the San Joaquin kangaroo rat (*Dipodomys nitratoides*), and is distinguished by its smaller average measurements (USFWS 1998).

#### **Historical Range**

The known historical geographic range of the Fresno kangaroo rat encompassed an area of grassland and chenopod scrub communities on the San Joaquin Valley floor, from about the Merced River, Merced County, on the north; to the northern edge of the marshes surrounding Tulare Lake, Kings County, on the south; and extending from the edge of the Valley floor near Livingston, Madera, Fresno, and Selma, westward to the wetlands of Fresno Slough and the San Joaquin River (USFWS 1998).

#### **Current Range**

There are no known populations within the circumscribed historical geographic range in Merced, Madera, and Fresno counties. A single male Fresno kangaroo rat was captured twice in autumn 1992 on the Alkali Sink Ecological Reserve, west of Fresno. Trapping at other sites in Merced, Madera, and Fresno counties between 1988 and 1995 has failed to locate other extant populations. In Kings County, two populations of San Joaquin kangaroo rats (*Dipodomys nitratoides*) were found on about 371 acres (ac.) in 1994 and 1995. One site, Lemoore Naval Air Station, is 97 ac. Whether these populations belong to the Fresno or Tipton subspecies is uncertain; historically, their ranges were contiguous (USFWS 1998).

#### **Distinct Population Segments Defined**

No

**Critical Habitat Designated**

Yes; 1/30/1985.

**Legal Description**

On January 30, 1985, the Service determined endangered status and critical habitat for the Fresno kangaroo rat.

**Critical Habitat Designation**

The critical habitat for the kangaroo rat comprises about 857 acres in western Fresno County, California. It is located generally to the south of the San Joaquin River, to the west of the town of Kerman, to the north of the Fresno Slough Bypass, and to the east of the Fresno Slough. Of this land, about 565 acres compose the State of California's Alkali Sink Ecological Reserve or are scheduled for addition to the Reserve, about 20 acres are part of the State-owned Mendota Wildlife Management Area, and the remainder is privately owned.

California. An area of land, water, and airspace in Fresno County, with the following components (Mt. Diablo Base Meridian): T 14 SR 15 E, E1/2 NW1/4 and NE1/4 Sec. 11, that part of W1/4 Sec. 12 north of the Southern Pacific Railroad, E1/2 Sec. 12; T14S R16E, that part of Sec. 7 south of the Southern Pacific Railroad.

**Primary Constituent Elements/Physical or Biological Features**

Within the critical habitat area, the major constituent elements that are known to require special management considerations or protection are:

- (1) the hummocks and substrate that provide sites for burrow construction, and
- (2) the natural alkali sink-open grassland vegetation that provides food and escape cover.

**Special Management Considerations or Protections**

Conversion of native vegetation for agricultural use destroys suitable habitat. Moderate to heavy livestock grazing adversely modifies habitat, so that the number of Fresno kangaroo rats that can be supported is severely reduced. Any other activities that disturb the native vegetation and ecosystem would probably also adversely affect the kangaroo rat. Conversely, the same kinds of actions could be affected by the protection of the critical habitat of the kangaroo rat, if they are likely to adversely modify such habitat, and if they are authorized, funded, or carried out by a Federal agency.

***Life History*****Feeding Narrative**

Adult: Fresno kangaroo rats collect and carry seeds in fur-lined cheek pouches. They feed primarily on seeds from native and nonnative forbs and grasses. Green, herbaceous vegetation and insects make up a small percentage of their diet (USFWS 1998; USFWS 2010). They are active year-round and nocturnal. The extent of competition is currently unknown; however, there is an ongoing study that will provide additional information on the extent of the threat posed by the Heermann's kangaroo rat (*Dipodomys heermanni*) on Fresno kangaroo rats

(USFWS 2010). Fresno kangaroo rats do not cache seeds in their burrows to the same extent as other kangaroo rats, because the soil where they live is damp much of the year. Therefore, they are obligated to forage on the surface year-round to a greater extent than kangaroo rats that cache seeds (USFWS 1998).

**Reproduction Narrative**

Adult: Fresno kangaroo rats have a lifespan of between 3 and 5 years. They reach sexual maturity around 82 days, and breeding is probably initiated in winter after the onset of the rainy season. Their gestation period is 32 days in captivity, and they can give birth two or more times per year. The reproductive potential of Fresno kangaroo rats is relatively low compared to most rodents; the average clutch size of this species is two. Limiting factors on populations are unknown, but the availability of suitable sites for burrows, free from winter flooding, is probably a major factor. Babies remain in the nest for up to 6 weeks, and are weaned at 21 to 24 days (USFWS 1998).

**Geographic or Habitat Restraints or Barriers**

Adult: Conversion of natural habitat to agriculture completely eliminates the use of the habitat by the Fresno kangaroo rat (50 FR 4222).

**Spatial Arrangements of the Population**

Adult: Clumped

**Environmental Specificity**

Adult: Narrow/specialist.

**Tolerance Ranges/Thresholds**

Adult: Low

**Site Fidelity**

Adult: High

**Dependency on Other Individuals or Species for Habitat**

Adult: No

**Habitat Narrative**

Adult: Fresno kangaroo rats occupy sands and saline sandy soils in chenopod scrub, and annual grassland communities on the Central Valley floor. They require hummocks and substrate that provide sites for burrow construction, and natural alkali sink-open grassland vegetation that provides food and escape cover. Associated plant species include seepweed (*Suaeda* sp.), iodine bush (*Allenrolfea* sp.), saltbushes (*Atriplex* sp.), peppergrass (*Lepidium* sp.), filaree (*Erodium* sp.), wild oats (*Avena* sp.), and mouse-tail fescue (*Festuca myuros*) (USFWS 1998). Conversion of natural habitat to agriculture completely eliminates the use of the habitat by the Fresno kangaroo rat (50 FR 4222).

**Dispersal/Migration****Motility/Mobility**

Adult: Low

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Low

**Immigration/Emigration**

Adult: No

**Dependency on Other Individuals or Species for Dispersal**

Adult: No

**Dispersal/Migration Narrative**

Adult: There are approximately 32,234 hectares (ha) (79,651 ac.) remaining in natural habitat that would be suitable for the Fresno kangaroo rat. However, these parcels are fragmented and separated by large expanses of unsuitable habitat. Due to the degree of habitat fragmentation, dispersal would not be possible (USFWS 2010).

***Population Information and Trends*****Population Trends:**

Decreasing

**Species Trends:**

Decreasing; the short-term trend is an estimated decline of 10 to 30 percent (NatureServe 2015).

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Population Growth Rate:**

Unknown (USFWS 2010)

**Number of Populations:**

Unknown (USFWS 2010). Between one and five populations are assumed to exist (NatureServe 2015).

**Population Size:**

Unknown (USFWS 2010). Population size is estimated between 1 and 1,000 individuals (NatureServe 2015).

**Resistance to Disease:**

Moderate

**Adaptability:**

Low

**Additional Population-level Information:**

By the time of the publication of the Recovery Plan, there were no known populations of Fresno kangaroo rats. In 1985, there were trappings of kangaroo rats reported in Fresno at the Alkali Sink Ecological Reserve and on adjacent public lands. In 1992, only one male was trapped at the Alkali Sink Ecological Reserve. Since that time, biologists have surveyed areas of suitable habitat in Fresno, Madera, and Merced counties, but have not been able to capture any Fresno kangaroo rats (USFWS 2010).

**Population Narrative:**

Since the publication of the Recovery Plan in 1998, there have been no Fresno kangaroo rats captured. The absence of this species during the surveys is of great concern. In 1985, there were reported trappings in Fresno of kangaroo rats at the Alkali Sink Ecological Reserve and on adjacent public lands. In 1992, only one male was trapped at the Alkali Sink Ecological Reserve. Since that time, biologists have surveyed areas of suitable habitat in Fresno, Madera, and Merced counties, but have not been able to capture any Fresno kangaroo rats. There are large tracts of natural land within the range of the species that have not been surveyed, including isolated and fragmented habitat in private ownership. Based on this lack of information, there is little known about the presence of existing populations (USFWS 2010).

***Threats and Stressors***

**Stressor:** Habitat destruction

**Exposure:** Conversion of 80 percent of habitat to crop production and urbanization (USFWS 2010).

**Response:** Fresno kangaroo rats will not occupy converted habitat.

**Consequence:** The range of the species has been greatly reduced and fragmented.

**Narrative:** Habitat conversion continues to be the primary threat to the Fresno kangaroo rat. Modification of habitat for agriculture and urbanization has reduced the available habitat by approximately 80 percent. Additionally, habitat fragmentation likely prevents dispersal of any existing populations (USFWS 2010).

**Stressor:** Flooding

**Exposure:** Currently, protected parcels of Fresno kangaroo rat habitat are within proximity of the San Joaquin River.

**Response:** Flooding of these areas is rare, but possible.

**Consequence:** Flooding could be catastrophic to Fresno kangaroo rats.

**Narrative:** Both of the parcels of land that are protected for the Fresno kangaroo rat are within proximity of the San Joaquin River. A previous levee break and the subsequent flooding of the Alkali Sink Preserve may have caused or contributed to the extirpation of the Fresno kangaroo rat from the location. Although flooding due to natural events or failure of man-made levees are rare events, they could be catastrophic to Fresno kangaroo rats (USFWS 2010).

**Stressor:** Illegal rodenticide

**Exposure:** Several vertebrate control agents have been identified as detrimental to the existence of kangaroo rats.

**Response:** The degree of threat to the Fresno kangaroo rat is unknown.

**Consequence:** Exposure to these rodenticides could be detrimental to potentially existing populations.

**Narrative:** Currently, there is a lack of information regarding the locations of potentially existing populations of Fresno kangaroo rats. The U.S. Fish and Wildlife Service has identified several vertebrate control agents as detrimental to the existence of kangaroo rats. Exposure of remaining populations to these contaminants could be detrimental (USFWS 2010).

**Stressor:** Competition with Heermann's kangaroo rats (*Dipodomys heermanni*)

**Exposure:** There is an ongoing study to evaluate the impacts of Heermann's kangaroo rats on other kangaroo rat populations.

**Response:** Unknown

**Consequence:** It is possible that the presence of Heermann's kangaroo rat populations could exert competition on populations of the Fresno kangaroo rat.

**Narrative:** There is an ongoing study to evaluate the potentially negative impacts of Heermann's kangaroo rats on existing kangaroo rat populations (USFWS 2010).

### ***Recovery***

**Reclassification Criteria:**

Establishment of one hundred percent of occupied habitat on public or conservation lands at three or more distinct sites, with each no less than about 950 ac. of usable habitat.

Management plans need to be approved and implemented for all inhabited areas identified as important to continued survival.

Population densities in three or more populations do not fall below two kangaroo rats per ha (one per ac.), and have a mean density of ten or more per ha (4 per ac.) during one precipitation cycle.

**Delisting Criteria:**

In addition to the criteria for downlisting, all of the following conditions have been met:

One additional site with about 2,500 ac. or more of occupied habitat, and

with a total of no less than 5,350 ac. of occupied habitat.

None

None

None

None

None

None

None

None

None  
None  
None  
None  
None  
None

**Recovery Actions:**

- Comprehensively survey and trap all remaining habitat within the range of the Fresno kangaroo rat, and locate any remaining populations or population remnants (USFWS 2010).
- Protect additional parcels of alkali sink scrub and grasslands within the Fresno kangaroo rat range (USFWS 1998); particularly any parcels on which Fresno kangaroo rats are discovered (USFWS, 2010).
- Consistently manage protected alkali sink scrub habitat for Fresno kangaroo rats (USFWS 2010).
- Recognizing that genetic and taxonomic studies and habitat surveys already are in progress, critical recovery actions needed now are:
- Complete the studies on relationships and taxonomic identity of isolated populations of San Joaquin kangaroo rats (USFWS 1998).
- Intensify and continue efforts to locate populations of Fresno kangaroo rats within the historical range of the species. If a population is found, captive breeding should be considered as a recovery option, depending on the size of the population (USFWS 1998).
- Continue and increase habitat management studies (USFWS 1998).
- Restore additional habitat for *D. nitratoides* at Lemoore Naval Air Station (USFWS 1998).
- Protect natural land between the Alkali Sink Ecological Reserve and the San Joaquin River to the north (Sandy Mush Road/South Grasslands Area) (USFWS 1998).
- Begin discussion and planning for conservation of natural lands in western Madera County; acquire title or easement to appropriate parcels from willing sellers (USFWS 1998).
- Recovery actions that will also be needed, after critical actions are implemented or completed, are:
- Protect additional habitat for Fresno kangaroo rats in Kings County, where populations of the species are discovered. Habitat should be in blocks of at least 384 ha (950 ac.), preferably larger, with one block no less than 1,012 ha (2,500 ac.) (USFWS 1998).
- Work with landowners in western Madera County to determine the presence or absence of the species there. If a population is found, assess translocating populations to public lands in Fresno County (USFWS 1998).
- Restore habitat for Fresno kangaroo rats on the Alkali Sink and Kerman Ecological Reserves. Restoration should include manipulation of the plant community to favor Fresno kangaroo rats over Heermann's kangaroo rats (USFWS 1998).
- Reintroduce Fresno kangaroo rats to restored and unoccupied habitats on ecological reserves and newly-protected parcels (USFWS 1998).
- Monitor all populations and their supporting biotic communities annually for a 10-year period, then at 3-year intervals until recovery is achieved (USFWS 1998).
- Manage habitat for Fresno kangaroo rats as needed (USFWS 1998).

***Conservation Measures and Best Management Practices:***

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***Additional Threshold Information:***

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## SPECIES ACCOUNT: *Dipodomys nitratoides nitratoides* (Tipton kangaroo rat)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; July 8, 1988 (53 FR 25608).

### **Physical Description**

The Tipton kangaroo rat (*Dipodomys nitratoides nitratoides*) is similar in general appearance to the other 20 species of kangaroo rats, but is smaller and differs substantially from all other species in several ways. Like all kangaroo rats, the Tipton kangaroo rat is adapted for survival in an arid environment. Adaptations for bipedal locomotion include elongated hind limbs, a long, tufted tail for balance, a shortened neck, and a large head in comparison to typical rodents. The skull is flattened from top to bottom, with enlarged auditory bullae (bony capsules containing the middle and inner ears). Other characteristics include large eyes placed near the top of the head and small, rounded ears. Forelimbs are comparatively short, with stout claws that facilitate digging burrows. On average, adult Tipton kangaroo rats weigh about 35 to 38 grams (1.23 to 1.34 ounces). The fur is dark yellowish-buff dorsally and white ventrally. A white stripe extends across the hips, continuing for the length of the prominently tufted tail. The base of the tail is circumscribed by white. Dorsal and ventral sides of the tail are blackish. Dark whisker patches on each side of the nose are connected by a black band (USFWS 1998; USFWS 2010).

### **Taxonomy**

The Tipton kangaroo rat is one of three subspecies of the San Joaquin kangaroo rat (*Dipodomys nitratoides*). The type specimen of the Tipton kangaroo rat was collected from Tipton, Tulare County, California, in 1893. The Fresno (*Dipodomys nitratoides exilis*) and Tipton kangaroo rats are similar in overall structure, and occupy contiguous geographic ranges on the floor of the Tulare Basin and southeastern half of the San Joaquin Basin in the San Joaquin Valley. It can be distinguished from other species within its geographic range by the presence of four toes on the hind foot (other subspecies have five toes). It is larger than the Fresno kangaroo rat but smaller than the short-nosed kangaroo rat (*Dipodomys nitratoides brevinasus*). The Tipton kangaroo rat can be distinguished from the Fresno kangaroo rat by its larger average measurements: average head and body length of 100 to 110 millimeters (mm) (3.9 to 4.3 inches [in.]); total length of 235 mm (9.25 in.) for males and 221 mm (8.7 in.) for females; length of hind foot of 34.7 mm (1.37 in.) for males and 33.6 mm (1.32 in.) for females; and mean inflation of the auditory bullae of 22.1 mm (0.87 in.) for males and 21.8 (0.86 in.) for females. The third subspecies, the short-nosed kangaroo rat, is found in the foothills and basins along the western side of the San Joaquin Valley south of Los Baños, Merced County, and western portions of the Tulare basin, the upper Cuyama Valley, and Carrizo Plain (USFWS 1998; USFWS 2010).

### **Historical Range**

The historical geographic range was more than 687,650 hectares (ha) (about 1.7 million acres [ac.]). Distribution was limited to arid-land communities occupying the valley floor of the Tulare Basin, extending from approximately the southern margins of Tulare Lake on the north, then eastward and southward approximately along the eastern edge of the valley floor in Tulare and Kern counties. The southern and western extent of their range was the foothills of the Tehachapi Mountains (southern boundary) and the marshes and open water of Kern and Buena

Vista lakes and the sloughs and channels of the Kern River alluvial fan (western boundary). Farther north, the western boundary was approximately along the Buena Vista slough of the Kern River channel into Goose Lake. The approximate line on the northwest is marked by the cities of Lost Hills, Kern County; Kettleman City, Kings County; and Westhaven, Fresno County. Prior to development of water-diversion and irrigation systems over the past several decades, this area bounded three large lakes—Tulare, Kern, and Buena Vista—together with marshlands that were unsuitable habitat for kangaroo rats. By 1985, the inhabited area had been reduced, primarily by cultivation and urbanization, to only about 4 percent of the historical acreage (USFWS 1998).

**Current Range**

The current range of the Tipton kangaroo rat is limited to scattered, isolated areas in Kings, Tulare, and Kern counties. By 1985, the inhabited area had been reduced, primarily by cultivation and urbanization, to only about 3.7 percent of the historical acreage. Based on California Natural Diversity Database occurrence records, the Tipton kangaroo rat is known to approximately 100 locations, but the surface area of these sites is not known. Given the absence of range-wide surveys and the dynamic nature of Tipton kangaroo rat populations, the current geographic distribution of the subspecies is not clearly defined (USFWS 2010).

**Distinct Population Segments Defined**

No

**Critical Habitat Designated**

No;

***Life History*****Feeding Narrative**

Adult: Tipton kangaroo rats collect and carry seeds in fur-lined cheek pouches. They eat mostly seeds from native and nonnative forbs and grasses, with small amounts of green, herbaceous vegetation, and supplement their diet with insects when available. In fall and winter, after the wet season commences, sprouts of seeds and tender new growth of grasses and forbs may be essential items in the diet of Tipton kangaroo rats. Green developing seed heads may be important in the spring months. Seeds, and perhaps insects, are the most important items in the diet in late spring, summer, and fall (USFWS 1998). Seeds of woody shrubs, especially saltbushes, are diligently sought out by Tipton kangaroo rats, including seeds of the woody and semiwoody shrubs, iodine bush (*Allenrolfea occidentalis*) and seepweed (*Sueda moquinii*). Other known foods include seeds of annual and perennial grasses, particularly wild oats (*Avena* spp.), brome grasses (red and ripgut [*Bromus diandrus*] brome, and soft chess [*B. hordeaceus*]), wild barley (*Hordeum* sp.), mouse-tail fescue (*Vulpia myuros*), alkali sacaton (*Sporobolus airoides*), and saltgrass (*Distichlis spicata*); and seeds of annual forbs such as filaree (*Erodium* spp.), peppergrass (*Lepidium* spp.), common spikeweed (*Hemizonia pungens*), and shepherds purse (*Capsella bursa-pastoris*) (USFWS 1998). Some food is later cached in small holes that are dug in the sides of burrows. Kangaroo rats have developed the ability to survive in the wild indefinitely without drinking water (USFWS 2010). They are nocturnal and active year round (USFWS 1998; NatureServe 2015). The presence of Heermann's kangaroo rat (*D. heermanni*) is a factor affecting the continued existence of the subspecies. The competitively dominant Heermann's kangaroo rat is larger than the Tipton kangaroo rat, more general in habitat requirements, and

more successful in maintaining populations in a fragmented landscape. Although the two taxa rarely occupy the same area, competition could become an important factor if the Tipton kangaroo rat is translocated to an area unknowingly occupied by Heermann's kangaroo rat, and may prevent establishment or survival of Tipton kangaroo rats in areas of range overlap (USFWS 2010).

### **Reproduction Narrative**

Adult: Little specific information has been published on reproduction of Tipton kangaroo rats. Generally, this aspect of their biology is extremely similar to that of the Fresno kangaroo rat. Nothing is known about pair bonds in wild populations, but there probably are no lasting male-female pair bonds formed. Females may breed with more than one male during a breeding cycle, though typically a single male attains dominance for mating purposes with one or more females in his territory, as is true of closely related kangaroo rat species (USFWS 1998). Reproduction occurs from December through September and peaks in late March and early April (USFWS 1998; Zeiner et al. 1999). For the Fresno kangaroo rat, gestation is approximately 32 days in captivity, and the age to first reproduction is approximately 82 days (USFWS 1998). Most females appear to have only a single litter, though some adult females have two or more, and females born early in the year also may breed (USFWS 1998). Litter size averages two to three young (range from one to three), and the parental investment is high; they are weaned at 21 to 24 days, and the young do not leave natal nest to forage for themselves for about 6 weeks. The lifespan of the Tipton kangaroo rat is approximately 3 to 5 years (USFWS 1998).

### **Geographic or Habitat Restraints or Barriers**

Adult: Physical barriers; populations are frequently separated by roads and canals that cannot be crossed by this subspecies (USFWS 2010).

### **Spatial Arrangements of the Population**

Adult: Clumped; occur in a mosaic pattern of small and isolated patches that are dynamic over time (USFWS 2010).

### **Environmental Specificity**

Adult: Narrow/specialist

### **Tolerance Ranges/Thresholds**

Adult: Low

### **Site Fidelity**

Adult: High

### **Dependency on Other Individuals or Species for Habitat**

Adult: Tipton kangaroo rats live in ground burrows. Most burrows probably are dug by the occupant or a predecessor of the same species. Except for young associated with females, each burrow system is typically occupied by a single individual (USFWS 1998).

### **Habitat Narrative**

Adult: Tipton kangaroo rats are limited to arid-land communities occupying the valley floor of the Tulare Basin in level or nearly level terrain. They occupy alluvial fan and floodplain soils ranging from fine sands to clay-sized particles with high salinity. Historically, populations

apparently were most numerous and persistent in Relictual Interior Dune Grassland and Sierra-Tehachapi Saltbush Scrub communities (e.g., valley sink scrub and valley saltbush scrub). Today, much of the occupied remnants of their range have one or more species of sparse-to-moderate woody shrub cover and a ground cover of mostly introduced and native annual grasses and forbs. Woody shrubs commonly associated with Tipton kangaroo rats are spiny and common saltbushes, arrowscale (*Atriplex phyllostegia*), quailbush (*Atriplex lentiformis*), iodine bush (*Allenrolfea occidentalis*), pale-leaf goldenbush (*Isocoma acradenia* var. *bracteosa*), and honey mesquite (*Prosopis glandulosa* var. *torreyana*). A conspicuous semiwoody species is seepweed (*Suaeda* sp.) (USFWS 1998; USFWS 2010). Tipton kangaroo rats live in ground burrows. Most burrows probably are dug by the occupant or a predecessor of the same species. Except for young associated with females, each burrow system is typically occupied by a single individual (USFWS 1998). Tipton kangaroo rats' burrow systems are located in open areas; only in areas of dense shrub cover are burrows usually located beneath shrubs. Burrows are commonly located in slightly elevated mounds, the berms of roads (where placed above ground level), canal embankments, railroad beds, and bases of shrubs and fences where windblown soils accumulate above the level of surrounding terrain. Soft soils such as fine sands and sandy loams, and powdery soils of finer texture and of higher salinity, are generally associated with greater densities of Tipton kangaroo rats than are less saline and alkaline sandy-loam, and clay-loam soils of portions of the eastern margins of their geographic range, which support terrace grasslands. This may relate to how crumbly the soils are, the type of plant communities they support, or both. Terrain not subject to flooding is important for permanent occupancy by Tipton kangaroo rats (USFWS 1998). Populations are frequently separated by physical barriers such as roads and canals that cannot be crossed by this subspecies. They occur in a mosaic pattern of small and isolated patches that are dynamic over time (USFWS 2010). Winter rains and runoff from the surrounding mountain ranges (Sierra Nevada to the east, Tehachapi Mountains to the south, and Temblor Range to the west) flood much of these low-lying communities occupied by Tipton kangaroo rats. During portions of winter and spring, areas with standing water (vernal pools) become alkaline playas when the water has evaporated, allowing Tipton kangaroo rats to recolonize these areas even though alkaline water lies close to the surface of the soil year-round. During flooding, individuals are presumably either drowned or captured by predators after being forced from their burrows, or escape to higher ground (USFWS 1998). spring (vernal pools) become alkaline playas when the water has evaporated allowing Tipton kangaroo rats to recolonize these areas even though alkaline water lies close to the surface of the soil, year around. Presumably during flooding, individuals are either drowned or captured by predators after being forced from their burrows, or escape to higher ground (USFWS 1998).

***Dispersal/Migration*****Motility/Mobility**

Adult: Low

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Low; reported average mean maximum distance moved of  $15.9 \pm 3.1$  meters (m) ( $52.2 \pm 10.2$  feet [ft.]) (USFWS 2010).

**Immigration/Emigration**

Adult: No

**Dependency on Other Individuals or Species for Dispersal**

Adult: No

**Dispersal/Migration Narrative**

Adult: The Tipton kangaroo rat has relatively low motility and is nonmigratory. Dispersal is low, with a reported average mean maximum distance moved of  $15.9 \pm 3.1$  m ( $52.2 \pm 10.2$  ft.) (USFWS 2010). Currently, the available natural habitat is fragmented. Areas that formerly supported Tipton kangaroo rat populations have been developed or used for agricultural purposes, and no longer provide suitable habitat for this subspecies (USFWS 2010). An aerial survey conducted in late 1983, together with selected ground inspections and other sources of information, provided an estimate of 44,562 ha (110,031 ac.) of undeveloped land out of a total of 1,035,296 ha (2,556,288 ac.) on the floor of the Tulare Basin (USFWS 1998). Estimated densities at Pixley National Wildlife Refuge ranged from  $3.0 \pm 0.9$  to  $3.8 \pm 1.6$  per ha ( $7.4 \pm 2.2$  to  $10.4 \pm 3.9$  per ac.) (Zeiner et al. 1999). Remaining natural lands represent the least desirable for development in the basin. Most of the remaining habitat of Tipton kangaroo rats is in areas that are already flooded periodically. Several parcels with extant natural lands in the 1970s now have private evaporation ponds into which salt-laden drain waters are being diverted. Habitat restoration would be required to promote dispersal of potential populations (USFWS 1998; USFWS 2010).

**Additional Life History Information**

Adult: Estimated densities at Pixley National Wildlife Refuge ranged from  $3.0 \pm 0.9$  to  $3.8 \pm 1.6$  per ha ( $7.4 \pm 2.2$  to  $10.4 \pm 3.9$  per ac.) (Zeiner et al. 1999).

***Population Information and Trends*****Population Trends:**

Declining; former range of 1.7 million ac. was reduced to about 63,400 ac. by July 1985 (NatureServe 2015).

**Species Trends:**

Declining; short-term trend decline of 10 to 30 percent (NatureServe 2015).

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Number of Populations:**

Based on California Natural Diversity Database (CNDDDB) occurrence records, the Tipton kangaroo rat occupies approximately 100 isolated locations (USFWS 2010).

**Population Size:**

Information about population trends is extremely limited, and only a few sites have been surveyed. Current information, based on surveys at about 10 sites, suggest that Tipton kangaroo rat abundance is low throughout the known range of the subspecies, and that populations continue to decline. Surveys at Coles Levee Ecosystem Preserve, for example, suggest that the local population is well below 1,000 individuals. Naval Air Station Lemoore is a much smaller area; it consists of 40.5 ha (100 ac.) and has even fewer Tipton kangaroo rats (estimates range from 0 to 300 individuals). These two sites are relatively secure and well-known with regard to the Tipton kangaroo rat. At the remaining sites, surveys suggest that several local Tipton kangaroo rat populations are well below 100 individuals per site, while others may no longer be extant (USFWS 2010). The total population is estimated at about 190,200 individuals.

**Resistance to Disease:**

Tipton kangaroo rats may be susceptible to disease as a consequence of exceptionally wet winters (USFWS 2010).

**Adaptability:**

Low

**Additional Population-level Information:**

Historical records suggest that Tipton kangaroo rat populations usually were small and subject to great variation in total population size from year to year (USFWS 2010). Documented density ranges from less than 1 per ha to about 88 per ha (217 per ac.) (NatureServe 2015).

**Population Narrative:**

The current population trend for the Tipton kangaroo rat is one of decline; the former range of 1.7 million ac. was reduced to about 63,400 ac. by July 1985. The total population size is estimated at about 190,200 individuals. Most extant populations may not be large enough to be viable indefinitely (NatureServe 2015). Based on CNDDDB occurrence records, the Tipton kangaroo rat occupies approximately 100 isolated locations, but the surface area of these sites is not known. Information about population trends is extremely limited, and only a few sites have been surveyed. Historical records suggest that Tipton kangaroo rat populations usually were small and subject to great variation in total population size from year to year. Current occurrences are limited to scattered, isolated areas clustered west of Tipton, Pixley, and Earlimart, around Pixley National Wildlife Refuge, Atlenworth Ecological Reserve, and Allensworth State Historical Park, Tulare County; between the Kern National Wildlife Refuge, Delano, and in natural lands surrounding Lamont (southeast of Bakersfield), Kern County; at the Coles Levee Ecosystem Preserve; and other, scattered units to the south in Kern County (USFWS 1998). Based on surveys at about 10 sites, suggest that Tipton kangaroo rat abundance is low throughout the known range of the subspecies and populations continue to decline. Surveys at Coles Levee Ecosystem Preserve, for example, suggest that the local population is well below 1,000 individuals. Naval Air Station Lemoore is a much smaller area; it consists of 40.5 ha (100 ac.) and has even fewer Tipton kangaroo rats (estimates range from 0 to 300 individuals). These two sites are relatively secure and well-known with regard to the Tipton kangaroo rat. At the remaining sites, surveys suggest that several local Tipton kangaroo rat populations are well

below 100 individuals per site, while others may no longer be extant. Documented density ranges from less than 1 per ha to about 88 per ha (217 per ac.) (NatureServe 2015). Resistance to disease is poorly understood, but the Tipton kangaroo rat may be susceptible to disease as a consequence of exceptionally wet winters (USFWS 2010).

### ***Threats and Stressors***

**Stressor:** Habitat loss, fragmentation

**Exposure:** Agricultural development, fragmentation from infrastructure development, flooding, urbanization.

**Response:** Displaced to marginalized habitat.

**Consequence:** Decline in populations.

**Narrative:** Agricultural development; infrastructure development, including the construction of dams and canals; flooding; and urbanization have all contributed to the loss of habitat of the Tipton kangaroo rat. At the time of listing, habitat loss associated with agricultural development was identified as the main factor contributing to the decline of the Tipton kangaroo rat. Although industrial and agriculturally related developments were the primary factors leading to habitat loss until the 1990s, urbanization is now becoming more of a factor leading to the destruction, modification, or curtailment of the habitat or range of the Tipton kangaroo rat. The current range is highly restricted, and the distribution of small populations occur in highly isolated fragments. Approximately 96 percent of the original range is no longer suitable for the Tipton kangaroo rat, and populations continue to decline (USFWS 2010).

**Stressor:** Disease

**Exposure:** Wet winters may contribute to increase in disease.

**Response:** Disease may cause range-wide population reduction.

**Consequence:** Decline in populations.

**Narrative:** Disease may be a factor in Tipton kangaroo rat population declines during wet winters (USFWS 2010).

**Stressor:** Rodenticides

**Exposure:** Rodenticides are widely used throughout the species' range.

**Response:** Large expanses of the original range are no longer occupied.

**Consequence:** Population decline.

**Narrative:** Illegal application of rodenticides may be an increasingly important threat to the conservation status of the Tipton kangaroo rat (USFWS 2010).

**Stressor:** Climate change

**Exposure:** Climate models predict for California an overall warming, but vary in predictions for precipitation.

**Response:** Years of successive drought, annual variations in precipitation, and changes in the structure and composition of vegetative communities.

**Consequence:** Population declines.

**Narrative:** Populations of giant kangaroo rats have been monitored more closely than Tipton kangaroo rats, so we can look to these data as a surrogate. Years of successive drought lead to dramatic declines in the numbers of giant kangaroo rats as observed on the Elkhorn Plain in 1991 and in the Panoche-Ciervo area in the late 1980s. In addition, it has been suggested that years of above-normal precipitation can result in significant declines in giant kangaroo rat populations,

particularly in areas that are not grazed. Little information is available for Tipton kangaroo rats; substantial population declines for the Tipton kangaroo rat followed heaving rains during 1994 and 1995 (USFWS 2010).

**Stressor:** Competition with the Heermann's kangaroo rat (*D. heermanni*)

**Exposure:** Could become an important factor if the Tipton kangaroo rat is translocated to an area unknowingly occupied by Heermann's kangaroo rat.

**Response:**

**Consequence:** May affect the establishment or survival in areas of range overlap.

**Narrative:** Presence of Heermann's kangaroo rat (*D. heermanni*) is a factor affecting the continued existence of the subspecies. The competitively dominant Heermann's kangaroo rat is larger than the Tipton kangaroo rat, more general in habitat requirements, and more successful in maintaining populations in a fragmented landscape. Although the two taxa rarely occupy the same area, competition could become an important factor if the Tipton kangaroo rat is translocated to an area unknowingly occupied by Heermann's kangaroo rat, and may prevent establishment or survival of Tipton kangaroo rats in areas of range overlap (USFWS 2010).

**Stressor:** Inadequacy of regulatory mechanisms

**Exposure:** Agricultural development, pesticides, and removal of habitat.

**Response:**

**Consequence:** Survival of the species.

**Narrative:** At the time of listing, the U.S. Fish and Wildlife Service indicated that existing regulatory mechanisms did not afford the Tipton kangaroo rat adequate protection. The regulatory mechanisms of agencies that permitted or funded agricultural development, as well as those that regulated the application of pesticides, were specifically cited as being inadequate. More recently, however, the adequacy of regulatory mechanisms has improved for the Tipton kangaroo rat. The following regulatory mechanisms pertain to this subspecies, but were not discussed at the time the Tipton kangaroo rat was federally listed. Although current regulatory mechanisms, aside from the Endangered Species Act (ESA), are tasked with various levels of environmental protection that are beneficial to threatened and endangered species, the ESA is the primary law providing protection for the Tipton kangaroo rat. However, despite what may be considered adequate protection for the species, ongoing habitat impacts continue to affect species survival (USFWS 2010).

## **Recovery**

### **Reclassification Criteria:**

Reclassification to threatened status will be evaluated when the species is protected in specified recovery areas from incompatible uses, management plans have been approved and implemented for recovery areas that include survival of the species as an objective, and population monitoring indicates that the species is stable. Downlisting criteria include:

Recovery areas are secure and protected from incompatible uses. Protection of occupied habitat: A) Three or more distinct areas with 2,000 ha (4,940 ac.) or more of contiguous, occupied habitat, and B) 30 percent each or more of the minimum acreage in public or conservation ownership (USFWS 1998; USFWS 2010).

A management plan that includes the survival of the Tipton kangaroo rat as an objective must be approved and implemented for all protected areas identified as important to continued survival (USFWS 1998; USFWS 2010).

Population monitoring in specified recovery areas show that the populations must be stable or increasing through a precipitation cycle (USFWS 1998; USFWS 2010).

**Delisting Criteria:**

Delisting will be considered when, in addition to the criteria for downlisting, all of the following conditions have been met:

Recovery areas are secure and protected from incompatible uses. A total of 9,000 ha (22,230 ac.) or more of occupied habitat are in public or conservation ownership (USFWS 1998; USFWS 2010); and,

Population monitoring in specified recovery areas show that protected sites have a mean density of 10 kangaroo rats per ha (4 per ac.) during a complete precipitation cycle (USFWS 1998; USFWS 2010).

**Recovery Actions:**

- Expand, coordinate, and continue habitat management studies of Tipton kangaroo rats at sites representing the range of existing habitat conditions for the species (USFWS 1998).
- Initiate studies of competition between Tipton and Heermann's kangaroo rats, focusing primarily on how different habitat management prescriptions affect the population dynamics of the two species at sites of coexistence (USFWS 1998).
- Design and implement a range-wide population monitoring program that measures population and environmental fluctuations at sites representative of the range of natural land sizes and habitat conditions for the species (USFWS 1998).
- Inventory and assess existing natural land and drainage-problem parcels contiguous to and near existing protected natural lands, and develop a protection plan that ranks parcels that may be available according to their size and potential for supporting Tipton kangaroo rats, with the objective of connecting and expanding: a. Pixley National Wildlife Refuge and the scattered parcels of the Allensworth Ecological Reserve. b. Kern National Wildlife Refuge and the scattered parcels of the Semitropic Ridge conservation lands. c. Kern River alluvial fan area, including the Kern Fan Element, Cole's Levee Ecosystem Preserve, and other mitigation parcels. d. Additional lands which after inventory and assessment are identified as important to the two key elements of the recovery strategy for Tipton kangaroo rats (USFWS 1998).
- Develop and implement research on restoration of habitat for Tipton kangaroo rats, including cost-effective mechanisms to protect both natural and restored habitats from flooding (USFWS 1998).
- Restore habitat on retired agricultural lands as needed (USFWS 1998).
- Determine the current distribution and abundance of the Tipton kangaroo rat through surveys. These results should also be used to help inform decision-makers about the acquisition of appropriate sites where Tipton kangaroo rats occur but are unprotected, to suggest sites that could be acquired for restoration (see below), and to develop an adaptive

management program that will achieve the recovery of the Tipton kangaroo rat (USFWS 2010).

- Develop a monitoring plan for the subspecies to monitor abundance and population trends at a few selected sites. Monitoring should either continue for at least 10 to 20 years or until population dynamics are well understood (USFWS 2010).
- Based on the status survey, as well as group consensus of the several species experts about the subspecies' habitat needs, key priority tracts should be protected through acquisition or easement. High priority should be given to large unprotected sites that are currently occupied by the subspecies, as well as to large, formerly occupied sites that are unprotected but have a high restoration potential (USFWS 2010).
- Research and development of a genetic profile should be implemented to differentiate the several taxa within *Dipodomys* (USFWS 2010).

***Conservation Measures and Best Management Practices:***

- 

***Additional Threshold Information:***

- 
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## SPECIES ACCOUNT: *Dipodomys stephensi* (incl. *D. cascus*) (Stephens' kangaroo rat)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; 09/30/1988; Proposed reclassification to threatened

#### **Physical Description**

A medium-sized, long-tailed, nocturnal, hopping rodent. A chunky, large-headed, long-tailed, five-toed rat with a dusky cinnamon buff dorsum, white belly, external fur-lined cheek pouches, and large hind legs; white tail stripe is about half as wide as the dorsal tail stripe at mid-tail; total length 277-300 mm; tail 164-180 mm, about 145% of head and body length; hind foot 39-43 mm; length of ear from notch 13-16 mm; skull length 38.3-40.6 mm; width of maxillary arch 5.1-6 mm; adults averages about 67 g (Ingles 1965, Bleich 1977). LENGTH:32 WEIGHT: 75 (NatureServe, 2015)

#### **Current Range**

The range encompasses approximately 2,870 square kilometers in the San Jacinto Valley and adjacent areas of western Riverside County, southwestern San Bernardino County (at least formerly), and northwestern and north-central San Diego County, California (Bleich 1977, Williams et al. 1993), at elevations of 55-1,250 meters (USFWS 1997). As of the late 1980s, most extant populations were in western Riverside County, but the largest known population was on the Warner Ranch near Lake Henshaw, San Diego County (see Burke et al. 1991). See also USFWS (1987).

#### **Critical Habitat Designated**

No;

#### **Life History**

#### **Feeding Narrative**

Adult: Probably similar to *D. HEERMANNI* and *D. PANAMINTIMUS* which feed primarily on seeds but also eat insects and herbaceous vegetation in the spring. Sagebrush may provide much of the food (Biosystems Analysis 1989). More likely to forage in open, lit spaces than is sympatric *D. AGILIS* (Burke et al. 1991).; Food Habits: Granivore (Adult, Immature) (NatureServe, 2015)

#### **Reproduction Narrative**

Adult: Probably produces 1 litter per year, 2 litters/year under high rainfall conditions, perhaps none under drought conditions. Average litter size is about 2.5. In Riverside County, a peak in recruitment occurred in spring (McClenaghan and Taylor 1993). In some areas, young are born in late spring or early summer, and at least sometimes as late as July. In some years, young-of-the-year may reproduce. Life span appear to be relatively short, generally less than a few years.; Population density estimates vary with location and season, range from about 5 to 58 per ha; perhaps about 20-40/ha would be typical (USFWS 1987, Bleich 1977, McClenaghan and Taylor 1993). Population densities can vary more than 10-fold in response to rainfall patterns (Price and Endo 1989). In Riverside County, peak numbers occurred in late spring-early summer; populations declined from late summer through winter; minimum monthly survival rates for

adults was 0.79-0.87 (McClenaghan and Taylor 1993). Mean home range size for 2 populations in Riverside County: 570 sq m and 970 sq m (Bleich 1977). Price et al. (1994) found that the median of the maximum distances moved between captures was about 29 m for 557 individuals; home ranges were stable over time. Predators include owls and various Carnivora.; (NatureServe, 2015)

**Habitat Narrative**

Adult: Habitats include annual grassland and coastal sage scrub with sparse shrub cover, the former more favorable than the latter, commonly in association with *Eriogonum fasciculatum*, *Artemisia californica*, and *Erodium cicutarium* (USFWS 1997). Typical habitat includes sparsely vegetated areas (perennial cover less than 30%) with loose, friable, well-drained soil (generally at least 0.5 m deep) and flat or gently rolling terrain. This species may recolonize abandoned agricultural land. It is most abundant where stands of native vegetation remain (Matthews and Moseley 1990) but decreases as bunchgrass density increases (see Burke et al. 1991). In western Riverside County, shrub removal resulted in increased kangaroo rat densities (Price et al. 1994). Periods of inactivity are spent in underground burrows. Individuals may construct their own burrows or may nest in old burrows of the California ground squirrel or in abandoned burrows of pocket gophers (see Burke et al. 1991, USFWS 1997). In captivity, females construct elaborate nests (Bleich 1977) (NatureServe, 2015).

***Dispersal/Migration*****Motility/Mobility**

Adult: High (inferred from NatureServe, 2015)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory (NatureServe, 2015)

**Dispersal**

Adult: Low: Adults maintained a home range center within 30 meters (100 ft) of where they were first observed. Maximum distances between captures varied between 170 and 350 meters. However, the researchers felt they were underestimating the frequency of long-distance dispersal (USFWS, 1997).

**Dispersal/Migration Narrative**

Adult: Adults maintained a home range center within 30 meters (100 ft) of where they were first observed. Maximum distances between captures varied between 170 and 350 meters. However, the researchers felt they were underestimating the frequency of long-distance dispersal (USFWS, 1997).

***Population Information and Trends*****Population Trends:**

Price and Endo (1988) estimated that the historical habitat had been reduced by about 60 percent by 1984. USFWS (1990) categorized the status as "declining." Decline of 50-70% (NatureServe, 2015)

**Population Growth Rate:**

Price and Endo (1988) estimated that the historical habitat had been reduced by about 60 percent by 1984. USFWS (1990) categorized the status as "declining." Decline of 50-70% (NatureServe, 2015)

**Number of Populations:**

6 - 80 (NatureServe, 2015)

**Population Size:**

10,000 to >1,000,000 individuals (NatureServe, 2015)

**Population Narrative:**

Price and Endo (1988) estimated that the historical habitat had been reduced by about 60 percent by 1984. USFWS (1990) categorized the status as "declining." Decline of 50-70% Total adult population size is unknown but exceeds 10,000. As of the late 1980s, the largest known population included about 14,000 individuals (Burke et al. 1991). Local population density may vary 10-fold with variations in rainfall (populations decline with drought). As in most small mammals, abundance is a misleading index to degree of jeopardy. As of the late 1980s, there were 79 known extant populations (O'Farrell and Uptain 1989; see also Burke et al. 1991). Some of these populations no longer exist whereas subsequent surveys have revealed previously undocumented populations (see USFWS 1997). USFWS (1997) mapped a dozen "significant populations," noting that additional small fragmented populations also exist. (NatureServe, 2015)

***Threats and Stressors***

**Stressor:** Climate change (USFWS, 2010)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Although it is uncertain how climate change will affect Stephens' kangaroo rat or its habitat, modeling predictions suggest more extreme weather events, which could impact the extent of suitable habitat or induce stresses on the species (USFWS, 2010).

**Stressor:** Habitat destruction or modification (USFWS, 2010)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Stephens' kangaroo rat habitat continues to be threatened by habitat degradation from urban development, conversion of native vegetation to nonnative annual grassland, and off-highway vehicles (which directly damage plant communities, as well as the soil crust and the burrow systems) now and in the foreseeable future throughout the Stephens' kangaroo rat's range. Grazing by ungulates is no longer considered to be a rangewide threat, assuming grazing is adequately managed (USFWS, 2010).

**Stressor:** Predation (USFWS, 2010)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Predation by feral and domestic cats is considered to be a threat to the Stephens' kangaroo rat rangewide, and in particular in western Riverside County, now and in the foreseeable future (USFWS, 2010).

**Stressor:** Rodenticides (USFWS, 2010)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Anticoagulant rodenticides, used to control nuisance species, may be consumed by nontarget species, including Stephens' kangaroo rat, even when elevated bait stations are used. Poison bait that falls to the ground or that is cached at ground level by targeted species poses a threat to Stephens' kangaroo rat if ingested during nocturnal foraging or encountered in use of abandoned burrows (USFWS, 2010).

**Stressor:** Small population size (USFWS, 2010)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Small population size continues to affect this species throughout its range and exacerbates the effects of other threats, making Stephens' kangaroo rat susceptible to stochastic events (USFWS, 2010).

## ***Recovery***

### **Reclassification Criteria:**

1. Establishment of four reserves, which encompass at least 6,070 hectares (15,000 acres) of occupied habitat and are permanently protected, funded, and managed, and are located in western Riverside County (inside or outside the Habitat Conservation Plan planning area) (USFWS, 1997).
2. Establishment of one ecosystem based reserve in either western or central San Diego County that is permanently protected, funded, and managed (USFWS, 1997).

### **Delisting Criteria:**

1. A minimum of five reserves in western Riverside County, of which one is ecosystem based, and that encompass at least 6,675 hectares (16,500 acres) of occupied habitat that is permanently protected, funded, and managed (USFWS, 1997).
2. Two ecosystem based reserves in San Diego County. One reserve needs to be established in the Western conservation Planning Area and one reserve needs to be established in the Central Conservation Planning Area. These reserves must be permanently protected, funded, and managed (USFWS, 1997).

### **Recovery Actions:**

- Preserve and protect populations of the Stephen's Kangaroo rat throughout representative portions of its range (USFWS, 1997).
- Protect conserved populations of the Stephen's Kangaroo rat and their habitat (USFWS, 1997).

- Eliminate unnatural mortality factors (USFWS, 1997).
- Establish research program (USFWS, 1997).
- Develop outreach program (USFWS, 1997).

***Conservation Measures and Best Management Practices:***

- Secure and conserve remaining large contiguous blocks of habitat and Stephens' kangaroo rat populations in southern portions of this species' range (i.e., San Diego County) that will lower the extirpation risks associated with lower genetic variability and smaller, fragmented populations (USFWS, 2011).
- Develop and adopt a systematic survey program for monitoring species status and trends across the range of Stephens' kangaroo rat. This objective will include standardization of habitat assessment methodologies across reserves and will require rigorous, detailed, and consistent surveys at appropriate regularity necessary to reliably determine an accurate population status and trend for the species (USFWS, 2011).
- Develop and adopt a centrally organized management plan which employs appropriate management techniques for maintaining suitable habitat quality for Stephens' kangaroo rat. The management plan will prevent habitat loss and degradation and restore degraded habitats through practices directed at increasing occupied habitat, including reduction of nonnative grass density and thatch buildup that inhibits species movement (USFWS, 2011).
- Revise the draft recovery plan to include threats-based and demographic criteria that objectively address current threats. Recovery criteria should be modified and updated to incorporate new information regarding current threats to the species and the quality and maintenance of remaining habitat (Service 2010, p. 51204) (USFWS, 2011).
- Increase funding and support for investigations, which support translocation activities. Encourage hypothesis-driven studies that investigate effects of grazing, controlled burns, vegetation mowing, tilling and scraping as habitat management tools and which test their efficacy in site-specific locations (USFWS, 2011).
- Conduct genetic studies to examine gene flow over a more recent period that will help to clarify impacts of recent habitat fragmentation on Stephens' kangaroo rat genetics, and provide information on the frequency with which genetic exchange occurs between existing populations (USFWS, 2011).

**References**

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## SPECIES ACCOUNT: *Emballonura semicaudata rotensis* (Pacific sheath-tailed bat)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; 11/02/2015; Pacific Region (R1) (USFWS, 2016)

### **Physical Description**

This is a small bat (forearm length about 45 millimeters (1.8 inches), weight 5.5 grams (0.19 ounces), rich brown to dark brown above and paler below. The common name sheath-tailed refers to the nature of the tail attachment; the tail pierces the tail membrane and its tip appears completely free on the upper surface of the membrane.

### **Taxonomy**

Four subspecies of *Emballonura semicaudata* are recognized (Koopman 1997): *Emballonura semicaudata* ssp. *rotensis*, endemic to the Mariana Islands (addressed here); *Emballonura semicaudata* ssp. *sulcata*, occurring in Chuuk and Pohnpei; *Emballonura semicaudata* ssp. *palauensis*, from Palau; and *Emballonura semicaudata* ssp. *semicaudata*, occurring in American and Independent Samoa, Tonga, Fiji, and Vanuatu. The *Emballonura* are an Old World bat family once common and widespread in Polynesia and Micronesia.

### **Historical Range**

U.S.: Guam; Northern Mariana Islands (Rota, Saipan, Tinian) and possibly Anatahan and Maug (Lemke 1986; Steadman 1999; Wiles and Worthington 2002).

### **Current Range**

U.S.: Commonwealth of the Northern Mariana Islands (CNMI), Island of Aguiguan. Aguiguan is uninhabited by humans and is the smallest of the southern islands of the CNMI, only 3 miles (mi) (5 kilometers (km)) long, .9 mi (1.5 km) wide, and 1,730 acres (7 km<sup>2</sup>) in area (Engbring et al. 1986).

### **Distinct Population Segments Defined**

Northern Mariana Islands

### **Critical Habitat Designated**

No;

### ***Life History***

### **Feeding Narrative**

Adult: The Pacific sheath-tailed bat is nocturnal and typically emerges around dusk to forage on insects (Hutson et al. 2001). Fecal pellets collected from two caves on Aguiguan show a diverse array of prey items, but mostly consisting of small-sized prey, with hymenopterans (ants, wasps, and bees), lepidopterans (moths), and coleopterans (beetles) being the three major food items in the diet of bats from both roosts (OShea and Valdez 2009, p. 4). Analysis of presence-absence of foraging bats from echolocation stations deployed across Aguiguan indicate that peak activity

and occurrence is related to canopy cover, vegetation structure, and distance to known roosts, and native limestone forest is preferred foraging habitat (OShea and Valdez 2009, p. 4).

***Dispersal/Migration******Population Information and Trends*****Population Trends:**

Increasing

**Species Trends:**

Increasing

**Representation:**

Low

**Redundancy:**

Low

**Number of Populations:**

5 (caves)

**Population Size:**

359-466 individuals in colonies

**Population Narrative:**

Surveys in 1995 indicated a population of roughly 150 to 250 animals roosting in 5 of 77 caves surveyed, with colony sizes ranging from 2 to 63 individuals (Wiles and Worthington 2002), while 2003 surveys indicated a population of about 400 to 500 animals, however, it was unclear if this difference reflected a population increase (Wiles 2007, pers. comm.). Subsequent surveys conducted in 2008 indicate a small population of Pacific sheath-tailed bat persists on Aguiguan, with a range of 359-466 individuals counted at 5 of 41 caves (OShea and Valdez 2009, p. 3). Comparison with past counts suggests that the population has increased over the past 13 years (OShea and Valdez 2009, p. 3).

***Threats and Stressors***

**Stressor:** Habitat loss and degradation: Past agricultural development and resultant loss of native vegetation.

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Large areas of native forests on the plateaus of the island of Aguiguan were cleared for sugarcane plantations and a large runway and other war-related structures in the 1930s during the Japanese occupation (Engbring et al. 1986, p. 8; Mueller-Dombois and Fosberg 1998, p. 264).

**Stressor:** Habitat loss and degradation: Ongoing grazing by feral goats.

**Exposure:****Response:****Consequence:**

**Narrative:** A large number of feral goats inhabit the island; in fact, the local name for the island is Goat Island. Continued grazing by feral goats poses a serious threat to the foraging habitat of the Pacific sheath-tailed bat (Wiles and Worthington 2002; Esselstyn et al. 2004). Feral goats eat native vegetation, trample roots and seedlings, cause erosion, and promote the invasion of alien plants. They are able to forage in extremely rugged terrain and have a high reproductive capacity (Clarke and Cuddihy 1980; van Riper and van Riper 1982; Scott et al. 1986; Tomich 1986; Culliney 1988; Cuddihy and Stone 1990). Goats on Aguiguan have already limited the regeneration of most tree species on the island, and, over time, could conceivably lead to the complete elimination of forest on the island (Esselstyn et al. 2004). Non-forest habitats that are created as a result of goat browsing and trampling are clearly not utilized by the sheath-tailed bat on Aguiguan (Esselstyn et al. 2004). The CNMI Division of Fish and Wildlife (DFW), considers habitat loss due to feral goat grazing to be the biggest threat to the bat on Aguiguan (Williams 2005, pers. comm.).

**Stressor:** Predation: Possible introduction of the brown tree snake (*Boiga irregularis*), monitor lizard (*Varanus indicus*), and rats (*Rattus* spp.).

**Exposure:****Response:****Consequence:**

**Narrative:** On Guam, the brown tree snake (*Boiga irregularis*) was accidentally introduced after World War II and has caused the decline, extirpation, and extinction of most of the native birds as well as having been involved in significant declines of the Mariana fruit bat (*Pteropus mariannus mariannus*) and various species of native lizards and invertebrates (Wiles 1987; Rodda and Fritts 1992; Wiles et al. 1995; Wiles et al. 2003). There is no evidence linking the brown tree snake to the loss of the Pacific sheath-tailed bat on Guam; however, it is possible that it may have played some role in the species extirpation (Wiles 2007, pers. comm.). Recently, experts agreed that the brown tree snake may be establishing a population on Saipan, though clear evidence of establishment or recruitment is lacking (Colvin et al. 2005; Rodda and Savidge 2007, p. 312) and a few brown tree snakes have been observed on Tinian, although it is not thought to be established there. There is potential for the spread of the brown tree snake to other islands in the CNMI from Guam or Saipan. This potential may be lower for Aguiguan because of the relatively low rate of human traffic to this island, but it is still a possibility. Introduced monitor lizards (*Varanus indicus*) and rats (*Rattus* spp.) also are potential predators of sheath-tailed bats on Aguiguan, but this has not been studied (Wiles and Worthington 2002).

**Stressor:** Other natural or manmade factors affecting its continued existence: The low numbers of individuals.

**Exposure:****Response:****Consequence:**

**Narrative:** The low numbers of individuals of this subspecies and the fact that only one population remains in the Mariana Islands places it at great risk of extinction from inbreeding and stochastic events such as storms (Wiles and Worthington 2002). This threat is particularly significant in cave dwelling species whose population is often highly localized (Wiles and Worthington 2002). Because there are relatively few (300 to 500) animals in only one location, a

severe storm, a disease outbreak, inbreeding depression, or even a disturbance to the roost caves could lead to the extirpation of this subspecies.

### **Recovery**

#### **Recovery Actions:**

- Repeat surveys and continue monitoring for Pacific sheath-tailed bats on Aguiguan every two years.
- Conduct genetic analyses to determine if this subspecies warrants species status.
- Remove feral goats from Aguiguan.
- Restore native forest on Aguiguan.
- Develop a reintroduction plan for reestablishment of populations on other Mariana islands.
- Prevent introduction of brown tree snake to Aguiguan.
- Control and remove mammalian predators on Aguiguan.
- Determine effects of predation by monitor lizards and implementing control measures, as necessary, on Anguiguan.

### **References**

USFWS. 2013. U.S. Fish and Wildlife Service Species Assessment and Listing Priority Assignment Form for *Emballonura semicaudata rotensis* (Pacific Sheath-tailed Bat), 12 p.

USFWS. 2014. Proposed Endangered Status for 21 Species and Proposed Threatened Status for 2 Species in Guam and the Commonwealth of the Northern Mariana Islands

Proposed Rule. 79 Federal Register 190, October 1, 2014. Pages 59362 - 59413

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USFWS. 2013. U.S. Fish and Wildlife Service Species Assessment and Listing Priority Assignment Form for *Emallonura semicaudata rotensis* (Pacific Sheath-tailed Bat), 12 p.

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Proposed Endangered Status for 21 Species and Proposed Threatened Status for 2 Species in Guam and the Commonwealth of the Northern Mariana Islands. Federal Register (79) 59362-59413.

## **SPECIES ACCOUNT: *Emballonura semicaudata semicaudata* (Pacific sheath-tailed bat)**

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### ***Species Taxonomic and Listing Information***

**Listing Status:** Endangered; 10/24/2016; Pacific Region (R1) (USFWS, 2016)

### **Physical Description**

Small bat, approximately 45 millimeters (1.8 inches), males weigh 5.5 grams (0.19 ounces) and females weigh 6.9 grams (0.24 ounces); rich brown to dark brown above and paler below.

### **Taxonomy**

This bat is recognized as one of four subspecies: *Emballonura semicaudata* ssp. *rotensis*, endemic to the Mariana Islands (Guam and the Commonwealth of the Northern Mariana Islands (CNMI)); *E. s. sulcata*, occurring in Chuuk and Pohnpei; *E. s. palauensis*, found in Palau; and *E. s. semicaudata*, occurring in American and Independent Samoa, Tonga, Fiji, and Vanuatu (Koopman 1997). This assessment addresses the population of *E. s. semicaudata* that occurs in American Samoa. After review of the available taxonomic information, we conclude that *E. s. semicaudata* is a valid subspecies.

### **Historical Range**

U.S.: American Samoa, County of Manu'a. American Samoa is made up of seven islands (Tau, Ofu, Olosega, Tutuila, Aunuu, Rose Atoll, and Swains Island). The Pacific sheath-tailed bat formerly occurred on Tau, Ofu, Olosega, collectively known as the Manua (Tau) Islands, Tutuila, and Aunuu (Amerson et al. 1982; Flannery 1995; Helgen and Flannery 2002; Department of Marine and Wildlife Resources (DMWR) 2006).

### **Current Range**

U.S.: American Samoa, County of Manu'a, Island of Tutuila. A precipitous decline of the Pacific sheath-tailed bat on the island of Tutuila has been documented since around 1990 (Amerson et al. 1982; Knowles 1988, Grant et al. 1994; Koopman and Steadman 1995; Helgen and Flannery 2002), and this subspecies may now be extirpated from American Samoa (Grant et al. 1994; Utzurrum 2005, pers. comm.; DMWR 2006). DMWR conducted surveys consisting of acoustic sweeps and cave checks on all of the main islands in an attempt to ascertain whether the species is still extant (Tulafono 2006, pers. comm.). Surveys were completed in 2008; no Pacific sheath-tailed bats were detected during those surveys. New surveys using different methodology are planned to begin in FY2012 (Tulafono 2011, pers. comm.).

### **Distinct Population Segments Defined**

Yes. The American Samoa population is found to be distinct from the Independent Samoa, Tonga, Fiji, and Vanuatu populations.

### **Critical Habitat Designated**

Yes;

### ***Life History***

**Feeding Narrative**

Adult: The Pacific sheath-tailed bat is nocturnal and typically emerges around dusk to forage on insects (Hutson et al. 2001). The biology of this species, including detailed information on reproduction, habitat use, and diet, is largely unknown (Hutson et al. 2001; Wiles and Worthington 2002). (USFWS, 2014)

**Reproduction Narrative**

Adult: The biology of this species, including detailed information on reproduction, habitat use, and diet, is largely unknown (Hutson et al. 2001; Wiles and Worthington 2002). Breeding of Pacific sheath-tailed bats is timed to coincide with offspring born during the onset of the rainy season when there are predictably greater numbers of insect prey. Pacific sheath-tailed bat females produce one pup per litter annually, which translates into relatively low fecundity for the species (Wiles et al. 2011, p. 303). (USFWS, 2015)

**Spatial Arrangements of the Population**

Adult: Colonies (USFWS, 2015)

**Habitat Narrative**

Adult: The biology of this species, including detailed information on reproduction, habitat use, and diet, is largely unknown (Hutson et al. 2001; Wiles and Worthington 2002). The Pacific sheath-tail bat appears to be cave-dependent, roosting during the day in a wide range of caves, including overhanging cliffs, crevices, and lava tubes (Grant 1993; Grant et al. 1994; Hutson et al. 2001; Palmeirim et al. 2005). Populations consist of several roosting colonies. (USFWS, 2014; USFWS, 2015)

***Dispersal/Migration******Population Information and Trends*****Population Trends:**

Declining (USFWS, 2015)

**Species Trends:**

Unknown

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Number of Populations:**

Unknown; may be extinct in American Samoa

**Population Size:**

Unknown

**Population Narrative:**

In an attempt to ascertain whether the species is still extant, American Samoas Department of Marine and Wildlife Resources completed surveys for the Pacific sheath-tailed bat in 2008 and a publication is planned by Dr. Ruth Utzurrum on the results of those surveys. No Pacific sheath-tailed bats were detected during those surveys. New surveys using different methodology are planned to begin in FY2012. These surveys will utilize passive acoustic detection systems that will be deployed to various locations on Tutuila and Manua for extended (two weeks to one month) periods of time. These systems will passively record high frequency sounds throughout the night and data will be downloaded periodically to search for call signatures of the Pacific sheath-tailed bat. Additionally a vehicle based system will automatically record high frequency noises when vehicles are driven at night. If a bat is detected, more intensive surveys will be immediately started.

**Threats and Stressors**

**Stressor:** Loss and degradation of roosting and foraging habitat: Storms (USFWS, 2014)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The reasons for the decline of the sheath-tailed bat in American Samoa are unclear; however, the loss of roosting caves as a result of severe storms in American Samoa is believed to be a threat to this species (Craig et al. 1993; Grant et al. 1994). Two caves at Anapeapea Cove were reported as roosting sites for most of the bats estimated in 1976 and 1977 (Amerson et al. 1982). Severe storms, particularly Typhoons Ofa (1990) and Val (1991), removed the dense vegetation that had obscured the entrance to the larger cave at Anapeapea, inundated the cave with water, filled the cave with coral and fallen trees, and washed the cave walls clean (Grant et al. 1994). No bats were reported in either cave at Anapeapea during 1993 surveys (Grant 1993; Grant et al. 1994). Only small numbers of bats have been observed in other caves during those surveys, but there is no information on how many other caves there are, or how many bats they could support (Grant 1993; Grant et al. 1994). In addition to the effects on the roost caves, storms are likely to have a negative impact on bats due to the deforestation (Palmeirim et al. 2005) that leads to loss of foraging habitat and result in starvation of the bats. (USFWS, 2014)

**Stressor:** Predation: Rats and feral domestic cats (USFWS, 2014)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Rats (*Rattus* spp.) have been postulated as a problem for the Mariana Island subspecies of the Pacific sheath-tailed bat (Wiles and Worthington 2002) and are potential predators of Pacific sheath-tailed bats in American Samoa; however, the extent of this possibility is unknown. In addition, it is well known that domestic cats (*Felis catus*) can capture low flying bats, and it has been documented that they wait for bats as they emerge from caves and capture them in flight (Tuttle 1977; Ransome 1990; Woods et al. 2003). Consequently, even a few cats can have a major impact on a population of cave-dwelling bats (Palmeirim et al. 2005). Palmeirim et al. (2005) indicates that, of the predators introduced to Fiji, cats are the most likely to prey on bats. Feral cats are present on Tutuila and on Manua Islands in American Samoa (Freifeld 2007,

pers. comm.). The role of these introduced predators in the decline of the Pacific sheath-tailed bat is not proven but it is strongly supported as a factor (Palmeirim et al. 2005). Likewise, the role of disease in the species decline is not known as it has not been studied; however, disease could be a factor, especially for a communally roosting species such as the Pacific sheath-tailed bat (Wiles and Worthington 2002). (USFWS, 2014)

**Stressor:** Other threats: Low numbers of individuals and populations (USFWS, 2014)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The low numbers of individuals and populations of this subspecies place this bat at great risk of extinction from inbreeding and stochastic events such as storms (Wiles and Worthington 2002). The threat is significant for cave-dwelling species whose populations are often highly localized with few numbers of animals that can easily be lost in a severe storm, disease outbreak, or disturbance to the roost caves (Wiles and Worthington 2002). (USFWS, 2014)

**Stressor:** Human disturbance (USFWS, 2015a)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** human disturbance of caves for guano mining and shelter during World War II, bombing and shelling during World War II, indiscriminate use of pesticides, predation by monitor lizards, rats, and brown treesnakes, increasingly isolated populations, and loss of foraging habitat due to human conversion and destruction and alteration by typhoons and nonnative plants and animals. (USFWS, 2015a)

**Stressor:** Habitat Destruction and Modification by Deforestation (USFWS, 2015b)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Deforestation can cause the destruction and modification of foraging habitat of the Pacific sheath-tailed bat as a result of the loss of cover and reduction of available insect prey. The loss of native plant diversity associated with the conversion of native forests to agriculture and other uses can result in a corresponding reduction in the diversity and number of flying insects (Hespenheide 1975, pp. 84, 96; Waugh and Hails 1983, p. 212; Tarburton 2002, p. 107). Deforestation results from logging, agriculture, and development (Government of Samoa 2001, p. 59; Wiles and Worthington 2002, p. 18) and from hurricanes. Based on the preference of the Mariana subspecies for foraging in forested habitats near their roost caves, Wiles et al. (2011, p. 307) predict that past deforestation in the Mariana archipelago may be a principal factor in limiting their current population to the island of Aguiuan, which has healthy native forest. Similarly, in Fiji, most sheath-tailed bat colonies are found roosting in caves in or near good forest (e.g., closed canopy, native forest) (Palmeirim et al. 2005, pp. 36, 44); however, much of it has been lost on the large Fijian islands (Palmeirim et al. 2007, p. 515). Deforestation has been extensive and is ongoing across the range of the Pacific sheath-tailed bat. On the island of Tutuila, American Samoa, agriculture and development cover approximately 24 percent of the island and are concentrated in the coastal plain and low-elevation areas where loss of forest is likely to have modified foraging habitat for sheath-tailed bats (American Samoa Community

College (ASCC) 2010, p. 13). In Samoa, the amount of forested area declined from 74 to 46 percent of total land area between 1954 and 1990 (Food and Agricultural Organization (FAO) 2005 in litt.). Between 1978 and 1990, 20 percent of all forest losses in Samoa were attributable to logging, with 97 percent of the logging having occurred on Savaii (Government of Samoa 1998 in Whistler 2002, p. 132). Forested land area in Samoa continued to decline at a rate of roughly 2.1 percent or 7,400 ac (3,000 ha) annually from 1990 to 2000 (FAO 2005 in litt.). As a result, there is very little undisturbed, mature forest left in Samoa (Watling 2001, p. 175; FAO 2005 in litt.). Today, only 360 ac (146 ha) of native lowland rainforests (below 2,000 ft or 600 m) remain on Savaii and Upolu as a result of logging, agricultural clearing, residential clearing (including relocation due to tsunami), and natural causes such as rising sea level and hurricanes (Ministry of Natural Resources and Environment (MNRE) 2013, p. 47). On Upolu, direct or indirect human influence has caused extensive damage to native forest habitat (above 2,000 ft or 600 m) (MNRE 2013, p. 13). Although forested, almost all upland forests on Upolu are largely dominated by introduced species today. Savaii still has extensive upland forests, which are for the most part undisturbed and composed of native species (MNRE 2013, p. 40). Although the large Fijian islands still have some areas of native forest, much of it has been lost (e.g., 17 percent between 1990 and 2000; FAO 2005 in litt.), and commercial logging continues (Palmeirim et al. 2007, p. 515). The best available information does not provide the current status of native forests and rates of forest loss in Tonga or Vanuatu. Native forests are preferred foraging habitat of the Pacific sheath-tailed bat, and deforestation is occurring in Fiji (where the last relatively large population occurs), and in Samoa, and has occurred in American Samoa. Therefore we conclude that habitat destruction and modification by deforestation is a current threat to the species in at least Fiji and Samoa, which comprise roughly 62 percent of the land area, and occupy the center, of the bat's range (USFWS, 2015b).

**Stressor:** Habitat Destruction and Modification by the Effects of Climate Change (USFWS, 2015b)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Climate change may have impacts to the habitat of the Pacific sheath-tailed bat. The terms "climate" and "climate change" are defined by the Intergovernmental Panel on Climate Change (IPCC). "Climate" refers to the mean and variability of different types of weather conditions over time, with 30 years being a typical period for such measurements, although shorter or longer periods also may be used (IPCC 2013, p. 1,450). The term "climate change" thus refers to a change in the mean or variability of one or more measures of climate (e.g., temperature or precipitation) that persists for an extended period, typically decades or longer, whether the change is due to natural variability, human activity, or both (IPCC 2013, p. 1,450). Various types of changes in climate can have direct or indirect effects on species. These effects may be positive, neutral, or negative and they may change over time, depending on the species and other relevant considerations, such as the effects of interactions of climate with other variables (e.g., habitat fragmentation) (IPCC 2007, pp. 8–14, 18). Climate change will be a particular challenge for the conservation of biodiversity because the introduction and interaction of additional stressors may push species beyond their ability to survive (Lovejoy 2005, pp. 325–326). The synergistic effects of climate change and habitat fragmentation are the most menacing facet of climate change for biodiversity (Hannah et al. 2005, p. 4). Currently, there are no climate change studies that address impacts to the specific habitat of the Pacific sheath-tailed bat. There are, however, climate change studies that address potential changes in the tropical Pacific on a broader scale. In our analyses, we reference the scientific assessment and climate change

predictions for the western Pacific region prepared by the Pacific Climate Change Science Program (PCCSP), a collaborative research partnership between the Australian Government and 14 Pacific Island countries, including Samoa, Tonga, Fiji, and Vanuatu (Australian BOM and CSIRO 2011 Vol. 1, p. 15). The assessment builds on the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC), and presents regional predictions for the area roughly between 25° S. to 20° N. and 120° E. to 150° W. (excluding the Australian region south of 10° S. and west of 155° E.) (Australian BOM and CSIRO 2011 Vol. 1, pp. 14, 20). The findings for Samoa (13° S. and 171° E.) may be used as a proxy for American Samoa (14° S. and 170° W.). The annual average air temperatures and sea surface temperatures are projected to increase in American Samoa, Samoa, Fiji, Tonga, and Vanuatu, as well as throughout the western Pacific region (Australian BOM and CSIRO 2011 Vol. 2, pp. 91, 198, 228, 258). The projected regional warming is around 0.5–1.0 °C by 2030, regardless of the emissions scenario. By 2055, the warming is generally 1.0–1.5 °C with regional differences depending on the emissions scenario. Projected changes associated with increases in temperature include, but are not limited to, changes in mean precipitation with unpredictable effects on local environments (including ecosystem processes such as nutrient cycling), increased occurrence of drought cycles, increases in the intensity and number of severe storms, sea-level rise, a shift in vegetation zones upslope, and shifts in the ranges and lifecycles of individual species (Loope and Giambelluca 1998, pp. 514–515; Pounds et al. 1999, pp. 611–612; IPCC AR4 2007, p. 48; Emanuel et al. 2008, p. 365; U.S. Global Change Research Program (US-GCRP) 2009, pp. 145–149, 153; Keener et al. 2010, pp. 25–28; Sturrock et al. 2011, p. 144; Townsend et al. 2011, pp. 14–15; Warren 2011, pp. 221–226; Finucane et al. 2012, pp. 23–26; Keener et al. 2012, pp. 47–51). In the western Pacific region, increased ambient temperatures is projected to lead to increases in annual mean rainfall, the number of heavy rain days (20–50 mm), and extreme rainfall events in American Samoa, Samoa, Fiji, Tonga, and Vanuatu (Australian BOM and CSIRO 2011 Vol. 1, p. 178; Australian BOM and CSIRO 2011 Vol. 2, pp. 87–88, 194–195, 224–225, 254–255). Impacts of increased precipitation on the Pacific sheath-tailed bat are unknown. Hurricanes are projected to decrease in frequency in this part of the Pacific but increase in severity as a result of global warming (Australian BOM and CSIRO 2011 Vol. 2, pp. 88, 195, 225, 255). The high winds, waves, strong storm surges, high rainfall, and flooding associated with hurricanes, particularly severe hurricanes (with sustained winds of 150 mi (240 km) per hour), have periodically caused great damage to roosting habitat of Pacific sheath-tailed bats and to native forests that provide their foraging habitat (Craig et al. 1993, p. 52; Grant et al. 1994, p. 135; Tarburton 2002, pp. 105–108; Palmeirim et al. 2005, p. 35), as described in the “Hurricanes” section, above. In the western Pacific region, sea level is projected to rise 1.18 to 6.3 in (30 to 160 mm) by 2030, 2.6 to 12.2 in (70 to 310 mm) by 2055, and 8.3 in to 2 ft (210 to 620 mm) by 2090 under the highemissions scenario (Australian BOM and CSIRO 2011 Vol. 2, pp. 91, 198, 228, 258). The Pacific sheath-tailed bat is known to roost in areas close to the coast and forage in the adjacent forested areas at or near sea-level, as well as inland and at elevations up to 2,500 ft (762 m). The impacts of projected sealevel rise on low-elevation and coastal roosting and foraging habitat are likely to reduce and fragment the bat’s habitat on individual high islands. In summary, although we lack information about the specific effects of projected climate change on the Pacific sheath-tailed bat, we anticipate that increased ambient temperature, precipitation, hurricane intensity, and sea-level rise and inundation would create additional stresses on the bat and on its roosting and foraging habitat because it is vulnerable to these disturbances. The risk of extinction as a result of the effects of climate change increases when a species’ range and habitat requirements are restricted, its habitat decreases, and its numbers and number of populations decline (IPCC 2007, pp. 8–11). In addition, the fragmented range, diminished number of populations, and low total number of individuals

have caused the Pacific sheath-tailed bat to lose redundancy and resilience rangewide. Therefore, we would expect the Pacific sheath-tailed bat to be particularly vulnerable to the habitat impacts of projected environmental effects of climate change (Loope and Giambelluca 1998, pp. 504– 505; Pounds et al. 1999, pp. 611–612; Still et al. 1999, p. 610; Benning et al. 2002, pp. 14,246–14,248; Giambelluca and Luke 2007, pp. 13–15). Based on the above information, we conclude that habitat impacts resulting from the effects of climate change are not a current threat but are likely to become a threat to the Pacific sheath-tailed bat in the future (USFWS, 2015b).

**Stressor:** Roost Disturbance (USFWS, 2015b)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Disturbance of roosting caves has contributed to the decline of the Pacific sheath-tailed bat throughout its range. Disturbance of roost caves by humans is likely to have occurred as a result of recreation, harvesting of co-occurring bat species, and, more commonly, guano mining (Grant et al. 1994, p. 135; Tarburton 2002, p. 106; Wiles and Worthington 2002, p. 17; Palmeirim et al. 2005, pp. 63, 66; Malotau 2012a in litt.; Malotau 2012b in litt.). Roost disturbance is a well-known problem for many cave-dwelling species (Palmeirim et al. 2005, p. 3). Roosts are important sites for bats for mating, rearing young, and hibernating (in mid- and highlatitude species). Roosts often facilitate complex social interactions, offer protection from inclement weather, help bats conserve energy, and minimize some predation risk (Kunz and Lumsden 2003, p. 3); therefore, disturbance at caves and being repeatedly flushed from their roosts may cause bats to incur elevated energetic costs and other physiological stress and potentially increased risk of predation while in flight. Roost disturbance thus would negatively affect the survival and reproduction of the Pacific sheath-tailed bat. In American Samoa, human disturbance at the two caves known to be historical roost sites for the bat is likely to be minimal. Guano mining occurred in the Anapeapea caves in the 1960s (Amerson et al. 1982, p. 74), but ceased due to the high salt content as a result of flooding with seawater during cyclones (Grant et al. 1994, p. 135). On Taveuni, Fiji, a cave known to be used as a roosting cave for the Pacific sheath-tailed bat is under more immediate threat by humans, as the cave is situated close to farmland, and is often used by locals (Malotau 2012a, p. 3). On Upolu, Samoa, caves previously known to support bats are well-known and often visited by tourists; one within O le Pupu Pue National Park and others on village land (Tarburton 2011, pp. 40, 44). Swiftlets (*Aerodramus* spp.) are still observed in significant numbers in these caves (Tarburton 2011, p. 40), but these birds may be more tolerant than bats of human disturbance. We do not have information on human disturbance of roosts in Tonga or Vanuatu. Goats are certain to enter caves for shelter from the sun and consequently can disturb roosting bats, although the extent of this disturbance is unknown (Scanlon 2015b, in litt.). Feral goats have been observed entering caves on Aguiuan Island for shelter, which disrupts colonies of the endangered swiftlet and is believed to disturb the Mariana subspecies of the Pacific sheath-tailed bat (Wiles and Worthington 2002, p. 17; Cruz et al. 2008, p. 243; Scanlon 2015b, in litt.). Researchers found that if caves that were otherwise suitable for bats were occupied by goats, there were no bats present in the caves (Guam Division of Aquatic and Wildlife Resources 1995, p. 95). On Yaqeta Island, Fiji, a cave once known to support several hundred Pacific sheath-tailed bats but now abandoned, is located within a small forest fragment frequented by goats (Scanlon et al. 2013, p. 453). Populations of the Pacific sheath-tailed bat are concentrated in the caves where they roost, and chronic disturbance of these sites can result in the loss of populations, as described above. Because so few populations

of this bat remain, loss of additional populations to roost disturbance further erodes its diminished abundance and distribution. Based on the above information, roost disturbance at caves accessible to humans and animals such as feral goats is a current threat and will likely continue to be a threat into the future (USFWS, 2015b).

**Stressor:** Pesticides (USFWS, 2015b)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The use of pesticides may negatively affect the Pacific sheath-tailed bat as a result of direct toxicity and a reduction in the availability of insect prey. Pesticides are known to adversely affect bat populations, either by secondary poisoning when bats consume contaminated insects or by reducing the availability of insect prey (Hutson et al., 2001, p. 138; Mickleburgh et al. 2002, p. 19). Pesticides may have contributed to declines and loss of the Mariana subspecies of Pacific sheath-tailed bat on islands where pesticides were once applied in great quantities (Guam, Saipan, and Tinian) (Wiles and Worthington 2002, p. 17). In American Samoa and Samoa, current levels of pesticide use are likely lower than several decades ago when their use, particularly during the years in which taro was grown on large scales for export (1975–1985), coincided with the decline of bats in both places and has been implicated as the cause (Tarburton 2002, p. 107). However, Grant et al. (1994, pp. 135–136) dismissed the role of insecticides in the decline of the bat in American Samoa based on the absence of a similar population crash in the insectivorous white-rumped swiftlet (*Aerodramus spodiopygius*) and the limited use of agricultural and mosquito-control pesticides. On the island of Taveuni in Fiji, where bat populations have persisted at low levels over the last 10 years (Palmeirim et al. 2005, p. 62, Malotau 2012, in litt.), several locals reported that pesticide use was quite widespread, and their use may be similar on other Fijian islands (Malotau 2012, in litt.). We do not have information about pesticide use in Tonga or Vanuatu. The best available information does not lead us to conclude that the use of pesticides is a current threat to the Pacific sheath-tailed bat or that it is likely to become one in the future (USFWS, 2015b)

**Stressor:** Habitat Destruction and Modification by the Feral Goats (USFWS, 2016)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Overgrazing by nonnative feral goats has resulted in the destruction and degradation of forests on island ecosystems (Esselsytn et al. 2004, p. 307; Palmeirim et al. 2005, p. 46; Berger et al. 2011, pp. 36, 38, 40, 42–47; CNMI–SWARS 2010, p. 15; Kessler 2011, pp. 320–323; Pratt 2011, pp. 2, 36; Welch et al. 2016). Overgrazing of the forest understory by goats resulted in little or no recruitment of canopy tree species in areas of known populations of the Pacific sheath-tailed bat on small islands in the Lau Group in Fiji (Palmeirim et al. 2005, p. 46) and on Aguiguan Island in the Northern Mariana Islands, where the endangered Mariana subspecies (*E. semicaudata rotensis*) occurs (Gorreson et al. 2009, p. 339). Palmeirim et al. (2005, p. 46) predicted that continued overgrazing would result in the demise of the forests that are so important for the Pacific sheath-tailed bat. Despite the reported negative impacts of goat browsing on tree recruitment, the current amount of well-developed forest canopy habitat and availability of food resources suggest that the bat is currently able to persist on islands where feral goat browsing is occurring (Esselsytn et al. 2004, p. 307; Palmeirim et al. 2005, pp. 28–29). However, because the direct and indirect impacts of goat browsing on the preferred foraging

habitat of the bat are currently occurring and expected to continue into the future in Fiji, we conclude that habitat destruction and degradation by goat browsing is a threat to the continued existence of the bat in Fiji (USFWS, 2016).

### **Recovery**

#### **Recovery Actions:**

- Complete surveys to determine whether the species is still extant on American Samoa;
- Continue to conduct studies to determine causes and extent of decline;
- Protect roost caves from disturbance;
- Control and remove nonnative predators in and around roost caves;
- Conduct genetic analyses to determine if this subspecies warrants species status;
- Develop species augmentation or reintroduction plan, as appropriate.

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11 p.

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Endangered Status for Five Species From American Samoa

Final Rule. Federal Register Vol. 81, No. 184.

## SPECIES ACCOUNT: *Eumops floridanus* (Florida bonneted bat)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; Southeast Region (R4) (USFWS, 2015)

### **Physical Description**

The taxon was originally listed as endangered in the State of Florida as the Florida mastiff bat (*Eumops glaucinus floridanus*) (Florida Administrative Code, Chapter 68). A biological status review recognized the taxon as the Florida bonneted bat, and the State's current threatened and endangered list uses both names, Florida bonneted (mastiff) bat, *Eumops (=glaucus) floridanus*. The dorsum is black or brownish-gray to cinnamon-brown, slightly paler grayish below; fur is short and glossy; hairs are bicolored, lighter at the base; distal half of tail projects beyond interfemoral membrane; the largest bat in Florida, where total length is 126-165 mm; forearm length 57.9-69.2 mm; hind foot 10.8-15.0 mm; leathery rounded ears are joined at the midline and project forward, ear length 19.9-31.0 mm; tragus broad and truncate distally; mass 30.2 g to at least 55.4 g in pregnant females (Belwood 1992, Hall 1981). Males and females are not significantly different in size (Timm and Genoways 2004). Timm and Genoways (2004) found no pattern of size related geographic variation in this species.

### **Taxonomy**

The taxon differs from *Tadarida brasiliensis* in being larger (maximum total length of *brasiliensis* is about 105 mm) and having the ears joined at the midline. No other Florida bats have a tail that extends far beyond the interfemoral membrane.

### **Historical Range**

Records indicating historical range are limited. Morgan (1991) indicated that *E. glaucinus* had been identified from four late Pleistocene (approximately 11,700 years ago) and Holocene (time period beginning 10,000 years ago) fossil sites in the southern half of the Florida peninsula. Late Pleistocene remains are known from Melbourne, Brevard County, and Monkey Jungle Hammock in Miami-Dade County (Allen 1932; Martin 1977, as cited in Belwood 1981 and Timm and Genoways 2004; Morgan 1991). Holocene remains are known from Vero Beach, Indian River County (Ray 1958; Martin 1977; and Morgan 1985, 2002 as cited in Timm and Genoways 2004; Morgan 1991), and also Monkey Jungle Hammock (Morgan 1991). The largest fossil sample (9 specimens) was reported from the Holocene stratum at Vero Beach (Morgan 1985 as cited in Morgan 1991). The fossil records from Brevard County and Indian River County are considerably farther north than where living individuals have typically been recorded (Timm and Genoways 2004; Marks and Marks 2008b). Most of the historical records and sightings for this species are several decades old from the cities of Coral Gables and Miami in extreme southeastern Florida, where the species was once believed to be common (Belwood 1992; Timm and Genoways 2004; Timm and Arroyo-Cabrales 2008). G.T. Hubbell also reported a female with young from Fort Lauderdale in Broward County; all of his sightings of Florida bonneted bats were near human dwellings (Belwood 1992). Prior to 1967, G.T. Hubbell regularly heard loud, distinctive calls at night as the bats foraged above buildings and he routinely obtained several individuals per year that were collected during the winter months from people's houses (Belwood 1992). Other early literature also mentioned Fort Lauderdale as an area where the species occurred (Barbour and Davis 1969; Belwood 1992). However, in their comprehensive review, none of the specimens examined by Timm and Genoways (2004) were from Broward County. Belwood

(1981) found a colony in Punta Gorda; however, the longleaf pine in which the bats roosted was felled during highway construction. Recent specimens are only known from extreme southern and southwestern Florida, including Miami-Dade County on the east coast and Charlotte, Collier, and Lee Counties on the Gulf coast (Timm and Genoways 2004).

### **Current Range**

Based upon available information, the Florida bonneted bat appears to be restricted to south and southwest Florida. The core range may primarily consist of habitat within Charlotte, Lee, Collier, Monroe, and Miami-Dade Counties. Recent data also suggest use of portions of Okeechobee and Polk Counties and possible use of areas within Glades County. However, given available data, it is not clear to what extent areas outside of the core range may be used. It is possible that areas outside of the south and southwest Florida are used only seasonally or sporadically. Alternatively, these areas may be used consistently, but the species was not regularly detected due to the limitations of available data, survey methods, and search efforts.

### **Critical Habitat Designated**

Yes;

### ***Life History***

#### **Feeding Narrative**

Adult: Foraging behavior – Precise foraging and roosting habits and long-term requirements are unknown (Belwood 1992). Active year-round, the species is likely dependent upon a constant and sufficient food supply, consisting of insects, to maintain its generally high metabolism. Based upon limited information, Florida bonneted bats feed on flying insects of the following orders: Coleoptera (beetles), Diptera (true flies), and Hemiptera (true bugs) (Belwood 1981; Belwood 1992; FBC 2005). An analysis of bat guano (droppings) from the colony using the pine flatwoods in Punta Gorda indicated the sample (by volume) contained coleopterans (55 percent), dipterans (15 percent), and hemipterans (10 percent) (Belwood 1981; Belwood 1992). Molossids, in general, seem adapted to fast flight in open areas (Vaughan 1966). Various morphological characteristics (e.g., narrow wings, high wing-aspect ratios (ratio of wing length to its breadth) make *Eumops* well-adapted for efficient, rapid, and prolonged flight in open areas (Findley et al. 1972; Freeman 1981; Norberg and Rayner 1987; Vaughan, 1959 as cited in Best et al. 1997). Barbour and Davis (1969) noted that the species flies faster than smaller bats, but cannot maneuver as well in small spaces. Belwood (1992) stated *E. glaucinus* is “capable of long, straight, and sustained flight,” which should allow individuals to travel large distances. Norberg and Rayner (1987) attributed long distance flights of Brazilian free-tailed bats to their high wing-aspect ratios, with that species capable of traveling 65 km (40 miles) from its roosting site to its foraging areas (Barbour and Davis 1969). Nonetheless, average foraging distances for the Florida bonneted bat are not known (G. Marks, personal communication 2012). Although the species can fly long distances, it likely does not travel farther than necessary to acquire food needed for survival (G. Marks, personal communication 2012). Bonneted bats are “fast hawking” bats that rely on speed and agility to catch target insects in the absence of background clutter, such as dense vegetation (Simmons et al. 1979; Belwood 1992; Best et al. 1997). Foraging in open spaces, these bats use echolocation to detect prey at relatively long range, roughly 3 to 5 m (10 to 16 ft) (Belwood 1992). Based upon information from G.T. Hubbell, Belwood (1992) indicated that individuals leave roosts to forage after dark, seldom

occur below 10 m (33 ft) in the air, and produce loud, audible calls when flying; calls are easily recognized by some humans (Belwood 1992; Best et al. 1997; Marks and Marks 2008a).

### **Reproduction Narrative**

Adult: The Florida bonneted bat has a fairly extensive breeding season during summer months (Timm and Genoways 2004). The maternity season for most bat species in Florida occurs from mid-April through mid-August (Marks and Marks 2008a). During the early portion of this period, females give birth and leave young in the roost while they make multiple foraging excursions to support lactation (Marks and Marks 2008a). During the latter portion of the season, young and females forage together until the young become sufficiently skilled to forage and survive on their own (Marks and Marks 2008a). The Florida bonneted bat is a subtropical species, and pregnant females have been found in April through September (FBC 2005; Marks and Marks 2008a; J. Myers, personal communication, 2015). Examination of limited data suggests this species may be polyestrous (having more than one period of estrous in a year), with a second birthing season possibly in January–February (Timm and Genoways 2004; FBC 2005). Information on reproduction and demography is sparse. The Florida bonneted bat has low fecundity; litter size is one (FBC 2005; Timm and Arroyo-Cabrales 2008). Lifespan – Relatively little is known about the Florida bonneted bat's life history. Lifespan is not known. Based upon the work of Wilkinson and South (2002), Gore et al. (2010) inferred a lifespan of 10 to 20 years for the Florida bonneted bat, with an average generation time of 5 to 10 years.

### **Habitat Narrative**

Adult: Habitat – Relatively little is known of the ecology of the Florida bonneted bat, and long-term habitat requirements are poorly understood (Robson 1989; Robson et al. 1989; Belwood 1992; Timm and Genoways 2004). Habitat for the Florida bonneted bat mainly consists of foraging areas and roosting sites, including artificial structures. At present, only two active, natural roost sites are known, and only limited information on historical sites is available. Recent information on foraging habitat has been obtained largely through acoustical surveys, designed to detect and record bat echolocation calls (Marks and Marks 2008a). Acoustical methods have generally been selected over mist netting as the primary survey methodology because this species flies and primarily forages at heights of 9 m (30 ft) or more (Marks and Marks 2008a). The Florida bonneted bat has a unique and easily identifiable call. While most North American bats vocalize echolocation calls in the ultrasonic range that are inaudible to humans, the Florida bonneted bat echolocates at the higher end of the audible range, which can be heard by some humans as high-pitched calls (Marks and Marks 2008a). Most surveys conducted using acoustical equipment can detect echolocation calls within a range of 30 m (100 ft); call sequences are analyzed using software that compares calls to a library of signature calls (Marks and Marks 2008a). Florida bonneted bat calls are relatively easy to identify because calls are issued at frequencies well below that of other Florida bat species (Marks and Marks 2008a). In general, open, fresh water and wetlands provide prime foraging areas for bats (Marks and Marks 2008c). Bats will forage over ponds, streams, and wetlands and drink when flying over open water (Marks and Marks 2008c). During dry seasons, bats become more dependent on remaining ponds, streams, and wetland areas for foraging purposes (Marks and Marks 2008c). The presence of roosting habitat is critical for day roosts, protection from predators, and the rearing of young (Marks and Marks 2008c). For most bats, the availability of suitable roosts is an important, limiting factor (Humphrey 1975). Bats in south Florida roost primarily in trees and manmade structures (Marks and Marks 2008a). Available information on roosting sites for the Florida bonneted bat is extremely limited. Roosting and foraging areas appear varied, with the

species occurring in forested, suburban, and urban areas (Timm and Arroyo-Cabrales 2008). Data from acoustical surveys and other methods suggests the species uses a wide variety of habitats (Marks and Marks 2008a; 2008b; 2008c; 2012; R. Arwood, Inside-Out Photography, Incorporated, personal communication 2008a, 2008b, 2012; Smith 2010; S. Snow, personal communication 2011a, 2011b, 2012).

### ***Dispersal/Migration***

#### **Motility/Mobility**

Adult: High

#### **Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory

#### **Dispersal/Migration Narrative**

Adult: The Florida bonneted bat is active year-round and does not have periods of hibernation or torpor. The species is not migratory, but there might have been seasonal shifts in roosting sites (Timm and Genoways 2004).

### ***Population Information and Trends***

#### **Population Trends:**

Unknown

#### **Number of Populations:**

~26 colonies

#### **Population Size:**

Unknown. Estimate 286

#### **Population Narrative:**

Little information exists on historical population levels. The Florida bonneted bat was considered common in the Miami-Coral Gables area because of regular collection of specimens from 1951 to 1965 (Robson 1989; Belwood 1992). Jennings (1958) indicated the species was not abundant, noting a total of 20 individuals had been taken from 1936 to 1958. Prior to 1967, G.T. Hubbell regularly heard loud, distinctive calls at R. Timm, personal communication 2012). night as the bats foraged above buildings in the Miami area, and he routinely obtained several individuals per year that were collected from people's houses (Belwood 1992). Barbour and Davis (1969) indicated that, on average, about two individuals per year are brought to the Crandon Park Zoo in Miami, due to injuries, but no time period was specified. Unpublished data from a survey of 100 pest control companies in 1982 on the southeastern coast of Florida showed that requests to remove "nuisance" bats from this area all but ceased beginning in the 1960s (Belwood 1992), indicating a sharp decline in bats in general. Timm and Genoways (2004) found only three records of Florida bonneted bats in the greater Miami area after 1965. The colony found near Punta Gorda in 1979 appeared to be the only recorded occurrence since 1967 (Belwood 1981). A 6-week field trip in 1980 to locate other occurrences was unsuccessful and led to the belief this species was "probably extinct in Florida" (Belwood 1992). No new evidence of this species was found from 1979 until 1988 when Robson et al. (1989) found a

pregnant female in Coral Gables (Robson 1989). Timm and Genoways (2004) surmised the Florida bonneted bat may have been uncommon for several decades, based upon the work of previous researchers (Barbour 1945 as cited in Timm and Genoways 2004; Jennings 1958; Layne 1974), who noted the scarcity of bats in southern Florida. Owre (1978) observed fewer than a dozen individuals in roughly 25 years and noted few mammalogists had success in finding the species. Robson (1989) indicated the decline of specimens and sightings in the mid-1960s is reflected in the museum record and noted the 1950s and 1960s was a period of rapid growth in the Miami area. Robson (1989) suggested the resulting disturbance and destruction of native habitat may have flushed a large number of specimens out of established roosts, resulting in a high collection rate. A status survey conducted in 1989, encompassing 25 sites within natural areas within a nine county area, found no new evidence of this species (Robson 1989). Based upon available data and information, the Florida bonneted bat occurs within a restricted range and in low abundance (Marks and Marks 2008a; 2012; Timm and Arroyo-Cabrales 2008; FWC 2011a, 2011b; Actual population size is not known, and no population viability analyses are available (FWC 2011b). However, population size is thought to be less than that needed for optimum viability (Timm and Arroyo-Cabrales 2008). As part of their evaluation of listing criteria for the species, Gore et al. (2010) found the extent of occurrence appears to have declined on the east coast, but trends on the west coast could not be inferred due to limited information. Results of the 2006-2007 range-wide survey suggested that the Florida bonneted bat is a rare species with limited range and low abundance (Marks and Marks 2008a). Based upon results of both the range-wide study and survey of select public lands, the species was found at 12 locations (Marks and Marks 2008b), but the number and status of the bat at each location are unknown. Based upon the small number of locations where calls were recorded, the low numbers of calls recorded at each location, and the fact that the species forms small colonies, Marks and Marks (2008a) stated that it is possible that the entire population of Florida bonneted bats may number less than a few hundred individuals. Results of the 2010 to 2012 surveys and additional surveys by other researchers identified new occurrences within the established range (i.e., within Miami area, areas of ENP and Big Cypress National Preserve [BCNP]) (S. Snow, personal communication 2011a, 2011b, 2012; R. Arwood, personal communication 2012; Marks and Marks 2012), however, not in sufficient numbers to alter previous population estimates. In their 2012 report on the status of the species, Marks and Marks (2012) provided an updated estimation of population size, based upon 120 nights of surveys at 96 locations within peninsular Florida, results of other known surveys, and personal communications with others involved in Florida bonneted bat work. Based upon an average colony size of 11 and an estimated 26 colonies within the species' range, researchers estimated the total Florida bonneted bat population at 286 bats (Marks and Marks 2012).

### ***Threats and Stressors***

**Stressor:**

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** This bat is vulnerable to habitat loss (in urban and forested areas), habitat alteration (removal of old trees with cavities, or buildings with spaces suitable for roosting), and detrimental effects of pesticide spraying for mosquitoes. The last may be responsible for the species' decline in the Miami area, as roosting sites are still abundant. Severe hurricanes may cause loss of older trees with roosting cavities. Hurricane Andrew, an intense Category 5

hurricane that struck southeastern Florida in 1992, may have had a significant impact upon the already low population of bonneted bats (Timm and Genoways 2004). USFWS (2013) summarized threats as follows: Habitat loss, degradation, and modification from human population growth and associated development and agriculture have impacted the Florida bonneted bat and are expected to further curtail its limited range. The effects resulting from climate change, including sea-level rise and coastal squeeze, are expected to become severe in the future and result in additional habitat losses, including the loss of roost sites and foraging habitat. The species is also facing threats from a wide array of natural and manmade factors, including small population size, restricted range, few colonies, slow reproduction, low fecundity, and relative isolation. Existing regulatory mechanisms are inadequate to reduce these threats. Overall, impacts from increasing threats, operating singly or in combination, place the species at risk of extinction. White-nose syndrome (WNS) is not currently known to be a threat to this species, but it is possible that disease will have a greater impact on the Florida bonneted bat in the future. The extent to which predation (e.g., by non-native species) may be impacting the Florida bonneted bat is unknown, but given the species' apparent small population size and overall vulnerability, it is reasonable to assume that predation is a potential threat, which may increase in the future. Further study is required to more fully assess the risk that pesticides and contaminants pose to the Florida bonneted bat. (NatureServe, 2015)

### ***Recovery***

#### **Recovery Actions:**

- Determine the distribution and status of the Florida bonneted bat; Protect and enhance known existing populations through regulatory mechanisms, education, and outreach; Minimize disturbance or mortality to the Florida bonneted bat through communication to utility and construction workers, nuisance animal companies, and the public; Continue to collaborate with state and federal agencies, other institutions and organizations to conduct research on the ecology and life history of the Florida bonneted bat, including foraging and roosting habitat associations, food habits, predators, parasites and disease, and the potential effects of pesticides; Develop standard methods to monitor the status of the Florida bonneted bat that allow for spatial and temporal comparisons; Develop a mechanism for sharing information; Protect and manage Florida bonneted bat habitat through land acquisitions and conservation easements, and by developing management plans with partners; Examine ecological processes in Florida bonneted bat habitat including fire, hydrology, and other factors that increase the likelihood of supporting foraging and roosting (USFWS, 2018).

#### ***Conservation Measures and Best Management Practices:***

- Management Recommendations 1. Discontinue pesticide spraying for any area known to be used by species (USFWS, 2016).
- 2. Educate public about bats, especially this very rare species (USFWS, 2016).
- Conducting acoustic surveys within the species' historic range to better understand movements, threats, and delineate range; Locating natural roosts and identifying factors influencing roost selection; Evaluating impacts to individuals living in urban areas; Using various techniques to accurately and safely monitor extant populations (USFWS, 2018).

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## SPECIES ACCOUNT: *Glaucomys sabrinus coloratus* (Carolina northern flying squirrel)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; 7/1/1985; Southeast Region (R4) (USFWS, 2015)

### **Physical Description**

A small squirrel. The length is 37 cm. (NatureServe, 2015). They possess a long, broad, flattened tail (80% of head and body length), prominent eyes, and dense, silky fur. The distinctive patagia (folds of skin between the wrists and ankles) are fully furred and supported by slender cartilages extending from the wrist bones; these plus the broad tail create a large gliding surface area and are the structural basis for the squirrel's characteristic gliding locomotion (Thorington and Heaney, 1981). Adults are dorsally gray with a brownish, tan, or reddish wash, and grayish white or buffy white ventrally (USFWS, 1990). It is distinguished from the West Virginia (WV) northern flying squirrel (*Glaucomys sabrinus fuscus*) by its larger size, longer tail length, and brighter coloration (Handley 1980) (USFWS, 2013).

### **Taxonomy**

Closely related to subspecies *coloratus*. The population of *Glaucomys sabrinus* in the Mount Rogers and Whitetop area of Virginia may be intergrades between subspecies *coloratus* and *fuscus* (Handley 1991). One of 25 currently recognized subspecies (NatureServe, 2015).

### **Historical Range**

The Carolina northern flying squirrel is a Pleistocene relict in the Southern Appalachians that is confirmed to a handful of isolated high-elevation peaks and ridges that support spruce-fir and northern hardwood forests. Fossil remains indicate a much larger range during the Pleistocene and early Holocene (Service 1990). At the time it was added to the Federal List of Endangered and Threatened Wildlife and Plants, the Carolina northern flying squirrel was known from only four areas--Roan Mountain (TN and NC), Great Smoky Mountains National Park (TN and NC), Mt. Mitchell (NC), and Whitetop Mountain (southwestern VA) (USFWS, 2013).

### **Current Range**

Occurs in the Southern Appalachian Mountains, Tennessee and North Carolina as well as isolated localities in Virginia. Subspecies identity is uncertain in the Mount Rogers and Whitetop area, Virginia; this may be an area of intergradation between subspecies *coloratus* and *fuscus* (Handley 1991) (NatureServe, 2015).

### **Distinct Population Segments Defined**

No

### **Critical Habitat Designated**

No;

### ***Life History***

### **Feeding Narrative**

Adult: Diet consists of both plant and animal material. Eats insects, nuts, lichens, fungi, buds, seeds, fruit during season; apparently can subsist on lichens and fungi for extended periods. Spends considerable time foraging on ground. It is active at night and throughout the year. Peak activity occurs from sunset to 2 hours after and 1 hour before sunrise (NatureServe, 2015).

### **Reproduction Narrative**

Adult: Gestation lasts 37 - 42 days. Apparently produces 1 litter/year, in spring or summer. Young are weaned at about 2 months. Sexually mature within one year. Highly social, especially in winter when nests may be shared. Apparently lives in family groups of adults and juveniles. (NatureServe, 2015). The northern flying squirrel can be relatively long-lived (4 to 7 years) and has a low reproductive rate (generally a single litter annually, with two to five young) (Weigl et al. 1999, Weigl 2007, Kelly 2008) (USFWS, 2013).

### **Geographic or Habitat Restraints or Barriers**

Adult: Typically occurs > 4,500 ft. elevation (USFWS, 2013)

### **Environmental Specificity**

Adult: Narrow (inferred from USFWS, 2013)

### **Habitat Narrative**

Adult: Prefers coniferous and mixed forest, but will utilize deciduous woods; riparian woods; optimal conditions: cool, moist, mature forest with abundant standing and down snags. Occupies tree cavities, leaf nests, and underground burrows. See Payne et al. (1989) for specific habitat characteristics in the southern Appalachians. Prefers cavities in mature trees as den sites. Small outside twig nests sometimes used for den sites. Will use nest box (NatureServe, 2015). The Carolina northern flying squirrel occupies high-elevation forests and is most often encountered at the ecotone between northern hardwood and spruce or spruce/fir forests. Habitat features important to the Carolina northern flying squirrel include old trees and abundant woody debris (habitat characteristics associated with old-growth forests), cool and moist conditions, substantial ground cover, and some degree of openness under the canopy (Weigl et al. 1999). This habitat exists at high elevations, typically above 1,372 meters (4,500 feet) and is most often found on north-facing mountainsides and drainages (USFWS, 2013).

### ***Dispersal/Migration***

### **Motility/Mobility**

Adult: Moderate (inferred from NatureServe, 2015)

### **Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory (NatureServe, 2015)

### **Dispersal**

Adult: Low (inferred from NatureServe, 2015); moderate (inferred from USFWS, 2013)

### **Dispersal/Migration Narrative**

Adult: This species is non-migratory. Summer home range was estimated at 2-3 ha in North Carolina (Austin et al., no date) (NatureServe, 2015). A study in the Unicoi Mountains revealed larger home range sizes (3 .3 to 51.4 hectares, with an average of 15.9 hectares). It is known

that squirrels are capable of going on long forays (especially males) of over 1.5 kilometers (Weigl et al. 1999, Weigl et al. 2002, Hughes 2006) (USFWS, 2013).

### ***Population Information and Trends***

#### **Population Trends:**

Not available

#### **Species Trends:**

Unknown (USFWS, 2013)

#### **Resiliency:**

Moderate (inferred from NatureServe, 2015; see current range/distribution)

#### **Population Size:**

Unknown (USFWS, 2013)

#### **Population Narrative:**

The species status is stable given continued presence at sites. Because of variability in detections, it is difficult to establish current population levels or trends for the NC population of the squirrel. Considerably less is known about northern flying squirrel populations in TN and VA (USFWS, 2013).

### ***Threats and Stressors***

**Stressor:** Habitat destruction and fragmentation (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** There has been an increase in residential development in the Southern Appalachians, and the human population is expected to continue to grow. Actual and potential loss and fragmentation of habitat to residential development threatens two GRAs (the Plott Balsams and Long Hope Valley). The loss of habitat in the Plott Balsams would break up connectivity with neighboring GRAs, and the loss of habitat in Long Hope Valley could result in the loss of an entire recovery area (Kelly 2008). In addition to habitat loss due to residential development, activities to accommodate an increasing demand for recreation at high elevations are also a significant threat (e.g., construction of parking areas and roads; vista management). For example, the construction of a high-elevation highway (Cherochala Skyway) in the Unicoi Mountains GRA resulted in a barrier to squirrel movement. This barrier has effectively cut the population into two isolated segments (Weigl et al. 2002, Hughes 2006). Forest pests and diseases are significant indirect threats to the existence and recovery of the Carolina northern flying squirrel, with the balsam woolly adelgid, hemlock woolly adelgid, and beech bark disease threatening the habitat this species occupies. The loss of fir and hemlock trees and declines in spruce trees may result in serious degradation of squirrel habitat since conifers are an important component of their habitat (USFWS, 2013).

**Stressor:** Pet trade (USFWS, 2013)

**Exposure:**

**Response:****Consequence:**

**Narrative:** Flying squirrels are highly desirable as pets; thus, collection for the pet trade is at least a potential threat (USFWS, 2013).

**Stressor:** Strongyloides robustus parasite (USFWS, 2013)

**Exposure:****Response:****Consequence:**

**Narrative:** The internal nematode parasite Strongyloides robustus has been identified as a potential problem for Carolina northern flying squirrels (Weigl et al. 1999). While it apparently does not have significant adverse effects on this species, it can be lethal or seriously debilitating to Carolina northern flying squirrels (Weigl 1968, Weigl et al. 1999). The prevalence of this parasite in Carolina northern flying squirrel populations has increased in recent years, and the role it plays in the viability of squirrel populations is poorly understood (Weigl et al. 1999, Weigl 2007) (USFWS, 2013).

**Stressor:** Interspecific competition (USFWS, 2013)

**Exposure:****Response:****Consequence:**

**Narrative:** Southern flying squirrels not only act as potential vectors for disease, but they also are generally more aggressive than the northern squirrels and have the potential to displace them (Weigl 1978, Weigl et al. 1999). Differences in habitat preferences, diets, and climatic tolerances have largely kept these species separate in the past (Weigl 2007), but this could change in human-altered landscapes. Southern flying squirrels are expanding into higher elevations in more southern latitudes (Odom et al. 2001, Weigl et al. 1999 in Smith 2007). While direct interspecific competition has not been widely reported, similarities in behavior and shared vital resources (e.g., tree cavities) coupled with expanding oak and hickory forests and warming climate could lead to more interactions (Weigl et al. 1999 in Smith 2007, Weigl 2007) (USFWS, 2013).

**Stressor:** Climate change (USFWS, 2013)

**Exposure:****Response:****Consequence:**

**Narrative:** Climate change could, if it results in appreciable increases in temperatures, threaten the Carolina northern flying squirrel. The squirrel is restricted to small areas with suitable habitat in the Southern Appalachians. These areas form small islands at high elevations and have reduced connectivity between them. If temperatures in the Southern Appalachians increase and precipitation decreases, it is anticipated that the areal extent of boreal forests will decrease. Warming at high elevations could allow for further invasion by southern flying squirrels and increase the viability of parasites such as Strongyloides as mentioned previously. Further, climate change may increase the susceptibility of associated forests to exotic and native forest pests and pathogens (USFWS, 2013).

**Stressor:** Pollution (USFWS, 2013)

**Exposure:****Response:**

**Consequence:**

**Narrative:** Pollution (in the form of acid rain and inputs of heavy metals) adversely impacts forest health and productivity, including that of the red spruce (Kelly 2008). Furthermore, high levels of mercury, lead, and other heavy metals found in the soils and fungi within the squirrel's habitat may threaten this mycophagous squirrel. Fungi, which form an important component of the flying squirrel's diet, can bioaccumulate heavy metals; this potential threat needs further evaluation (USFWS, 2013).

**Recovery****Reclassification Criteria:**

1. Squirrel populations are stable or expanding (based upon biennial sampling over a 10-year period) in a minimum of 80 percent of all Geographic Recovery Areas (GRAs) (USFWS, 2013).
2. Sufficient ecological data and timber management data have been accumulated to assure future protection and management (USFWS, 2013).
3. GRAs are managed in perpetuity to ensure: (a) Sufficient habitat for population maintenance/expansion and (b) habitat corridors, where appropriate elevations exist, to permit migration among GRAs (USFWS, 2013).

**Delisting Criteria:**

In addition to the downlisting criteria, the existence of the high-elevation forests on which the squirrels depend is not itself threatened by introduced pests, such as the balsam wooly adelgid, or by environmental pollutants, such as acid precipitation or toxic substance contamination (USFWS, 2013).

**Recovery Actions:**

- Survey for new populations and monitor known populations (USFWS, 1990).
- Study habitat requirements (USFWS, 1990).
- Study diet, interactions with other squirrels and genetics (USFWS, 1990).
- Study effects of various land use practices (mining, logging, recreation) (USFWS, 1990).
- Ensure implementation of appropriate habitat management guidelines, based on results of 1 -3. (This would include periodic monitoring, even following de-listing.) (USFWS, 1990).

**Conservation Measures and Best Management Practices:**

- Develop and institute improved survey methods that reliably assess the status, population levels, and population trends of the species throughout its range (USFWS, 2013).
- Determine the distribution and status of populations of northern flying squirrels in TN and southwestern VA using reliable and more intensive surveys of the areas that are currently known to support the species as well as additional sites that appear to provide suitable habitat (USFWS, 2013).
- Determine the taxonomic status of the southwestern VA population of the northern flying squirrel (USFWS, 2013).
- Develop predictive models of habitat utilized by the Carolina northern flying squirrel throughout its range in order to provide managers with an additional tool to manage and protect the species as well as provide insight into additional areas that may support the squirrel (USFWS, 2013).

- Restore spruce where appropriate, and use spruce restoration as a tool to create and maintain corridors to connect GRAs. Concurrently, remaining stands of northern hardwood need to be protected (USFWS, 2013).
- Establish a group of federal and state biologists and land managers familiar with the species and its requirements who will coordinate activities related to the assessment, protection, and management of the Carolina northern flying squirrel. This group could, if properly constituted, form the core of a team to develop a revised recovery plan (USFWS, 2013).
- Revise the recovery plan to reflect current knowledge of the Carolina northern flying squirrel, including objective and measurable recovery criteria and updated actions needed to recover the species (USFWS, 2013).

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## **SPECIES ACCOUNT: *Herpailurus (=Felis) yagouaroundi cacomitli* (Gulf Coast jaguarundi)**

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### ***Species Taxonomic and Listing Information***

**Listing Status:** Endangered; 06/14/1976; Southwest Region (Region 2) (USFWS, 2016)

### **Physical Description**

The jaguarundi is a small cat, slightly larger than a house cat (*Felis catus*). With a slender build, long neck, short legs, small and flattened head, and long tail, it resembles a weasel (*Mustela* sp.) more than other felines (Tewes and Schmidly 1987, Oliveira 1998). Jaguarundis are not spotted and have two color phases – blackish to brownish gray or reddish yellow to chestnut. The Service now knows that red and gray kittens can be found in the same litter and the color phases are the same species (Goodwyn 1970). The long tails range in length from 11 to 24 in [28 to 61 cm] and standing height at the shoulder is typically 11 in (28 cm); total body length, including tail, of adult males is 42 in (107 cm) (TPWD 2011). The standing height at the shoulder of the Gulf Coast subspecies is typically slightly smaller at 10 in (26 cm) (Caso and Tewes in prep). Weights range from 3.8 to 9 kg [8.4 to 19.8 lbs] with an average of 6 kg (13.2 lb) (Guggisberg 1985, Silva-Pereira et al. 2011), but the Gulf Coast subspecies weighs 6.6 kg (14.5 lbs) at most (Caso and Tewes in prep). (USFWS, 2013)

### **Taxonomy**

The jaguarundi was originally included in the genus *Felis* and the Gulf Coast jaguarundi was originally listed under the ESA as *Felis yagouaroundi cacomitli* in 1976. Later, genus classification was changed from *Felis* to *Herpailurus* (Wozencraft 1993) and this widely accepted change was subsequently made to the ESA listing. Thus, this subspecies is currently listed under the ESA as *Herpailurus (=Felis) yagouaroundi cacomitli*. However, more recent genetic work assigns the jaguarundi to the genus *Puma* (Johnson and O'Brien 1997, Johnson et al. 2006) and this has become the generally accepted nomenclature (Wilson and Reeder 2005). Therefore, in keeping with this current information, we refer to the Gulf Coast jaguarundi subspecies as *Puma yagouaroundi cacomitli* throughout this recovery plan and we officially accept the new scientific name of the jaguarundi as *Puma yagouaroundi*. (USFWS, 2013)

### **Historical Range**

The Gulf Coast jaguarundi's historical range is from the Lower Rio Grande Valley in southern Texas into the eastern portion of Mexico in the States of Coahuila, Nuevo Leon, Tamaulipas, San Luis Potosi, and Veracruz. In Texas, jaguarundis historically were limited to the southern portion of the state including Cameron, Hidalgo, Willacy, and Starr counties (Bailey 1905, Davis 1974). In a boundary survey of the U.S. and Mexico, Baird (1859) notes that evidence of jaguarundi existing along the Rio Grande. (USFWS, 2013)

### **Current Range**

The last confirmed sighting of this subspecies within the U.S. was in April 1986, when a roadkilled specimen was collected two miles east of Brownsville, TX and positively identified by the Smithsonian National Museum of Natural History as a jaguarundi. (USFWS, 2013)

### **Distinct Population Segments Defined**

No

**Critical Habitat Designated**

Yes;

***Life History*****Feeding Narrative**

Juvenile: Jaguarundis prey mainly on birds, small mammals, and reptiles, with a mean prey mass of 380 grams (0.84 pounds) (Guggisberg 1985, Caso et al. 2008). The jaguarundi is the only cat in northeastern Mexico which is primarily active during the day, whereas the other cats, such as ocelot, are primarily nocturnal. In his research in Tamaulipas, Mexico, Caso (1994) reported the activity pattern of jaguarundis to be 14.4 percent nocturnal and 85.6 percent diurnal and Sanderson (2012a) noted that in Suriname in areas where no people were present, jaguarundis were strictly diurnal.

Adult: Jaguarundis prey mainly on birds, small mammals, and reptiles, with a mean prey mass of 380 grams (0.84 pounds) (Guggisberg 1985, Caso et al. 2008). The jaguarundi is the only cat in northeastern Mexico which is primarily active during the day, whereas the other cats, such as ocelot, are primarily nocturnal. In his research in Tamaulipas, Mexico, Caso (1994) reported the activity pattern of jaguarundis to be 14.4 percent nocturnal and 85.6 percent diurnal and Sanderson (2012a) noted that in Suriname in areas where no people were present, jaguarundis were strictly diurnal. Oliveira et al. (2010), in their synthesis of available data on ocelots (*Leopardus pardalis*) and other small cats in the lowlands of the Neotropics, along with data from a study of ocelots in Brazil, found that ocelots and other small cats, including jaguarundi, coexist within the same habitats. (USFWS, 2013)

**Reproduction Narrative**

Adult: Information on life history aspects of jaguarundi in the wild, including age of sexual maturity, minimum and maximum breeding age, and mating behavior, is limited (Tewes and Schmidly 1987, Caso 1994, TPWD 2012a). In a study of captive felids, Mellen (1993) reported the estrous cycle of the jaguarundi lasted  $53.6 \pm 2.4$  days ( $n = 8$ ). Hulley (1976) reported that a captive jaguarundi exhibited her first estrus at about two years of age and every six months thereafter. Gestation period of captive animals was 72 to 75 days. Reported litter size is one to four young, with a mean of 1.9 (Oliveira 1998). Jaguarundis may have two litters per year (Guggisberg 1985). Jaguarundis are solitary, except during mating season or when a female is raising kittens. The mating season in Mexico is November and December, while in the tropics it is year-round (Oliveira 1998, TPWD 2012a). (USFWS, 2013)

**Environmental Specificity**

Adult: Low (inferred from USFWS, 2013)

**Tolerance Ranges/Thresholds**

Adult: Moderate (USFWS, 2013)

**Habitat Narrative**

Adult: The jaguarundi is a lowland species, inhabiting forest and bush (Guggisberg 1985). The cacomitli subspecies is found in the Tamaulipan Biotic Province of northeast Mexico and south

Texas (Caso 1994). Within Mexico it occurs in the eastern lowlands and has not been recorded in the Central Highlands (Tewes and Schmidly 1987). In southern Texas, jaguarundis used dense thorny shrublands. Typical habitat consists of mixed thornscrub species which include the following: brasil (*Condalia hookeri*), desert yaupon (*Schaefferia cuneifolia*), wolfberry (*Lycium berlandieri*), lotebush (*Ziziphus obtusifolia*), amargosa (*Castela erecta*), white-brush (*Aloysia gratissima*), catclaw (*Acacia greggii*), blackbrush (*Acacia rigidula*), lantana (*Lantana aachyranthifolia*), guayacan (*Guajacum angustifolium*), cenizo (*Leucophyllum frutescens*), elbowbush (*Forestiera angustifolia*), and Texas persimmon (*Diospyros texana*). Trees that may be interspersed within the thornscrub include mesquite (*Prosopis* sp.), live oak (*Quercus* sp.), ebony (*Ebenopsis ebano*), and hackberry (*Celtis laevigata*). River and creek riparian habitat are also sometimes used (TPWD 2012a). Jaguarundis will use bunchgrass pastures if dense brush or woody cover is nearby. Consequently, patchworks of bunchgrass pastures with tracts of dense brush used by ocelots will also be used by jaguarundis (Caso 1994, Tewes and Caso 2011). (USFWS, 2013)

***Dispersal/Migration*****Motility/Mobility**

Adult: High (NatureServe, 2013)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory

**Dispersal**

Adult: High (NatureServe, 2015)

**Dispersal/Migration Narrative**

Adult: Travels widely in a huge home range (Emmons and Feer 1990). (NatureServe, 2015)

***Population Information and Trends*****Population Trends:**

Not available.

**Resiliency:**

Low (inferred from USFWS, 2013)

**Number of Populations:**

Unknown

**Population Size:**

Unknown

**Population Narrative:**

The last confirmed sighting of this subspecies within the U.S. was in April 1986, when a roadkilled specimen was collected two miles east of Brownsville, TX and positively identified by the Smithsonian National Museum of Natural History as a jaguarundi. Numerous unconfirmed sightings have been reported since then, some with unidentifiable photographs, but no reports

have been confirmed as jaguarundi since 1986, despite significant camera-trapping effort and live-trapping effort. Texas population probably consists of only a few individuals (USFWS, 2013; NatureServe, 2015)

### ***Threats and Stressors***

**Stressor:** Habitat loss (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The main threats to the jaguarundi throughout its range are habitat loss, degradation, and fragmentation. In the U.S., the habitat historically used by the Gulf Coast jaguarundi was once extensive throughout the Lower Rio Grande Valley (LRGV) but has been converted to agriculture and urban development (TPWD 2012a). In the LRGV of Texas, it has been estimated that over 95 percent of the dense thornscrub habitat that supported the Gulf Coast jaguarundi has been altered for agricultural and urban development (Jahrsdoerfer and Leslie 1988). In Cameron County, 91 percent of native woodlands were lost during the mid-1900s, primarily for agricultural uses (Tremblay et al. 2005). Currently, rapid population growth in the region is causing agricultural land to be converted to more urban development resulting in land and habitat fragmentation (Wilkins et al. 2000). The human population in the LRGV increased 39.8% from 1990 to 2000, compared to an increase of 22.8% in Texas and 13.2% in the U.S. during the same period (Murdock et al. 2002). Largely because of its relatively high birth rate, the LRGV population is expected to increase by more than 1 million, from 1.5 million in 1995 to 2.6 million in 2020 (Texas Comptroller 2012). (USFWS, 2013)

**Stressor:** Roads - fragmentation and mortality (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Roads may have two potential impacts on Gulf Coast jaguarundi populations. First, collisions with motor vehicles in Texas and in Mexico may be a source of mortality. While the Service only has one documented case of a jaguarundi being killed by a motor vehicle collision, collisions with motor vehicles are the leading cause of known mortality for ocelots in Texas (USFWS 2010). If jaguarundi populations were to expand into or be reintroduced to southern Texas, road mortality may be an issue. While some underpasses and culverts have been installed for ocelots in Texas, more are needed and correct size, design and placement is critical for them to be used by ocelots as travel corridors (USFWS 2010). If jaguarundi populations were once again found in Texas, these underpasses and culverts may also be useful in facilitating jaguarundi recovery. Second, roads can fragment habitat and decrease the probability of successful dispersal between patches of suitable habitat, thus increasing demographic and genetic isolation of populations. In their study on the effects of highway and associated wildlife mitigation features on bobcats in southern Texas, Cain et al. (2003) stated that projects to reduce the impacts of roads on wildlife should consider which impact, road mortality or habitat fragmentation, is likely to be the most detrimental to the population and ensure that efforts to mitigate one impact do not increase the other. In addition, to the extent that jaguarundis might avoid areas of high road density, some otherwise suitable habitats may not be occupied by jaguarundis. Future recovery efforts would benefit from information on how jaguarundis locate home ranges relative to roads, or use culverts or underpasses to negotiate roads. (USFWS, 2013)

**Stressor:** Disease (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** It is reasonable to suspect that diseases that affect free-ranging domestic and feral cats could be a source of mortality and reduced fitness for jaguarundi (e.g., feline leukemia, feline HIV, rabies, etc.). However, the Service has no specific information on diseases or predation of jaguarundi that indicates these are threats to the subspecies. (USFWS, 2013)

**Stressor:** Competition with other small cats (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Oliveira et al. (2010), in their synthesis of available data on ocelots (*Leopardus pardalis*) and other small cats in the lowlands of the Neotropics, along with data from a study of ocelots in Brazil, found that ocelots and other small cats, including jaguarundi, coexist within the same habitats. Sanderson (2012b), in his camera trap study of wildlife in forests of Suriname, found that ocelot and jaguarundi active periods overlapped a small percentage, similar to the patterns in Caso's study. In their study of six wild cat species in Argentina, Di Bitetti et al. (2010) also found that four species (puma [*Puma concolor*], ocelot, oncilla [*Leopardus tigrinus*], and jaguar [*Panthera onca*]) alternated their peaks of activity in relation to the relative order of their body weights, i.e., the two larger species of cats (puma and jaguar) did not have the same peak activity times and the two smaller species of cats (ocelot and oncilla) also did not have the same peak activity times. Other biologists (USFWS 2012) have also theorized that bobcats could play a role in limiting jaguarundi populations in the northern part of their range. In southern Texas, bobcats are fairly common. (USFWS, 2013)

**Stressor:** Border issues (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Increased border monitoring associated with illegal immigration, and homeland security, may impact future jaguarundi recovery efforts. Borderland factors that could impact Gulf Coast jaguarundis include urbanization (e.g. brush clearing for buildings, sewage dumped into the Rio Grande and its tributaries, and road construction and maintenance), water development (e.g. brush clearing, channeling, draining), agriculture (e.g. brush clearing, pesticide run-off), U.S. Border Patrol Operations (e.g. lighting; road construction and maintenance; tower construction and maintenance; brush clearing; human activity, including on and off-road vehicular activity) (Jahrsdorfer and Leslie 1988), and the construction of fences along the border (Defenders of Wildlife 2006 and 2012a,b; Gaskill 2011; McCorkle 2011). Also, there are 11 existing international bridges plus an international dam, and four more bridges under consideration within Cameron, Hidalgo, and Starr Counties in Texas that may act as east-west barriers for Gulf Coast jaguarundi movement. Barriers such as these can affect regional biodiversity, including jaguarundis, by destroying, fragmenting, and degrading habitat; disrupting the social structure of wildlife populations; reducing access to resources and habitats; and isolating and fragmenting animal populations (List 2007). The implementation of bridge projects in the region should seek opportunities to minimize potential wildlife impacts. (USFWS, 2013)

**Stressor:** Hunting (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Hunting jaguarundi is not legal in Mexico or in the U.S. However, jaguarundis may be subject to low intensity hunting pressure around settlements (Nowell and Jackson 1996). In a study of hunting practices in the tropical forests of Calakmul, Mexico, researchers found that while jaguarundi were present in their study area, and may have been occasionally hunted, they were never recorded as hunted (Escamilla et al 2000). Inskip and Zimmerman (2009), reviewed and analyzed conflicts between people and felids worldwide and found a low level of conflict for jaguarundi (all subspecies), meaning there was some livestock depredation by jaguarundi, with no risk to humans, but with some retaliatory killing. Retaliatory killing typically consists of local farmers killing jaguarundi because they have killed poultry (Caso et al. 2008). In a study of human-felid interactions in three mestizo communities in Chiapas, Mexico, Garcia-Alaniz et al. (2010) found that jaguarundi appeared to be the most common wild felid (out of five species) preying on domestic animals. This may reflect a greater tolerance to human disturbance or just a greater abundance than the other four felid species in this study area. While jaguarundis are viewed as a nuisance species and are hunted when they cause damage to domestic livestock, this study did not investigate the correlation between hunting practices and the actual population abundance of felids. The authors did believe that small felids represent significant predators of domestic livestock and their populations are affected negatively by hunting. Overall, based on all of the available information, while localized hunting of jaguarundi takes place, it is not a major threat to the Gulf Coast jaguarundi. (USFWS, 2013)

**Stressor:** Climate change (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Our analyses under the Endangered Species Act include consideration of ongoing and projected changes in climate. The terms “climate” and “climate change” are defined by the Intergovernmental Panel on Climate Change (IPCC). The term “climate” refers to the mean and variability of different types of weather conditions over time, with 30 years being a typical period for such measurements, although shorter or longer periods also may be used (IPCC 2007a). The term “climate change” thus refers to a change in the mean or variability of one or more measures of climate (e.g., temperature or precipitation) that persists for an extended period, typically decades or longer, whether the change is due to natural variability, human activity, or both (IPCC 2007a). The Service does not know whether the changes that have already occurred have affected Gulf Coast jaguarundi populations or distribution, nor can we predict how the species will be affected by the type and degree of climate changes forecasted by a range of models, particularly since we have no population estimates or trend information for this subspecies. But, ongoing and future changes in climate have the potential to affect the jaguarundi within the next 50 to 100 years. Stochastic threats such as drought and wildfires in jaguarundi habitat may make this species more vulnerable. Likewise, changes in prey populations due to climate change could influence jaguarundi distribution. Monitoring of habitat and populations will be needed to address the potential threat of climate change. Therefore, monitoring the species and its habitat is necessary and the Service will adapt our recovery and management strategies as needed to address the changing conditions. (USFWS, 2013)

**Recovery****Reclassification Criteria:**

1. We have sufficient scientific information on the Gulf Coast jaguarundi to show that 3 or more separate populations with a combined total of at least 250 individuals rangewide are stable or increasing for at least 10 years and there is sufficient interchange between those populations to maintain genetic variability. (USFWS, 2013)

2. Threats from habitat loss, degradation, and fragmentation, have been reduced such that the Gulf Coast jaguarundi is no longer in danger of extinction. Total protected habitat area should include at least 2,200 km<sup>2</sup> (850 mi<sup>2</sup>) of suitable habitat to support jaguarundi populations for the foreseeable future, and potential corridors and mechanisms must be identified to restore habitat connectivity between populations if necessary. Populations can include those found in Mexico, any newly discovered populations in southern Texas, a population that re-establishes in southern Texas through natural expansion, or a population established in southern Texas through translocation or reintroduction. (USFWS, 2013)

**Delisting Criteria:**

1. We have sufficient scientific information on the Gulf Coast jaguarundi to show that 3 or more separate populations with a combined total of at least 500 individuals rangewide are stable or increasing for at least 20 years and there is sufficient interchange between those populations to maintain genetic variability. (USFWS, 2013)

2. Threats from habitat loss, degradation, and fragmentation, have been reduced such that the Gulf Coast jaguarundi is no longer in danger of extinction. Total protected habitat area should include at least 4,400 km<sup>2</sup> (1,700 mi<sup>2</sup>) of suitable habitat to support jaguarundi populations for the foreseeable future, and potential corridors and mechanisms must be identified to restore habitat connectivity between populations if necessary. Populations can include those found in Mexico; any newly discovered populations in southern Texas; a population that re-establishes in southern Texas through natural expansion; or a population established in southern Texas through translocation or reintroduction. (USFWS, 2013)

**Recovery Actions:**

- Assess, protect, and enhance potential Gulf Coast jaguarundi habitat and connectivity in the U.S. (USFWS, 2013)
- Develop more effective survey techniques for jaguarundis. (USFWS, 2013)
- Support efforts to ascertain the status, better understand ecological and conservation needs, and promote conservation of the Gulf Coast jaguarundi and its habitats in Mexico. (USFWS, 2013)
- Reduce the effects of human population growth and development on potential Gulf Coast jaguarundi habitat in the U.S. (USFWS, 2013)
- Assure the long-term success of Gulf Coast jaguarundi conservation through partnerships, landowner incentives, community involvement, application of regulations, and public education and outreach. (USFWS, 2013)
- Reduce the risk of jaguarundi mortality from vehicle collisions. (USFWS, 2013)

- Determine the relationship among bobcats, coyotes, ocelots and jaguarundis. (USFWS, 2013)
- Practice adaptive management in which recovery is monitored and recovery tasks are revised by FWS as new information becomes available. (USFWS, 2013)

***Conservation Measures and Best Management Practices:***

- Conservation measures are not available.

**References**

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## **SPECIES ACCOUNT: *Lasiurus cinereus semotus* (Hawaiian hoary bat)**

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### ***Species Taxonomic and Listing Information***

**Listing Status:** Endangered; 10/13/1970; Pacific Region (R1) (USFWS, 2016)

### **Physical Description**

The Hawaiian hoary bat is medium-sized (0.5 to 0.8 ounces), with a wingspan of 10.5 to 13.5 inches, and is nocturnal, insectivorous with thick, rounded ears and a furry tail. “Hoary” refers to the white-tinged, frosty appearance of the bat’s grayish brown or reddish brown fur. Although females are slightly larger than males, forearm lengths are similar in both genders. These bats are not colonial, and roost solitarily in tree foliage (Service 1998, pp. 8-10).

### **Taxonomy**

The Hawaiian hoary bat is classified under the Family Vespertilionidae of the Suborder Microchiroptera, and is one of three recognized hoary bat subspecies. The other two subspecies are *Lasiurus cinereus cinereus*, one of the most common and widespread bats in North America, and *Lasiurus cinereus vilosissimus*, which occurs in South America and the Galapagos (Shump and Shump 1982, pp. 1-5). Morphologically, the Hawaiian hoary bat may have diverged significantly from the North American form, as Hawaiian hoary bats are about 45 percent smaller. Nonetheless, preliminary genetic analysis indicates the Hawaiian hoary bat may be derived from the North American hoary bat. The low degree of genetic divergence, however, suggests subspecies classification may be appropriate (Service 1998, pp. 8-9).

### **Historical Range**

See current range/distribution

### **Current Range**

The Hawaiian hoary bat is endemic to the State of Hawaii where it is the only existing, native terrestrial mammal. The Hawaiian hoary bat is known to reside on Hawaii, Maui, Oahu, Lanai, Molokai and Kauai, with the largest populations likely on Hawaii and Kauai. There are no population estimates for the Hawaiian hoary bat and few historical or current records. Unsubstantiated population estimates across the State have ranged from hundreds to a few thousand individuals (Service 1998, p. 14). Data are limited because no feasible method currently exists for surveying the abundance and distribution of solitary, tree-roosting bats. The Hawaiian hoary bat’s distribution may be broader than indicated by the current limited information resulting from localized search efforts (Service 1998, p. 14). Hawaiian hoary bats have been observed year-round in a wide variety of habitats and elevations below 7,500 feet, and a few sightings from limited surveys have been reported as high as 13,199 feet. Hawaiian hoary bats have been detected in both wet and dry areas of Hawaii but seem to be more abundant on the drier leeward side (Jacobs 1994, p. 199) and generally less abundant in wet areas (Kepler and Scott 1990, p. 62). Only three researchers have examined spatial and temporal variation in occurrence patterns of bats in Hawaii, with conflicting conclusions about possible altitudinal or regional migration (Jacobs 1994, pp. 193-200; Menard 2001, pp. 1-149; Tomich 1986, pp. 1-30).

### **Critical Habitat Designated**

No;

## ***Life History***

### **Reproduction Narrative**

Adult: As with other life history parameters, little is known about the breeding biology of Hawaiian hoary bats. Females of most temperate, autumn-breeding, insectivorous bat species become pregnant in the spring by delayed ovulation and fertilization, and young are cared for exclusively by the female. The breeding cycle of the Hawaiian hoary bat on the island of Hawaii consists of pregnancy (April to June), with pups born in May or June; lactation (June through early August and possibly to September); post-lactation, after pups have fledged (September to December); and pre-pregnancy (January to March) (Menard 2001, p. 35). Like North American hoary bats, Hawaiian hoary bat females are believed to give birth to two young at a time. North American hoary bat pups cling to the mother at the roost tree during the day, where she leaves them hanging on a twig while she forages at night (Shump and Shump 1982, p. 3), and Hawaiian hoary bats are presumed to behave similarly. Female North American hoary bats adjust their foraging behavior to meet the increasing energy demands of pregnancy and lactation (Barclay 1989, pp. 31-37). Because newborn bats cannot thermoregulate very well in tree-foliage roosts, the mother's foraging activity may be constrained by the need to roost periodically with her young to keep them warm. Thus, foraging behavior changes with reproductive condition, and females with non-volant young may forage at different times of night and perhaps in different habitats than other bats. Preliminary evidence indicates that pregnant and lactating female Hawaiian hoary bats on Hawaii may prefer roosting in lowland areas rather than in the cooler highlands, perhaps because the warmer lowland environment promotes faster juvenile growth (or, alternatively, because insect food sources may be more readily available) (Menard 2001, pp. 52-105).

### **Habitat Narrative**

Adult: A comprehensive life history assessment for the Hawaiian hoary bat is lacking. Furthermore, the existing information on population status and habitat ecology is often conflicting. Hawaiian hoary bats roost in a variety of tree species, both native and non-native, during the day and forage in a wide range of habitat types during the night (Service 1998, pp. 12-13). There is no information on the Hawaiian hoary bat's average life span, age at first reproduction, and survivorship, or on how age and reproductive condition affect its food habits, habitat selection, home range size, and movement patterns. A few studies have documented Hawaiian hoary bats in a wide range of locations and habitat types on the island of Hawaii. Bats observed along 611 miles of forest bird survey transects and incidentally elsewhere on Hawaii during 1976-1983, at elevations from sea level to 10,007 feet, were more frequently associated with non-native vegetation (64 percent), such as tall eucalyptus and other exotic plants, than with native vegetation (19 percent) (Kepler and Scott 1990, p. 61). Visual observations and echolocation detections at 22 sites in southeast Hawaii, however, found no significant differences in bat activity among native or non-native vegetation types (Reynolds et al. 1998, pp. 153-157). In addition, 57 percent of all bat activity was noted at open sites, forest edges, lava flows, volcanic pit craters, residential and agricultural clearings, and roads. Foraging bats at 14 survey sites over a range of altitudes were more frequently associated with native vegetation (44 percent) than non-native (16 percent) or mixed (9 percent) vegetation (Jacobs 1993, p. 22). Bats were detected most often in native mesic koa-ohia forest vegetation at 13 sites in, and adjacent to, Hakalau Forest National Wildlife Refuge (Cabrera 1996, p. 238). All reports of bat occurrences may be biased to varying degrees by sampling efforts concentrated along roads and

forest edges. Roosting habitat for the Hawaiian hoary bat is sparsely documented. However, Dr. Frank Bonaccorso's current research project utilizing radio-tracking with more than 30 Hawaiian hoary bats, reveals all the bats studied roost in trees and all roost more than 20 feet off the ground (Bonaccorso 2009b, pers. comm.). North American hoary bats roost 10 to 16 feet above the ground, mostly in hardwood trees (Shump and Shump 1982, p. 3). Hawaiian hoary bats have been observed in a wide variety of trees, including native species (*Metrosideros polymorpha*; *Pandanus tectorius*; *Styphelia tameiameia*), Polynesian-introduced species (*Aleurites moluccana*), and post-contact introduced species (*Syzygium cumini*) (Service 1998, p. 13). Bats also have been occasionally observed in fern clumps, low scrub, rock crevices, macadamia nut orchards, and buildings (Tomich 1986, pp. 11-24). Hawaiian hoary bats forage in a variety of open and vegetated habitats, including open fields, lava flows, open-ocean in bays near shore, and streams and ponds. Hawaiian hoary bats on Hawaii forage in both relatively closed habitats near vegetation (such as clearings in lowland mesic ohia forest or town parks) as well as in open habitats and forest edges (Jacobs 1993a; Tomich 1974, pp. 10–13). Foraging generally occurs three to 492 feet above the ground or open water, three to 50 feet above the ground in closed forest habitats, and up to 100 feet and more above tree canopy (Service 1998, p. 10).

***Dispersal/Migration*****Motility/Mobility**

Adult: High

**Dispersal/Migration Narrative**

Adult: Apparently does not migrate within the island of Hawaii (Jacobs 1994) (NatureServe, 2015). Hawaiian hoary bats have been observed year-round in a wide variety of habitats and elevations below 7,500 feet, and a few sightings from limited surveys have been reported as high as 13,199 feet. Hawaiian hoary bats have been detected in both wet and dry areas of Hawaii but seem to be more abundant on the drier leeward side (Jacobs 1994, p. 199) and generally less abundant in wet areas (Kepler and Scott 1990, p. 62). Only three researchers have examined spatial and temporal variation in occurrence patterns of bats in Hawaii, with conflicting conclusions about possible altitudinal or regional migration (Jacobs 1994, pp. 193-200; Menard 2001, pp. 1-149; Tomich 1986, pp. 1-30).

***Population Information and Trends*****Number of Populations:**

Unknown (NatureServe, 2015)

**Population Size:**

2500 - 10,000 individuals (NatureServe, 2015)

**Population Narrative:**

There are no population estimates for the Hawaiian hoary bat and few historical or current records. Unsubstantiated population estimates across the State have ranged from hundreds to a few thousand individuals (Service 1998, p. 14). Data are limited because no feasible method currently exists for surveying the abundance and distribution of solitary, tree-roosting bats. The

Hawaiian hoary bat's distribution may be broader than indicated by the current limited information resulting from localized search efforts (Service 1998, p. 14).

### ***Threats and Stressors***

**Stressor:**

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The major threats to the Hawaiian hoary bat are assumed to be the same as those that threaten many bat species in general (Harvey et al. 1999, p. 13; Service 1998, p. 15). Bats have the slowest reproductive rate and the longest life-span of all mammals of their size (Barclay and Harder 2003, pp. 209-256). Thus, any mortality of breeding-age adults, particularly females, constrains the recovery of the subspecies. The main factor limiting recovery is thought to be habitat loss, primarily the availability of roosting sites as suitable roosting habitat is particularly important to pregnant and lactating females and non-volant young (Service 1998, p. 15). Other possible threats identified in the recovery plan may include: roost disturbance, predation by native hawks and non-native feral cats, pesticide use (either directly or by impacting prey species), and alteration of prey availability due to introduction of non-native insects. In addition, occasional instances of Hawaiian hoary bat mortality due to collisions with vehicles and structures have been documented (Kepler and Scott 1990, p. 60; Kuhn 2009; Menard 2001, p. 136; Tomich 1986, pp. 11-30). Clearing of vegetation in areas where there are non-volant bat pups may result in the injury or death of those young. Hoary bats also may be impaled on barbed wire in the continental United States (Anderson 2002; Wisely 1978, p. 53) and in Hawaii (Burgett 2009, pers. comm.; Jeffrey 2007, pers. comm.; Marshall 2008, pers. comm.).

### ***Recovery***

#### **Recovery Actions:**

- The overall recovery strategy for the Hawaiian hoary bat is to rely on research that can provide information on the subspecies' abundance and distribution, life history, and habitat associations. The primary recovery goal should be to conduct research essential to the conservation of the Hawaiian hoary bat. Research should focus on developing standardized survey and monitoring protocols for determining abundance and distribution, roosting habitat associations, basic life history biology, and food habits. Other recovery goals are to protect and manage current populations by identifying and managing threats, including protection of key roosting and foraging areas; conduct a public education program; evaluate progress towards recovery; and revise recovery criteria as necessary (Service 1998, pp. 18-20). The Service, Hawaii Department of Land and Natural Resources – Division of Forestry and Wildlife (DOFAW), and Bat Conservation International (BCI, a non-profit conservation and education organization) are stakeholders in a public-private Hawaiian Hoary Bat Research Cooperative (Cooperative) which collaboratively prioritizes and funds management-oriented research on the Hawaiian hoary bat's abundance, distribution, and habitat requirements. Major stakeholders include private landowners, agricultural and commercial forestry interests, environmental groups, local governments, and Federal and State agencies. Most of the Cooperative's current funding is provided by the Service's Cooperative Endangered Species Conservation Fund (Section 6 of the Endangered Species Act) grants to the State. The Cooperative awarded funding to the U.S. Geological Survey –

Biological Resources Division for telemetry research in years 2004 to 2007, to complete baseline surveys to document Hawaiian hoary bat movements on the island of Hawaii. The Cooperative secured other funding to continue this research through 2009. The Service is also working with several private landowners in the state to develop Habitat Conservation Plans for the Hawaiian hoary bat. All may provide conservation benefits to the population as a whole as well as provide essential information regarding policy and management decisions.

## References

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## SPECIES ACCOUNT: *Leopardus (=Felis) pardalis* (Ocelot)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered throughout its range, including 22 countries; March 28, 1972; Southwest Region (R2)

### **Physical Description**

The ocelot (*Leopardus pardalis*; Linnaeus, 1758) is a medium-sized spotted cat. The coloration of the upper parts of the body is pale gray to cinnamon. There are spots on the head, two black stripes on the cheeks, and four to five longitudinal black stripes on the neck. The body shows elongated black-edged spots arranged in chain-like bands. The rounded ears are black dorsally, with a conspicuous white spot. The underparts are whitish, spotted with black. The tail is marked with dark bars or incomplete rings (Hall 1981). The dental formula is  $i3/3, c1/1, pm3/2, m1/1$ , for a total of 30 teeth. Weights range from 7-16 kg (15-35 lbs), with males weighing more than females.

### **Taxonomy**

The ocelot belongs to the genus *Leopardus* which also includes the margay (*Leopardus wiedii*) and the oncilla (*Leopardus tigrinus*). The ocelot is further divided into as many as 11 subspecies that ranged from the southwestern U.S. to northern Argentina (Pocock 1941, Cabrera 1961, Eizirik et al. 1998). Two subspecies occurred in the United States: the Texas/Tamaulipas ocelot (*L. p. albescens*) and the Arizona/Sonora ocelot (*L. p. sonoriensis*) (Hall 1981). We are using these common names for these subspecies.

### **Historical Range**

The late 19th century range of the Arizona/Sonora ocelot (*Leopardus pardalis sonoriensis*) included southeastern Arizona as far north as Fort Verde (Cockrum 1960, Hall 1981). Hoffmeister (1986) questioned the validity of the Fort Verde specimen and believed its origin may have been Mexico or Texas. In Mexico, the Arizona/Sonora ocelot occurs in the foothills of the Sierra Madre Occidental and associated sky island ranges from northeastern Sonora south into northern Sinaloa (Hall 1981), but it is absent from the desert scrub of western Sonora.

### **Current Range**

U.S.: Arizona and Texas; other: is known to or is believed to occur in Mexico and Central and South America. Currently, the ocelot ranges from extreme southern Texas and southern Arizona (although recent documentation in Arizona is sparse) through the coastal lowlands of Mexico to Central America, Ecuador and northern Argentina. It does not occur south of the Province of Entre Rios in Argentina (Denis 1964, Redford and Eisenberg 1992). The ocelot also is known from Trinidad and Isla de Margarita, Venezuela, but not from the Antilles (Tewes and Schmidly 1987, Sunquist and Sunquist 2002). There are no recent verified reports of ocelots from California or Florida.

### **Distinct Population Segments Defined**

No.

### **Critical Habitat Designated**

No;

## ***Life History***

### **Feeding Narrative**

Adult: The ocelot primarily eats cottontail rabbits and rodents, with 30 percent of food items represented by birds (M. Tewes, Texas A&M at Kingsville, unpubl.data). There is possible competition with bobcats and other small and medium-sized carnivores, but direct competition is minimal. Studies that have examined ocelot food habits relative to sympatric small and medium-sized carnivores suggest that direct competition with the ocelot is minimal (Brisbal 1986, Konecny 1989, Sunquist et al. 1989). Due to its similar size, relatively aggressive nature, and overlapping distribution, the bobcat (*Lynx rufus*), which does not occur in the tropical systems referenced above, is most likely to compete with the ocelot for food and space in Texas and northern Mexico. Regardless, the current ecological separation may reflect past competition (Connell 1980). Competition between the ocelot and the bobcat may be minimal because the ocelot is more of a habitat specialist than the bobcat, and it lives in areas of dense cover and high prey populations (Sunquist 2002).

### **Reproduction Narrative**

Adult: There are few data on reproductive rates of wild ocelots. Based on captive studies, gestation averages 80 days with a range of 77 – 83 days (Eaton 1977, Fowler 1978, Mansard 1990a and 1990b, William Swanson pers. comm. 2013, Tewes and Schmidly 1987). Although breeding has been suggested to peak during autumn in Texas (Tewes and Schmidly 1987) year-round breeding is known (Eaton 1977, Laack et al. 2005). Inter-estrous periods of two to three weeks have been documented in captive animals (Swanson 2002, Stoops et al. 2007). Wild ocelots probably first produce young at about 18 to 30 months (Eaton 1977, Bragin 1994, Tewes and Schmidly 1987). Laack (1991) observed first reproduction in wild female ocelots between 30 and 45 months. Bragin (1994) reported that reproduction among females in captivity occurred between about 20 months to 15 years and for male captive ocelots between about 22 months to 17 years. Eaton (1977) reported that captive ocelots produce an average of about 1.4 young per litter, but suggested that the average in the wild may be closer to 2.0. Laack et al. (2005) reported an average of 1.2 kittens per litter for 16 litters born to 12 ocelots in Texas. Emmons (1988) suggested that females in Peru produced young “about every other year.” However, this estimate did not take into account occasional losses of complete litters, which would add to the interval of successful reproduction. No comparable data are available for Texas. Although an inter-birth interval of two years or less suggests that half or more of adult females breed each year, felid populations may contain adult females that never produce litters (Beier 1993, Logan and Sweanor 2001), and this can have important effects on demography (Beier 1993). There are no published reports on what fraction of female ocelots fail to produce litters. (narrative from FWS 2016)

### **Geographic or Habitat Restraints or Barriers**

Adult: Inadequate brush or tree cover

### **Spatial Arrangements of the Population**

Adult: Ocelot spatial patterns are strongly linked to dense cover or vegetation (Emmons 1988, Horne et al. 2009).

### **Habitat Narrative**

Adult: The ocelot uses a wide range of habitats throughout its range (Tewes and Schmidly 1987, Shindle and Tewes 1998, López González et al. 2003, Avila-Villegas and Lamberton-Moreno 2013). In south Texas, the species occurs predominantly in dense thornscrub communities (Navarro-Lopez 1985, Tewes 1986, Laack 1991). Tewes (1986) found that core areas of ocelot home ranges contained more thornscrub than peripheral areas of their home ranges on LANWR in southern Texas. Laack (1991) also found ocelot use of dense thornscrub on LANWR. Caso (1994) found ocelots used primarily forest or woody communities in Tamaulipas, Mexico, and used the open pastures much less often. The pastures that were seldom used by ocelots supported little woody cover and were dominated by guinea grass (*Panicum maximum*). Jackson et al. (2005) suggested that the ocelot in Texas preferred closed canopy over other land cover types, but that areas used by this species tended to consist of more patches with greater edge. Horne et al. 2009 reported that ocelots in Texas selected woodland communities with >75% visually-estimated canopy cover. Other microhabitat features important to ocelots appear to be canopy height (>2.4 m) and vertical cover (90.4% visual obscurity at 1-2 m). Ground cover at locations used by ocelots was characterized by a high percentage of coarse woody debris (50%) and very little herbaceous ground cover (3%), both consequences of the dense woody canopy (Horne 1998). Little is known about ocelot habitat use in Arizona and Sonora; however, López González et al. (2003) found that 27 of the 36 records (75%) of ocelots in Sonora were associated with tropical or subtropical habitat, namely subtropical thornscrub, tropical deciduous forest, and tropical thornscrub.

### ***Dispersal/Migration***

#### **Motility/Mobility**

Adult: Moderate

#### **Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory

#### **Dispersal**

Adult: Dispersal period of juveniles lasts 7 to 9.5 months, suggesting a lack of vacant territories

#### **Dispersal/Migration Narrative**

Adult: One ocelot was recorded (Laack 1991) to have dispersed at 14 months-of-age, one at 20 months-of-age, and five at 30-35 months-of-age. Two others were captured as dispersers at <23 months-of-age. There was no obvious sex difference in age at dispersal. Only four dispersers (3 M, 1 F) lived to establish home ranges, but one female was still dispersing at the end of the study. Duration of successful dispersal (time elapsed between leaving natal range and establishing an independent home range) was 7 to 9.5 months. The one successful female disperser moved 2.5 km (distance between home range centers) whereas the successful males moved 7 to 9 km. In addition to these dispersers, one subadult female did not disperse but remained in her natal range after reaching adulthood. The extensive explorations of these animals, use of agricultural and other marginal habitat during dispersal, and the dispersal interval suggest that suitable habitat in LANWR was saturated with resident ocelots during this study (Laack 1991). All dispersers in the Laack (1991) study used narrow (5-100 m) corridors of brush during dispersal. Most of these were along resacas (remnants of former river meanders) and drainage ditches. Each disperser avoided areas occupied by adults, and repeatedly returned to one or more small, semi-isolated patches of thornscrub during dispersal. The established

adult home ranges of these same animals did not include these semi-isolated patches. Transient home ranges were often farther from the natal range than the animal's eventual home range. Dispersal from LANWR typically involved movements to and beyond the point that scrub habitat became scarce, use of transient home ranges centered on small scrub patches within agricultural or pasture land, and eventual settling within the source population when a vacancy occurred, or when the animal gained enough mass and experience to displace a resident adult. Dispersal was generally frustrated and circular – similar to that of pumas (*Puma concolor*) in urban southern California (Beier 1995) and panthers (*P. c. coryi*) in south Florida (Maehr et al. 2002). Dispersers may have moved farther if corridors or even stepping stones of suitable habitat were available. Although such dispersal zones are technically “sink habitats” (sites where mortality exceeds reproduction), they also serve as areas free of resident adults where some young ocelots survive as a pool of non-breeding “floaters” that can be an important stabilizing force in population dynamics. According to Laack (1991), no disperser successfully joined a population outside of LANWR.

### ***Population Information and Trends***

#### **Population Trends:**

Declining

#### **Species Trends:**

Declining

#### **Resiliency:**

Low

#### **Representation:**

Low

#### **Redundancy:**

Low

#### **Number of Populations:**

Two separate populations in south Texas; 4 individuals in Arizona

#### **Population Size:**

Texas: 53 individuals; Arizona: 5 individuals (1 dead, 4 alive)

#### **Population Narrative:**

As of August 2015, there were 53 total known individuals in the two separate populations in south Texas (Hilary Swarts unpubl. data 2015, Mike Tewes pers. comm. 2015). A third and much larger population of the Texas-Tamaulipas ocelot (*L. p. albescens*) occurs in Tamaulipas, Mexico (Caso 1994, Carvajal-Villarreal et al. 2012, Stasey 2012, Conservación y Desarrollo de Espacios Naturales 2014), but it is thought to be isolated from ocelots in Texas (Walker 1997, Janečka et al. 2014). In Arizona, five individual ocelots (four live and one dead) have been detected between 2009 and 2015 (Avila-Villegas and Lamberton-Moreno 2013, Culver et al. 2016). Prior to these five recently-known individuals the last documentable ocelot in Arizona was a male that had been killed by a vehicle near the town of Oracle in 1967 (López González et al. 2003). In

addition to the recent Arizona sightings, ocelots have been documented in Sonora, Mexico (López González et al. 2003, Gómez-Ramírez 2015), including a female with a kitten about 48 km south of the U.S.-Mexico border in 2011 (Avila-Villegas and Lamberton-Moreno 2013). Major focus for recovery goals is the Texas-Tamaulipas Management Unit (TTMU) and the Arizona-Sonora Management Unit (ASMU). Currently, the Texas ocelot population is estimated at 80 ocelots, which are found in two separated populations in southern Texas. This estimate is based on a combination of 55 known individuals, identified by their unique coat patterns, and the extrapolation of an additional 25 ocelots based on existing suitable habitat on private lands near or adjacent to existing ocelot-occupied habitat. (USFWS 2018)

### ***Threats and Stressors***

**Stressor:** Habitat conversion, fragmentation, and loss

**Exposure:** Not assessed; see narrative.

**Response:** Not assessed; see narrative.

**Consequence:** Not assessed; see narrative.

**Narrative:** Habitat conversion, fragmentation, and loss comprise the primary threats to the ocelot today. Human population growth and development continue throughout the ocelot's range. In Texas, more than 95 percent of the dense thornscrub habitat in the Lower Rio Grande Valley has been converted to agriculture, rangelands, or urban land uses. The amount of ocelot habitat loss in Arizona and Sonora has not been quantified, but it is thought to be much less than in the TTMU. A current threat to ocelot habitat in Arizona includes mining development (Erin Fernandez pers. comm. 2015, USFWS 2016).

**Stressor:** Small, disconnected populations

**Exposure:** Not assessed; see narrative.

**Response:** Not assessed; see narrative.

**Consequence:** Not assessed; see narrative.

**Narrative:** Small population sizes in Texas and isolation from conspecifics in Mexico threaten the ocelot in Texas with inbreeding. Connectivity among ocelot populations or colonization of new habitats is inhibited by road mortality among dispersing ocelots. Issues associated with border barrier development and patrolling the boundary between the United States and Mexico further exacerbate the isolation of Texas and Arizona ocelots from those in Mexico.

**Stressor:** Agricultural pesticides and herbicides

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Agricultural pesticides and herbicides may have negative impacts on the ocelot. While common contaminants have appeared in ocelots, they occur at low levels and do not seem to be a major problem (Mora et al. 2000). On the other hand, impacts to potential prey have been observed. For example, in the Lower Rio Grande Valley of Texas the number of amphibian and reptile species has been reduced by 65 percent and 51 percent respectively, perhaps the result of decreased prey availability (Jahrsdorfer and Leslie 1988).

**Stressor:** Hunting, poaching, fur-trapping and predator control

**Exposure:** Likely almost all illegal; predator control methods have been changed to protect the ocelot

**Response:** Considered to be reduced by limitations on trap types and corridors

**Consequence:** Possible take but not significant threat to the species

**Narrative:** It was recognized in the Biological Opinion that the WDM Program staff limit their activities in travel corridors and occupied ocelot habitat and that there has not been an incident of ocelot capture in Arizona or Texas by ADC/Wildlife Services staff in over 30 years.

### ***Recovery***

#### **Reclassification Criteria:**

Populations south of Tamaulipas and Sonora (Mexico) continuously qualify for “Least Concern” under the International Union of Concerned Scientists (IUCN) Red List criteria (World Conservation Union, 2006, <http://www.iucnredlist.org>) for 5 years; and threats from habitat loss, habitat fragmentation, and poaching have been reduced such that the ocelot is no longer in danger of extinction.

The Texas/Tamaulipas (TTMU) population is estimated through reliable scientific monitoring to be above 200 in Texas and 1,000 in Tamaulipas for at least 5 years. The 200 ocelots in Texas should be distributed as either (1) a single core population of at least 150 ocelots with interchange between it and ocelots in Tamaulipas that is sufficient to maintain genetic variability; or (2) at least 2 core populations of 75 ocelots each, with interchange between the 2 core populations and between the core populations and ocelots in Tamaulipas that is sufficient to maintain genetic variability. Interchange may be facilitated by moving ocelots between populations to simulate natural dispersal and recruitment. Core populations may include the current populations at LANWR and in Kenedy-Willacy Counties, but may also include a reintroduced population established within currently unoccupied historical range. Habitat protection must be in place to support core ocelot populations for the foreseeable future, and potential corridors and mechanisms must be identified to restore habitat connectivity between core populations.

The Arizona/Sonora Management Unit (ASMU) population is estimated through reliable scientific monitoring to be above 1,000 animals for 5 years. Habitat linkages to support an ASMU metapopulation have been identified. Threats to this population have been identified and are determined to be below the threshold of endangerment of extinction within the foreseeable future. Methods to address significant threats have been identified.

#### **Delisting Criteria:**

Populations south of Tamaulipas and Sonora (Mexico) continue to qualify for “Least Concern” under the IUCN Red List criteria (World Conservation Union, 2006, <http://www.iucnredlist.org>) for 10 years and populations are stable or increasing; and threats from habitat loss, habitat fragmentation, and poaching are reduced such that the ocelot can maintain healthy, viable populations for the foreseeable future.

The TTMU population is estimated through reliable scientific monitoring to be above 200 in Texas and 1,000 in Tamaulipas for at least 10 years. The 200 ocelots in Texas should be distributed as either (1) a single core population of at least 150 ocelots with interchange between it and ocelots in Tamaulipas that is sufficient to maintain genetic variability; or (2) at least 2 core populations of 75 ocelots each, with interchange between the 2 core populations and between the core populations and ocelots in Tamaulipas that is sufficient to maintain

genetic variability. Interchange among populations must occur through natural dispersal rather than by moving ocelots between population; or (3) if natural interchange between Texas and Tamaulipas is impossible, cross-border interchange may be facilitated by moving ocelots to simulate natural dispersal and recruitment and an additional population of at least 75 ocelots is established within currently unoccupied historical range in Texas. This additional population should be established in a location that would expand the geographical range of the species in Texas to provide sufficient insurance against loss of the entire Texas population from catastrophic weather events or infectious disease. Habitat protection must be in place to support and connect all core ocelot populations within Texas and within Tamaulipas for the foreseeable future.

The ASMU population is estimated through reliable scientific monitoring to be above 1,000 animals for 10 years. Significant threats to this population have been identified and addressed. Habitat linkages to facilitate an ASMU metapopulation have been identified and are conserved for the foreseeable future.

**Recovery Actions:**

- Assess, protect, and enhance ocelot populations and habitat in the borderlands of the U.S. and Mexico;
- Reduce the effects of human population growth and development on the ocelot;
- Maintain or improve genetic fitness, demographic conditions, and health of the ocelot in borderland populations;
- Assure the long-term success of ocelot conservation through partnerships, landowner incentives, community involvement, application of regulations, and public education and outreach;
- Practice adaptive management in which recovery is monitored and recovery tasks are revised by USFWS in coordination with recovery implementation team subgroups as new information becomes available;
- Support efforts to ascertain the status and conserve ocelot populations south of Tamaulipas and Sonora, Mexico.
- The potential need for the creation of a captive population to serve as “insurance” in the event the wild population suffers catastrophic losses, as well as to facilitate research that could not be done in the wild (Wiese and Hutchins 1994) should be examined for the ocelot.

***Conservation Measures and Best Management Practices:***

- Expand refuges;
- Restore agricultural land to native thornscrub;
- Restore areas impacted by wildfires;
- Protect existing habitat and create corridors between existing habitats;
- Create reserves in Mexico;
- Install highway underpasses for ocelots to prevent mortalities due to vehicle collisions.

**References**

USFWS 2010. Ocelot Recovery Plan (*Leopardus pardalis*) Draft First Revision, U.S. Fish and Wildlife Service, Southwest Region, Albuquerque, New Mexico, 185 p.

USFWS 2016. Recovery Plan for the Ocelot (*Leopardus pardalis*) First Revision, U.S. Fish and Wildlife Service, Southwest Region, Albuquerque, New Mexico, 237p.

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USFWS. 2010. Ocelot Recovery Plan (*Leopardus pardalis*) Draft First Revision, U.S. Fish and Wildlife Service, Southwest Region, Albuquerque, New Mexico, 185 p. USFWS. 2016. Recovery Plan for the Ocelot (*Leopardus pardalis*) First Revision, U.S. Fish and Wildlife Service, Southwest Region, Albuquerque, New Mexico, 237p.

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## SPECIES ACCOUNT: *Leptonycteris nivalis* (Mexican long-nosed bat)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered

#### **Physical Description**

A leaf-nosed bat. This grayish brown bat has a leaflike nose projection and no tail; forearm 55-60 mm (Whitaker 1996). LENGTH:9 WEIGHT: 21 (NatureServe, 2015)

#### **Taxonomy**

The Mexican long-nosed bat is a member of the family Phyllostomidae (New World leaf-nosed bats) and is grouped in the subfamily, Glossophaginae, with several other pollen-, fruit-, and nectar-eating bats. The genus *Leptonycteris* is characterized by two dental features, lack of the third molar and presence of lower incisors (Walker 1975). *Leptonycteris* means “slender bat” (leptos - slender, nycteris — bat), and the specific name *nivalis* refers to the fact that the type specimen was caught near snow line on the 17,816 feet (5,747 m) extinguished volcano, Mt. Orizaba, in Veracruz, Mexico. The original description by Saussure (1860), named these bats *Ischnoglossa nivalis*. Many changes in nomenclature have characterized these bats, and only recently the situation seems to have been settled by Arita and Humphrey (1988, see their paper for a review of classification and nomenclature). Arita and Humphrey (1988) analyzed measurements from 1,951 long-nosed bat specimens in the genus *Leptonycteris* and determined that *L. nivalis* is a monotypic species. Some studies prior to 1988 may have referred to *L. nivalis*, but because of Arita and Humphrey’s determination those individuals were actually found to be *L. curasoae*. Thus, literature prior to this time should be carefully scrutinized before conclusions about *L. nivalis* are made (USFWS, 1994).

#### **Current Range**

Range includes northern and central Mexico, southwestern Texas, and southwestern New Mexico, generally at elevations of about 500 to 3,000+ meters. Most occurrences in Mexico are at elevations of 1,000-2,200 meters, but this bat been captured at an elevation of 3,780 meters (see Arita 1991), and the type specimen reportedly was caught near snow line at 17,816 feet (5,747 meters) on Mt. Orizaba, in Veracruz, Mexico (USFWS 1994). In Texas, the species has been captured in Big Bend National Park (Brewster County) and the Chinati Mountains (Presidio County); Emory Peak Cave in the Chisos Mountains (elevation 2,290 meters) hosts the only known roosting population in Texas (Ammerman et al. 2012). Two specimens of *Leptonycteris* taken in Hidalgo County, New Mexico (in 1963 and 1967), were determined to be *L. nivalis*. The presence of this species in New Mexico was reconfirmed in Hidalgo County in 1992 (Hoyt et al. 1994). Populations exist in the Animas and Big Hatchet mountain in New Mexico (P. Cryan, pers. comm., cited by Ammerman et al. 2012). The range has been described as extending into Guatemala and adjacent southern Mexico (Hensley and Wilkins 1988; Simmons, in Wilson and Reeder 2005), but specimens collected from those areas were assigned to *L. yerbabuenae* by Arita and Humphrey (1988) and Arita (1991). Simmons (in Wilson and Reeder 2005) described the range of *L. nivalis* as including southeastern Arizona, but no actual records for Arizona are known.

#### **Critical Habitat Designated**

Yes;

***Life History*****Feeding Narrative**

Juvenile: Diet includes mainly nectar and pollen of at least 21 plant species representing 10 plant families (Sánchez and Medellín 2007). In the northern part of the range, the bats often feed at the flowers of cacti and paniculate agaves. In Texas; nectar of mescal and Chisos agave flowers probably are the main food (Schmidly 1977). The diet may include insects associated with flowers, and probably some fruits, especially in the southern part of the range.; Food Habits: Nectarivore (Adult, Immature), Frugivore (Adult, Immature). Activity occurs throughout the year. Emergence to feed occurs relatively late in the evening.; (NatureServe, 2015)

Adult: Diet includes mainly nectar and pollen of at least 21 plant species representing 10 plant families (Sánchez and Medellín 2007). In the northern part of the range, the bats often feed at the flowers of cacti and paniculate agaves. In Texas; nectar of mescal and Chisos agave flowers probably are the main food (Schmidly 1977). The diet may include insects associated with flowers, and probably some fruits, especially in the southern part of the range.; Food Habits: Nectarivore (Adult, Immature), Frugivore (Adult, Immature). Activity occurs throughout the year. Emergence to feed occurs relatively late in the evening.; (NatureServe, 2015)

**Reproduction Narrative**

Adult: Litter size normally is 1. Young are born apparently in spring (April-June), primarily in Mexico before females arrive in Texas, though pregnant females have been captured in Texas in late April (Brown 2008, Ammerman et al. 2012). In Texas, lactating females have been observed in June-July, flying juveniles in late June. Young are weaned in July or August. These bats are highly colonial.; These bats are effective pollinators of cacti and agave; the plants are dependent on bats for sexual reproduction.; (NatureServe, 2015)

**Spatial Arrangements of the Population**

Adult: Clumped roosting/Uniform feeding (NatureServe, 2015)

**Environmental Specificity**

Adult: Narrow/specialist (NatureServe, 2015)

**Tolerance Ranges/Thresholds**

Adult: Moderate (Feeding areas), Low (Roosting areas) (NatureServe, 2015)

**Site Fidelity**

Adult: High (NatureServe, 2015)

**Habitat Narrative**

Adult: Habitats include desert scrub, open conifer-oak woodlands, and pine forests in the Upper Sonoran and Transition Life Zones; generally arid areas where agave plants are present (USFWS 1994). Colonies roost in caves (or similar mines and tunnels), sometimes in culverts, hollow trees, or unused buildings. Roosting habitat requirements are not well known. (NatureServe, 2015). Moderate ecological integrity of the population and tolerance ranges for feeding areas are based on the fact that this is a mobile species and can cover large areas in search of food. High ecological integrity of the population and low tolerance ranges for roosting/maternity

caves is based on the fact that these caves are limited in number and need to be relatively undisturbed for the species to thrive. High site fidelity is based on the species reliance on caves for roosting and maternity sites.

***Dispersal/Migration*****Motility/Mobility**

Adult: High (NatureServe, 2015)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Migratory (NatureServe, 2015)

**Dispersal**

Adult: Moderate (NatureServe, 2015)

**Immigration/Emigration**

Adult: Unlikely (NatureServe, 2015)

**Dispersal/Migration Narrative**

Adult: This species is migratory in the northern portion of its range (Wilson et al. 1985, Schmidly 1991, Ammerman et al. 2012), but movements are not well-known. Seasonal movements likely correspond with food availability. It has been recorded in the United States from June to August (Ammerman et al. 2012). Most northward migrants in Texas are females, but in New Mexico the sex ratio is more balanced (Hoyt et al. 2004; P. Cryan, pers. comm., cited by Ammerman et al. 2012).; This species is highly mobile and is capable of traveling great distances, including its migration from Mexico to the southern U.S. Dispersal is listed as moderate because of the species likelihood of using the same caves year after year. Unlikely immigration/emigration is based on the lack of information concerning this species leaving areas it has been known to occur or inhabiting new territory. (NatureServe, 2015)

***Population Information and Trends*****Population Trends:**

Unknown (NatureServe, 2015)

**Resiliency:**

Moderate (NatureServe, 2015)

**Representation:**

Moderate (NatureServe, 2015)

**Redundancy:**

Moderate (NatureServe, 2015)

**Population Growth Rate:**

Long-term trend is unclear. Extent of occurrence and area of occupancy probably have not changed much, but the number of occurrences or subpopulations and population size may have dramatically decreased in some locations during the last three decades. Wilson (1985) found

that this species was either completely absent or present in reduced numbers in known roosts. The number of bats found represented only a fraction of the total reported in previous studies. For example, in an abandoned mine in Nuevo Leon, Mexico, where an estimated population of 10,000 was observed in 1938, no individuals of *L. nivalis* were found in 1983 (Wilson 1985). Another mine in Nuevo Leon had a ceiling covered with newborn bats in 1967, but only one bat was found in 1983. A few other roosts had reduced numbers of bats compared to findings during previous surveys. These changes could indicate a decline in the overall population, but they might reflect movement of bats among different roosting sites in different years, or they could result from seasonal changes in bat distribution (survey dates varied). A colony of *L. nivalis* in Morelos, Mexico, increased from an estimated 5,000 in 1996 to 8,000-10,000 in 2001-2002 (Medellín 2003). Abundance at Emory Peak Cave in Texas fluctuates widely from year to year (0 to 10,000+ individuals). Reasons for the fluctuations are not completely understood, but they apparently reflect annual variations in regional food resources (number of flowering agave plants) (USFWS 1994, Ammerman and Tabor 2008); a similar pattern has been observed at a cave in Nuevo Leon, Mexico (Moreno-Valdez et al. 2004). Historical count data for Emory Peak Cave may not be completely reliable; bats present in the cave may go undetected (Ammerman et al. 2009, 2012). Ammerman et al. (2012) noted a lack of consistency among various trend estimates or indications for this species (Wilson 1985, Arita and Humphrey 1988, Cockrum and Petryszyn 1991, USFWS 1994, Medellín 2003, Ammerman and Tabor 2008, Ammerman et al. 2009). Despite some inconsistency, most authors have concluded that the species is declining in abundance, though the degree of decline is highly uncertain. However, better population data based on improved monitoring methods are needed before a reliable trend determination can be made (Ammerman et al. 2012). Decline of 30-70% (NatureServe, 2015)

**Number of Populations:**

21 - 80 (NatureServe, 2015)

**Population Size:**

10,000 - 1,000,000 individuals (NatureServe, 2015)

**Population Narrative:**

Long-term trend is unclear. Extent of occurrence and area of occupancy probably have not changed much, but the number of occurrences or subpopulations and population size may have dramatically decreased in some locations during the last three decades. Wilson (1985) found that this species was either completely absent or present in reduced numbers in known roosts. The number of bats found represented only a fraction of the total reported in previous studies. For example, in an abandoned mine in Nuevo Leon, Mexico, where an estimated population of 10,000 was observed in 1938, no individuals of *L. nivalis* were found in 1983 (Wilson 1985). Another mine in Nuevo Leon had a ceiling covered with newborn bats in 1967, but only one bat was found in 1983. A few other roosts had reduced numbers of bats compared to findings during previous surveys. These changes could indicate a decline in the overall population, but they might reflect movement of bats among different roosting sites in different years, or they could result from seasonal changes in bat distribution (survey dates varied). A colony of *L. nivalis* in Morelos, Mexico, increased from an estimated 5,000 in 1996 to 8,000-10,000 in 2001-2002 (Medellín 2003). Abundance at Emory Peak Cave in Texas fluctuates widely from year to year (0 to 10,000+ individuals). Reasons for the fluctuations are not completely understood, but they apparently reflect annual variations in regional food resources (number of flowering agave plants) (USFWS 1994, Ammerman and Tabor 2008); a similar pattern has been

observed at a cave in Nuevo Leon, Mexico (Moreno-Valdez et al. 2004). Historical count data for Emory Peak Cave may not be completely reliable; bats present in the cave may go undetected (Ammerman et al. 2009, 2012). Ammerman et al. (2012) noted a lack of consistency among various trend estimates or indications for this species (Wilson 1985, Arita and Humphrey 1988, Cockrum and Petryszyn 1991, USFWS 1994, Medellín 2003, Ammerman and Tabor 2008, Ammerman et al. 2009). Despite some inconsistency, most authors have concluded that the species is declining in abundance, though the degree of decline is highly uncertain. However, better population data based on improved monitoring methods are needed before a reliable trend determination can be made (Ammerman et al. 2012). Decline of 30-70% Total adult population size is unknown but presumably exceeds 10,000 and may exceed 100,000, based on the extensive range and the periodic presence of several thousand individuals in the single roost in Texas. This species is widespread but "scarce" in most of its range (Arita 1993, Arita and Santos-del-Prado 1999). In Mexico, the species forms cave colonies of fewer than 100 individuals or 100-10,000 individuals (Arita 1993). Moreno-Valdez et al. (2004) recorded a population of up to a few thousand individuals at El Infierno Cave, Nuevo Leon, Mexico, in the late 1990s.

Roost at Emory Peak Cave (Texas): 10,650 bats in 1967, 5,000 in 1968, 3,900 in 1969, 0 in 1970, 8025 in 1971, 1000 in 1983, 4,942 -5,990 in 1988, 5,000+ in 1991, 0 in 1992, and 2,859 in 1993 (Matthews and Moseley 1990, see USFWS 1994). Large annual fluctuation may be in part due to some counts being made before or after the main period of bat occupation of the cave (Cockrum and Petryszyn 1991), or it may be an artifact of some bats being present but not detected (Ammerman et al. 2009), or it may reflect the movements of bats in response to available food resources (Moreno-Valdez et al. 2004, Ammerman and Tabor (2008). The number of occurrences or subpopulations has not been determined using standardized/meaningful criteria; if based on regularly occupied roost sites, the number probably exceeds 20 and may not exceed 80. Arita and Humphrey (1988) and Arita (1991) mapped 43 confirmed collection sites for *L. nivalis* in Mexico, though not all of these necessarily represent roost sites. Just a few roosting sites exist in the United States, but there could be additional sites that have not yet been detected. (NatureServe, 2015). Moderate resiliency, representation and redundancy are based on the number of use sites and their relatively wide geography as well as the overall population size.

### ***Threats and Stressors***

**Stressor:** Disturbance and destruction of roost sites (USFWS, 1994)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Modification or destruction of roost sites is listed as a threat to this species (USFWS, 1994).

**Stressor:** Modification of foraging habitat (USFWS, 1994)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Foraging habitat disruption and destruction has also been identified as a threat to *L. nivalis*. Foraging habitat can be modified or destroyed by the harvesting of agave for mescal and pulque, the expansion of agriculture, and other land uses. The main threat to food plants is from "moonshining" not from government regulated liquor industries (D. Howell and G. Nabhan, pers.

comm.). The large fields of planted agaves like those around Jalisco probably supplanted few natural agaves prior to the tequila industry. Public relations people from Jos6 Cuervo tequila have investigated the advisability of letting a few rows in each cultivated field go to flower to provide a food source for bats (Howell, pers. comm.). Nabhan and Fleming (1993) have estimated that bootleg mescal makers are eliminating between 500,000 and 1,200,000 wild paniculate agaves a year in Sonora alone. Nabhan (pers. comm.) indicated that in no place were agaves completely wiped out but that the agaves left to bloom in the Sonora study area are often widely dispersed or in inaccessible areas which make harvesting unproductive (USFWS, 1994). Although it is not known how far *L. nivalis* will fly to forage or how clumped the resource must be to be energetically productive, at some point widely spaced flowering stalks and distance to clumps become inefficient and affects reproduction and survival. Nabhan and Fleming (1993) suggest that the “tequila connection” is not as important as was once thought. “There are few places in Sonora or elsewhere in Mexico where wild Agave harvesting has eliminated a significant percentage of nectar—producing genets... because indigenous harvesters know how to disrupt apical dominance..., to encourage vegetative offshooting... before removing the ‘mother plant’ for mescal production.” However, by removing the flowering stalk “head” thus encouraging vegetative offshooting, they delay flowering (until the vegetatively produced plants mature) and eliminate the possibility of the flowering stalk becoming available to the bats that year. The impact of alcoholic beverage production on Mexican long—nosed bat foraging and survival is far from clear (USFWS, 1994).

**Stressor:** Pesticides (USFWS, 1994)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat/loss of individuals

**Narrative:** The use of pesticides may also negatively affect *L. nivalis*. Because long—nosed bats are nectarivorous, they are probably not as susceptible to pesticide effects as insectivorous bats. However, pesticides may be applied in a way that covers everything that is exposed, and thus, might fall on the bat’s food plants. When bats feed on the nectar, soft fruits, or incidentally on insects, pesticides might be consumed by the bats. Reidinger (1976) found lesser long-nosed bats in Arizona and Sonora, Mexico contained the least amount of organochlorine residues of all bats sampled. Reidinger (1976) did not speculate on the possible effects of the pesticide level he did find in *Leptonycteris* (USFWS, 1994).

**Stressor:** Competition (USFWS, 1994)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Interspecific competition may occur between *L. nivalis* and *L. curasoae* and *Choeronycteris mexicana* (USFWS, 1994). Competition for roost space may also occur with other bat species, particularly where caves are not abundant and cattle ranching and livestock production have artificially increased vampire bat populations by providing easy and abundant prey. Vampire bats commonly occupy the highest, darkest, warmest places in caves (Medellin, pers. obs.; Turner 1975). On several occasions, vampire bats have been found to replace non—vampire species (Medellin, pers. obs.). Turner (1975) also noted a similar trend; when the number of vampire bats increased, the number of non—vampire bats in the roost decreased or remained constant, but rarely increased (USFWS, 1994).

**Stressor:** Disease (USFWS, 1994)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** One study has suggested that rabies may be present in Mexican long-nosed bats (Villar and Jimenez 1962). However, there is some doubt regarding the specific identification of the bats in that report. Additionally, the incidence of rabies is very low in non-sanguivorous (non-blood eating) bats, less than half of 1 percent (no higher than that seen in many other animals) (Tuttle 1988). No real threat is apparently posed by other diseases for this species, although this factor can not be completely discounted (USFWS, 1994).

**Stressor:** Predation (USFWS, 1994)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** Although there are no documented cases of predation of *L. nivalis*, they probably experience predation from owls, hawks, snakes, and mammals (including raccoons, cats, and ringtails) similar to other bat species (Tuttle and Stevenson 1982). In the case of *L. nivalis*, predation does not seem to be a particularly important limiting factor. However, the impact of predation is likely much greater than generally realized and low reproductive rates of most bats greatly increase the importance of even low predation rates (Tuttle and Stevenson 1982). Anthropogenically caused increased populations of domestic or feral cats and other predators may affect survival of bat colonies, particularly maternity colonies near human habitations (USFWS, 1994).

**Stressor:** Natural catastrophes/climate (USFWS, 1994)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Other natural events that may impact Mexican long-nosed bats are climate and natural catastrophes. Some particularly severe winters may have an effect on the amount of food availability. For example, in mid-elevation areas a late- or early-season freeze may dramatically reduce the number of live flowers, particularly since these flowers are open at night when the coldest temperatures occur. Such conditions could cause starvation or migration of bat colonies. Additionally, roost destruction due to earthquakes, floods, or other natural causes may destroy entire bat colonies. These factors would not pose a serious threat to the species if populations were at their original numbers. However, if the species is receiving additional pressure from human activities, natural disasters may play a critical role in the species' survival (USFWS, 1994).

## **Recovery**

### **Reclassification Criteria:**

(1) The five major roosts (Cueva del Diablo, Aguacatitla Tunnel, El Infierno Cave, El Rosillo Cave, and Mount Emory Cave) have protections in the form of enforcement, management and education to reduce human interaction, from a non-permitted person, with the bat colony to zero and eliminate construction that could threaten the structural integrity of the roost, impede entry by the bats into the roost, or cause abandonment of the roost. (2) Cueva del Diablo

maintains a colony size of at least 10,000 bats over a 10-year period. (3) A continuous, standardized monitoring program is implemented for food sources of the bat to manage for the threats of land use change and climate change (USFWS 2018).

**Delisting Criteria:**

(1) The five major roosts (Cueva del Diablo, Aguacatitla Tunnel, El Infierno Cave, El Rosillo Cave, and Mount Emory Cave) have protections in the form of enforcement, management and education to reduce human interaction, from a non-permitted person, with the bat colony to zero and eliminate construction that could threaten the structural integrity of the roost, impede entry by the bats into the roost, or cause abandonment of the roost. (2) In addition to the five major roosts, there are at least five additional roosts with a minimum colony size of 500 bats over a 10-year period. (3) Cueva del Diablo maintains a colony size of at least 12,000 bats over a 10-year period. (4) A continuous, standardized monitoring program is implemented for food sources of the bat to manage for the threats of land use change and climate change (USFWS 2018)

**Recovery Actions:**

- Reclassification to threatened cannot occur until what constitutes a population and how a population migrates and uses habitat is understood. Criterion 1 will entail considerable work to determine what constitutes a Mexican long-nosed bat population. Currently (1994), there are no data that describe a population of Mexican long-nosed bats. This information will be collected as the result of research described in the recovery tasks and should include searches for other roost sites as well. Then, protection for at least six populations should be established. The six populations is a tentative number based on Arita and Humphrey's (1988) grouping of specimens. Information gathered from recovery task implementation can be used to refine the number of populations required for downlisting (USFWS, 1994).
- Then, protection for at least six populations should be established. The six populations is a tentative number based on Arita and Humphrey's (1988) grouping of specimens. Information gathered from recovery task implementation can be used to refine the number of populations required for downlisting. These reclassification criteria are preliminary and may be revised as new information becomes available (including research specified as recovery tasks in this plan). The estimated date for attaining the objective of this plan (downlisting to threatened) is the year 2014. This estimated date is based on about 10 years to complete the research necessary to determine the 6 populations and 10 years after that to ensure the protected populations are maintained (USFWS, 1994).

***Conservation Measures and Best Management Practices:***

- Most research on long-nosed bats has been done on the lesser long-nosed bat, with some projects in northern Mexico and Arizona ongoing. Very few studies have examined *L. nivalis*, and even fewer have been published. The surveys examining the majority of the species' range were done in 1983 and 1984 (Wilson 1985). No known complete surveying or monitoring of these or other sites has since been conducted (USFWS, 1984).
- Big Bend National Park's Mt. Emory cave is the only known protected Mexican long-nosed bat roost on public land in the U.S. and is the only roost that has had a multi-year monitoring effort. The National Park Service plans to continue this monitoring effort (M. Fleming, BBNP, pers. comm.) (USFWS, 1994).
- An on-going project conducted by Dr. Alfonso Valiente, of the Centro de Ecología, Universidad Nacional Autónoma de México, indirectly involves *L. nivalis*. He is monitoring nectar and pollen

production, determining pollinating agents, and studying the reproductive biology of the cactaceae in the xeric Valle de Tehuachn, State of Puebla, which is within an area of known *L. nivalis* occupancy (Medellin, pers. obs.) (USFWS, 1994).

- A vampire bat control/education project is intermittently ongoing in different areas of Mexico and is planned to coalesce into a national program. Also, an initiative is planned to approach tequila producers in the highlands of Jalisco with information about protecting bats. This project is planned to begin operation in mid-1994 (USFWS, 1994).

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## SPECIES ACCOUNT: *Lynx canadensis* (Canada Lynx)

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### *Species Taxonomic and Listing Information*

**Commonly-used Acronym:** None

**Listing Status:** Threatened: March 24, 2000 (65 FR 16052); Clarification of Findings: July 3, 2003 (68 FR 40076), January 10, 2007 (72 FR 1186). Recommended for delisting due to recovery (USFWS 2017).

### **Physical Description**

Canada lynx (*Lynx canadensis*) is a medium-sized cat typically measuring 75 to 90 centimeters (cm) (30 to 35 inches [in.]) long and weighing anywhere between 8 and 10.5 kilograms (kg) (18 to 23 pounds [lbs.]). Adult males average 10 kg (22 lbs.) in weight and 85 cm (33.5 in.) in length (head to tail), and females average 8.5 kg (19 lbs.) in weight and 82 cm (32 in.) in length. Canada lynx have large, well-furred feet and long legs, highly adapted for walking on deep snow. They have tufts on the ears and a black-tipped tail. The winter coat of the lynx is dense and has a grizzled appearance, with grayish-brown mixed with buff or pale brown fur on the back; and grayish-white or buff-white fur on the belly, legs, and feet. Summer fur of the lynx is more reddish to gray-brown (USFWS 2005; USFWS 2016).

### **Taxonomy**

Canada lynx is a member of the family Felidae. There are two currently recognized North American subspecies: *Lynx canadensis canadensis*, *Lynx canadensis mollipilosus* and *Lynx canadensis subsolanus* (ITIS 2016). Lynx hybridization with bobcats has been documented in Minnesota, Maine, and New Brunswick, where male bobcats bred with female lynx to produce fertile offspring with lynx-like ear tufts, intermediate foot-size, and bobcat-like fur. The lynx's large, well-furred paws, long black ear tufts, and short, black-tipped tail distinguish it from the bobcat (USFWS 2013). It is still not known to what extent hybridization occurs between the two species, but it has probably occurred at low levels, especially at the southern edge of lynx range and northern edge of bobcat, range where lynx and bobcat come into contact.

### **Historical Range**

The historical range of the Canada lynx in the contiguous United States is in the southern extensions of the boreal forest in the Northeast, Great Lakes, Rocky Mountains, and Cascade Mountains. The range extends from Alaska across much of Canada (except for coastal forests), with southern territories extending into parts of the western United States, the Great Lakes states, and New England. Canada lynx distribution is directly correlated with the distribution of snowshoe hares and boreal forests. Boreal forests extend southward from the arctic tundra in the far north, to boreal/hardwood forest ecotones in the Midwest and eastern United States, and to subalpine forests in the western United States (NPS 2013).

### **Current Range**

Canada lynx are endemic to North America. Canada lynx in the contiguous United States are at the southern margins of a widely distributed range across Canada and Alaska. The lynx is listed in the 14 States that support boreal forest landscapes, including Colorado, Idaho, Maine, Michigan, Minnesota, New Hampshire, New York, Oregon, Montana, Utah, Vermont, Washington, Wisconsin, and Wyoming (USFWS 2005). The areas with the most continuous occupation of lynx populations in the contiguous United States are defined as "core areas." Six

core areas have persistent occurrences and recent evidence of reproduction, including Northeast (Northern Maine/northern New Hampshire), Great Lakes (northern Minnesota), Northern Rockies/Cascades (Northern Montana/northeastern Idaho), Northern Cascades (Washington), Kettle/Wedge (Washington), and Greater Yellowstone Area (portions of Wyoming, Montana, and Idaho). There is one provisional area in the continental United States, which contains a reintroduced population in the southern Rocky Mountains (USFWS 2005). The revised critical habitat of the lynx designates approximately 100,891 square kilometers (km<sup>2</sup>) (38,954 square miles [sq. mi.]) of critical habitat in five units in the states of Idaho, Maine, Minnesota, Montana, Washington, and Wyoming. This rule consists of: (1) replacement of the existing state-boundary-based definition of the range of the lynx Distinct Population Segment (DPS) with a definition that extends the Endangered Species Act's protections to lynx "where found" in the contiguous United States; and (2) a final designation of revised critical habitat for the contiguous United States DPS of the Canada lynx (79 FR 54782).

**Distinct Population Segments Defined**

Yes: Designated March 24, 2000 (65 FR 16052). Clarification of findings for contiguous United States DPS: July 3, 2003 (68 FR 40076), January 10, 2007 (72 FR 1186).

**Critical Habitat Designated**

Yes; 11/9/2006.

**Legal Description**

On September 12, 2014, the U.S. Fish and Wildlife Service, finalized two actions: designating revised critical habitat for the contiguous United States distinct population segment of the Canada lynx (*Lynx canadensis*) under the Endangered Species Act of 1973, as amended, and revising the boundary of the Canada lynx distinct population segment. This rule revised critical habitat for the lynx and extended the Endangered Species Act's protections to the species wherever it occurs in the contiguous United States, including New Mexico.

**Critical Habitat Designation**

Five units are designated as critical habitat for the Canada lynx DPS. The critical habitat areas described below constitute our best assessment at this time of areas that meet the definition of critical habitat. The designated units are: Unit 1 in northern Maine (Aroostook, Franklin, Penobscot, Piscataquis, and Somerset Counties); Unit 2 in northeastern Minnesota (Cook, Koochiching, Lake, and St. Louis Counties); Unit 3 in the Northern Rocky Mountains of northwest Montana (Flathead, Glacier, Granite, Lake, Lewis and Clark, Lincoln, Missoula, Pondera, Powell and Teton Counties) and northeast Idaho (Boundary County); Unit 4 in the North Cascade Mountains of north-central Washington (Chelan and Okanogan Counties); and Unit 5 in the Greater Yellowstone Area of southwest Montana (Carbon, Gallatin, Park, Stillwater, and Sweetgrass Counties) and northwest Wyoming (Fremont, Lincoln, Park, Sublette, and Teton Counties). All units were occupied by lynx populations at the time of listing and are currently occupied by lynx populations.

Unit 1: Northern Maine Unit 1 consists of 10,123 mi<sup>2</sup> (26,218 km<sup>2</sup>) located in northern Maine in portions of Aroostook, Franklin, Penobscot, Piscataquis, and Somerset Counties. This area was occupied by the lynx at the time of listing and is currently occupied by the species (Hoving et al. 2003, entire; Vashon et al. 2012, pp. 12–14, 58–60; Interagency Lynx Biology Team 2013, pp. 39–42). This area contains the physical and biological features essential to the conservation of the

lynx DPS as it comprises the PCE and its components laid out in the appropriate quantity and spatial arrangement. Lynx in northern Maine have high productivity: 91 percent of available adult females (greater than 2 years) produced litters, and litters averaged 2.83 kittens (Vashon et al. 2005b, pp. 4–6; Vashon et al. 2012, p. 18). This area is also important for lynx conservation because it is the only area in the northeastern region of the lynx's range within the contiguous United States that currently supports a resident breeding lynx population and likely acts as a source or provides connectivity with Canada for more peripheral portions of the lynx's range in the Northeast. Timber harvest and management are the dominant land uses within the unit; therefore, special management may be required depending on the silvicultural practices implemented (68 FR 40075). Timber management practices that provide for a dense understory are beneficial for lynx and snowshoe hares. In this area, climate change is predicted to significantly reduce lynx habitat and population size. Carroll (2007, pp. 1100–1103) modeled a 59 percent decline in lynx numbers in the northeastern United States and eastern Canada by 2055 due to climate change, with greater vulnerability among small, peripheral, low-elevation populations like that in Maine. Under this modeled scenario, populations would have difficulty sustaining themselves, and the lynx distribution would likely contract to the core of the population on the Gaspé Peninsula in Quebec, Canada (Carroll 2007, p. 1102). Gonzalez et al. (2007, p. 14) modeled potential climate-induced loss of snow and concluded that snow suitable for lynx may disappear from Maine entirely by the end of this century. Therefore, climate change represents a potential habitat-related threat to lynx in this unit. Changing forest management practices are also likely to result in reduced hare and lynx habitat in this unit. Much of the lynx and hare habitat in this unit is the result of broad-scale clear-cut timber harvest in the 1970s and 1980s in response to a spruce budworm outbreak and the subsequent treatment of some clearcuts with herbicide to promote conifer regeneration. These clear-cut stands are now at a successional (regrowth) stage (about 35 years postharvest) that features very dense conifer cover and provides optimal hare and lynx habitats, likely supporting many more hares and lynx than occurred historically. The Maine Forest Practices Act (1989) limited the size of clearcuts, resulting in a near complete shift away from clearcuts to partial harvesting. This transition to partial harvest timber management is unlikely to create or maintain the extensive tracts of hare and lynx habitats that currently exist as a result of previous clearcutting. As the clear-cut stands continue to age, their habitat value to hares and lynx is expected to decline. Even in the absence of climate change considerations, forest succession and reduced clearcutting are expected to result in a substantially smaller lynx population in this unit by 2035 (Simons 2009, pp. 153–154, 162–165, 206, 216–220; Vashon et al. 2012, pp. 58–60). Therefore, the potential for forest management practices to result in reduced quantity and quality of lynx and hare habitats represents a habitat-related threat to lynx in this unit. Other potential habitat-related threats to lynx in this unit are habitat loss and fragmentation due to road and highway construction (along with associated increases in traffic volumes and/or speeds) and commercial, recreational, and wind-energy development. In this final rule, we have not designated critical habitat on Tribal lands in this unit nor on lands managed in accordance with the Natural Resources Conservation Service's Healthy Forest Reserve Program.

Unit 2: Northeastern Minnesota Unit 2 consists of 8,069 mi<sup>2</sup> (20,899 km<sup>2</sup>) located in northeastern Minnesota in portions of Cook, Koochiching, Lake, and St. Louis Counties, and Superior National Forest. In 2003, when we formally reviewed the status of the lynx, numerous verified records of lynx existed from northeastern Minnesota (68 FR 40076). The area was occupied at the time of listing and is currently occupied by the species (Moen et al. 2008b, pp. 29–32; Moen et al. 2010, entire; Catton and Loch 2010, entire; 2011, entire; 2012, entire;

Interagency Lynx Biology Team 2013, pp. 44–47). Lynx are currently known to be distributed throughout northeastern Minnesota, as has been confirmed through DNA analysis, radio- and GPS-collared animals, and documentation of reproduction (Moen et al. 2008b, entire; Moen et al. 2010, entire). This area contains the physical and biological features essential to the conservation of the lynx DPS as it comprises the PCE and its components laid out in the appropriate quantity and spatial arrangement. This area is essential to the conservation of lynx because it is the only area in the Great Lakes Region for which there is evidence of recent lynx reproduction. It likely acts as a source or provides connectivity for more peripheral portions of the lynx's range in the region. mstockstill on DSK4VPTVN1PROD with RULES2 VerDate Mar2010 19:18 Sep 11, 2014 Jkt 232001 PO 00000 Frm 00044 Fmt 4701 Sfmt 4700 E:\FR\FM\12SER2.SGM 12SER2 Federal Register / Vol. 79, No. 177 / Friday, September 12, 2014 / Rules and Regulations 54825 Timber harvest and management are dominant land uses (68 FR 40075). Therefore, special management may be required depending on the silvicultural practices implemented. Timber management practices that provide for a dense understory are beneficial for lynx and snowshoe hares. In this area, climate change may affect lynx and their habitats; however, Gonzalez et al. (2007, p. 14) suggested that snow conditions in northern Minnesota should continue to be suitable for lynx through the end of this century. Nonetheless, because climate change may alter vegetation communities and, hence, hare densities, it still represents a potential habitat-related threat to lynx in this unit. Fire suppression or fuels treatment, habitat fragmentation associated with roadbuilding (and associated increases in traffic volumes and/or speeds), and commercial, recreational, and energy/ mineral development pose other potential habitat-related threats to lynx in this unit. Incidental capture of lynx in traps set for other species has been documented recently in Minnesota, as have lynx mortalities from vehicle collisions (U.S. Fish and Wildlife Service 2013d, unpubl. database). The Service has not designated critical habitat on Tribal lands in this unit.

Unit 3: Northern Rocky Mountains Unit 3 consists of 9,783 mi<sup>2</sup> (25,337 km<sup>2</sup>) located in northwestern Montana and a small portion of northeastern Idaho in portions of Boundary County in Idaho and Flathead, Glacier, Granite, Lake, Lewis and Clark, Lincoln, Missoula, Pondera, Powell, and Teton Counties in Montana. It includes National Forest lands and BLM lands in the Garnet Resource Area. This area was occupied by lynx at the time of listing and is currently occupied by the species (Squires et al. 2010, entire; Squires et al. 2012, entire; Squires et al. 2013, entire; Interagency Lynx Biology Team 2013, pp. 57–61). Lynx are known to be widely distributed throughout this unit, and breeding has been documented in multiple locations (Gehman et al. 2004, pp. 24–29; Squires et al. 2004a, pp. 8– 10, 2004b, entire, and 2004c, pp. 7– 10). This area contains the physical and biological features essential to the conservation of the lynx DPS as it comprises the PCE and its components laid out in the appropriate quantity and spatial arrangement. This area is essential to the conservation of lynx because it appears to support the highest density lynx populations in the Northern Rocky Mountain region of the lynx's range. It likely acts as a source for lynx and provides connectivity to other portions of the lynx's range in the Rocky Mountains, particularly the Greater Yellowstone Area. Timber harvest and management are dominant land uses (68 FR 40075); therefore, special management may be required depending on the silvicultural practices implemented. Timber management practices that provide for a dense understory are beneficial for lynx and snowshoe hares. In this area, climate change is expected to result in the potential loss of snow conditions suitable for lynx by the end of this century (Gonzalez et al. 2007, p. 14). Therefore, climate change represents a potential habitat-related threat to lynx in this unit. Fire suppression or fuels treatment, habitat fragmentation associated with road-building (and associated increases in traffic volumes and/or

speeds), and commercial, recreational, and energy/mineral development pose other potential habitat-related threats to lynx in this unit. The Service has not designated critical habitat on Tribal lands in this unit nor on lands managed in accordance with the MDNRC HCP.

**Unit 4: North Cascades** Unit 4 consists of 1,834 mi<sup>2</sup> (4,751 km<sup>2</sup>) located in north-central Washington in portions of Chelan and Okanogan Counties and includes mostly Okanogan-Wenatchee National Forest lands as well as BLM lands in the Spokane District and Loomis State Forest lands. This area was occupied at the time lynx was listed and is currently occupied by the species (Interagency Lynx Biology Team 2013, pp. 64–65). This area contains the physical and biological features essential to the conservation of the lynx DPS as it comprises the PCE and its components laid out in the appropriate quantity and spatial arrangement. This unit supports the highest densities of lynx in Washington (Stinson 2001, p. 2). Evidence from recent research and DNA analysis shows lynx distributed within this unit, with breeding being documented (von Kienast 2003, p. 36; Koehler et al. 2008, entire; Maletzke et al. 2008, entire). Although researchers have fewer records in the portion of the unit south of Highway 20, few surveys have been conducted there. This area contains boreal forest habitat and the components essential to lynx conservation. Further, it is contiguous with the portion of the unit north of Highway 20, particularly in winter when deep snows close Highway 20. The northern portion of the unit adjacent to the Canada border also appears to support few recent lynx records; however, it is designated wilderness, so access to survey this area is difficult. This northern portion also contains extensive boreal forest vegetation types and the components essential to lynx conservation. Additionally, lynx populations exist in British Columbia directly north of this unit (Interagency Lynx Biology Team 2013, p. 65). This area is essential to the conservation of the lynx DPS because it is the only area in the Cascades region of the lynx's range that is known to support breeding lynx populations. Timber harvest and management are dominant land uses; therefore, special management may be required depending on the silvicultural practices implemented. Timber management practices that provide for a dense understory are beneficial for lynx and snowshoe hares. In this area, Federal land management plans are being amended to incorporate lynx conservation. Climate change is expected to reduce lynx habitat and numbers in this unit, with potential loss of snow suitable for lynx (Gonzalez et al. 2007, p. 14) and the potential complete disappearance of lynx from the area by the end of this century (Johnston et al. 2012, pp. 7–11). Therefore, climate change represents a potential habitat-related threat to lynx in this unit. Fire suppression or fuels treatment, habitat fragmentation associated with road-building (and associated increases in traffic volumes and/or speeds), and recreational and energy/mineral development pose other potential habitat-related threats to lynx in this unit. The Service has not designated critical habitat in this unit on lands managed in accordance with the WDNR Lynx Habitat Management Plan.

**Unit 5: Greater Yellowstone Area** Unit 5 consists of 9,146 mi<sup>2</sup> (23,687 km<sup>2</sup>) located in Yellowstone National Park and surrounding lands of the Greater Yellowstone Area in southwestern Montana and northwestern Wyoming. Lands in this unit are found in Carbon, Gallatin, Park, Stillwater, and Sweetgrass Counties in Montana; and Fremont, Lincoln, Park, Sublette, and Teton Counties in Wyoming. This area was occupied by lynx at the time of listing and is thought to be currently occupied by a small but persistent lynx population (Squires and Laurion 2000, entire; Squires et al. 2001, entire; Murphy et al. 2006, entire; Interagency Lynx Biology Team 2013, pp. 57–61). This area contains the physical and biological features essential to the conservation of the lynx DPS as it comprises the PCE and its components laid out in the appropriate quantity and spatial arrangement. The Greater Yellowstone Area is naturally

marginal lynx habitat with highly fragmented foraging habitat (68 FR 40090; 71 FR 66010, 66029; 74 FR 8624, 8643–8644; Hodges et al. 2009, entire). For this reason lynx home ranges in this unit are likely to be larger and incorporate large areas of non-foraging matrix habitat. Timber harvest and management are dominant land uses on National Forest System lands in this unit; therefore, special management may be required depending on the silvicultural practices implemented. Timber management practices that provide for a dense understory are beneficial for lynx and snowshoe hares. Climate change is expected to reduce lynx habitat and numbers in this unit, with potential loss of snow suitable for lynx over most of the area by the end of this century, though with potential snow refugia in the Wyoming Range Mountains (Gonzalez et al. 2007, p. 14). Therefore, climate change represents a potential habitat-related threat to lynx in this unit. Fire suppression or fuels treatment, habitat fragmentation associated with road-building (and associated increases in traffic volumes and/or speeds), and recreational and energy/mineral development pose other potential habitat-related threats to lynx in this unit. Therefore, special management is required depending on the fire suppression and fuels treatment practices conducted and the design of highway and energy development projects. The Service has not designated critical habitat in this unit on lands managed in accordance with the MDNRC HCP.

#### **Primary Constituent Elements/Physical or Biological Features**

Critical habitat units are designated for the following States and counties: (i) Idaho: Boundary County; (ii) Maine: Aroostook, Franklin, Penobscot, Piscataquis, and Somerset Counties; (iii) Minnesota: Cook, Koochiching, Lake, and St. Louis Counties; (iv) Montana: Carbon, Flathead, Gallatin, Glacier, Granite, Lake, Lewis and Clark, Lincoln, Missoula, Park, Pondera, Powell, Stillwater, Sweetgrass, and Teton Counties; (v) Washington: Chelan and Okanogan Counties; and (vi) Wyoming: Fremont, Lincoln, Park, Sublette, and Teton Counties. Within these areas the primary constituent element for the Canada lynx is boreal forest landscapes supporting a mosaic of differing successional forest stages and containing:

- (i) Presence of snowshoe hares and their preferred habitat conditions, which include dense understories of young trees, shrubs or overhanging boughs that protrude above the snow, and mature multistoried stands with conifer boughs touching the snow surface;
- (ii) Winter conditions that provide and maintain deep fluffy snow for extended periods of time;
- (iii) Sites for denning that have abundant coarse woody debris, such as downed trees and root wads; and
- (iv) Matrix habitat (e.g., hardwood forest, dry forest, non-forest, or other habitat types that do not support snowshoe hares) that occurs between patches of boreal forest in close juxtaposition (at the scale of a lynx home range) such that lynx are likely to travel through such habitat while accessing patches of boreal forest within a home range.

#### **Special Management Considerations or Protections**

Critical habitat does not include manmade structures (such as buildings, aqueducts, runways, roads, and other paved areas) and the land on which they are located existing within the legal boundaries on October 14, 2014.

In listing the lynx as threatened under the Act due to the inadequacy of existing regulatory mechanisms to ensure the conservation of the DPS, the Service recognized the need for special

management considerations or protection for lynx in the contiguous United States. The need for specific management direction and conservation measures for lynx was likewise recognized during development of the interagency Lynx Conservation Assessment and Strategy (LCAS; Ruediger et al. 2000, entire). The U.S. Forest Service (USFS), Bureau of Land Management (BLM), National Park Service, and the Service developed the LCAS using the best available science at the time specifically to provide a consistent and effective approach to conserve lynx and lynx habitat on Federal lands.

Lynx conservation depends on management that supports boreal forest landscapes of sufficient size to encompass the temporal and spatial changes in habitat and snowshoe hare populations to support interbreeding lynx populations over time.

Although lynx and hare habitats were likely created historically by natural forest disturbances (e.g., fire, insects and disease, and windthrow), the current extensive habitats in northern Maine are the result of large-scale industrial forest management. Maintaining lynx populations there will require forest management practices that produce extensive stands supporting high hare densities into the future. The Service developed Canada Lynx Habitat Management Guidelines for Maine (McCollough 2007, entire), which specify the special management— recommendations on land use, forest conditions, landscape conditions, and silviculture requirements—needed to support lynx populations based on the best available science.

Four northern Maine landowners with collective ownership of approximately 8.5 percent of occupied lynx habitat have developed lynx forest management plans through the Natural Resource Conservation Service's Healthy Forest Reserve Program. These landowners commit to employ the Service's lynx habitat management guidelines (McCollough 2007, entire), which include greater use of even-aged silviculture that creates large patches of high-quality.

### ***Life History***

#### **Feeding Narrative**

Adult: Canada lynx are crepuscular to diurnal ambush predators. They ambush small prey and are highly adapted for swift, agile movement on dense snow (NatureServe 2015). Mammals comprise the largest portion of the Canada lynx's diet, followed by birds. Snowshoe hares (*Lepus americanus*) make up the greatest biomass (35 to 99 percent) of prey consumed year-round (USFS 2013). Females caring for kittens were more active during the day compared to pre- or post-denning periods. In winter, males were most active during daylight hours (NPS 2013). Canada lynx are highly adapted for hunting snowshoe hare, their primary food resource, in the dense snows of the boreal forest (USFWS 2005). Lynx prey opportunistically on a wide variety of secondary prey when snowshoe hare are not readily available. Prey is readily available in the summer months, but is limited in the winter months. Secondary prey includes red squirrels (*Tamiasciurus hudsonicus*), grouse (*Bonasa umbellus*, *Dendragapus* sp., and *Lagopus* sp.), northern flying squirrel (*Glaucomys sabrinus*), ground squirrels (*Spermophilus parryi*, *S. richardsonii*, and *Urocitellus columbianus*), porcupine (*Erethizon dorsatum*), beaver (*Castor canadensis*), mice (*Peromyscus* sp.), voles (*Microtus* sp.), shrews (*Sorex* sp.), weasels (*Mustela* sp.), fish, and ungulates as carrion. Male lynxes have been known to opportunistically kill white-tailed deer (*Odocoileus virginianus*) and mule deer (*Odocoileus hemionus*) when deep snow hindered deer movements and increased their vulnerability to predation (NPS 2013; USFWS 2013). When hare numbers are low, lynx experience widespread food shortages; many die of

starvation or abandon home ranges to search for adequate prey (USFWS 2013). Lynx are highly mobile and have a propensity to disperse long distances, particularly when prey becomes scarce (USFWS 2005). Bobcats (*Lynx rufus*), coyotes (*Canis latrans*), and mountain lions (*Felix concolor*) compete directly with Canada lynx for food (NPS 2013).

### **Reproduction Narrative**

Adult: The Canada lynx has moderate fitness and reproductive capacity, and may live up to 16 years in the wild (NPS 2013). Canada lynx breed from late winter to early spring (March to April) in North America (NatureServe 2015; NPS 2013). It is unclear when females and males attain sexual maturity, but most research indicates that breeding does not occur until the second year of life (USFS 2013). Male lynxes may be incapable of breeding during their first year (NPS 2013). Some females give birth as yearlings, but their pregnancy rate is lower than that of older females (NatureServe 2015). Adult females produce one litter every 1 to 2 years, with gestation lasting between 60 and 65 days (NatureServe 2015; USFS 2013). Litters of generally two or three kittens (but potentially as many as five) are born from May to July in the contiguous United States. Females that have lost their litter may produce a second litter, born in August (NPS 2013). Reproductive success is directly correlated to the snowshoe hare abundance (USFS 2013). Litters of four or five and high kitten survival are common when hare numbers are high; when they are low, little or no reproduction may occur (USFWS 2013). Prey scarcity suppresses breeding, and may result in mortality of nearly all young (NatureServe 2015). For 1 to 2 years following a snowshoe hare decline, Canada lynx birth rate declines. Adult females may continue to conceive, but live births are few or none (USFS 2013). Den sites tend to be in mature or old-growth stands with a high density of logs (NatureServe 2015). Natal dens are typically located under large logs that provide protection for kittens. Kitten survival directly correlates with hare abundance. Canada lynx kittens remain with their mother for 9 to 10 months following birth to nurse and learn how to hunt (USFS 2013). Male lynxes do not take part in parental care (NPS 2013).

### **Geographic or Habitat Restraints or Barriers**

Adult: Roads and other human-made structures were used as boundaries for critical habitat where they clearly delineated areas with confirmed records of lynx and the presence of the physical and biological features essential to lynx. If climate change results in landscape-scale reductions in hare densities, some areas that currently support lynx populations may become less capable of doing so, and lynx could decline or disappear from these areas regardless of the diversity or abundance of alternate prey species. Such climate-induced impacts to hare habitats and populations could be accompanied by projected reductions in snow quantity (79 FR 54782).

### **Spatial Arrangements of the Population**

Adult: Clumped according to resources; random (USFWS 2013).

### **Environmental Specificity**

Adult: Narrow/specialist.

### **Tolerance Ranges/Thresholds**

Adult: Moderate

### **Site Fidelity**

Adult: High

**Dependency on Other Individuals or Species for Habitat**

Adult: Boreal forest vegetation, snow conditions, and abundant snowshoe hare prey base constitute preferred lynx habitat (NPS 2013).

**Habitat Narrative**

Adult: Canada lynx, which demonstrate moderate ecological integrity and thresholds and have high site fidelity, are narrow specialists, evolutionarily adapted for hunting on snow. Lynx thrive in dense snowfall, a key factor for suitable snowshoe hare habitat (NPS 2013). Lynx need persistent deep, powdery snow, which limits competition from other hare predators; their denning habitat generally consists of log piles, windfalls, or dense vegetation that provide security for kittens (USFWS 2013). Lynx occur in mesic coniferous forests that have cold, snowy winters and provide a prey base of snowshoe hare (USFWS 2005). In the United States, lynx thrive in conifer and conifer-hardwood forests that support snowshoe hares, their primary prey (USFWS 2005). Lynx have been documented to use both coniferous and mixed coniferous/deciduous vegetation types dominated by spruce (*Picea* sp.), balsam fir (*Abies balsamea*), pine (*Pinus* sp.), northern white cedar (*Thuja occidentalis*), eastern hemlock (*Tsuga canadensis*), sugar maple (*Acer saccharum*), aspen (*Populus tremula*), and paper birch (*Betula papyrifera*). Snow-tracking revealed that lynx avoid large openings, demonstrating their dependence on the dense understory of boreal forests (NPS 2013). In the western United States, 83 percent of lynx occurrences are associated with Rocky Mountain Conifer Forest, and most (77 percent) fall within the 1,500- to 2,000-m (4,920- to 6,560-ft.) elevation zone. Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), and lodgepole pine (*Pinus contorta*) forest cover types occurring on cold, moist potential vegetation types provide suitable habitat for lynx (NPS 2013). Boreal forest vegetation, snow conditions, and abundant snowshoe hare prey base constitute preferred lynx habitat (NPS 2013). These boreal forests include a dense, multi-layered understory that optimizes cover and foraging at both ground level and at varying snow depths throughout the winter. Despite the variety of habitats and settings, good habitat for snowshoe hare and, therefore, Canada lynx, have this common factor: dense, horizontal vegetative cover 1 to 3 m (3 to 10 ft.) above the ground or snow level (USFWS 2005). In northern habitats, snow depths are relatively uniform and only moderately deep, with total annual snowfall of 100 to 127 cm (39 to 50 in.) (NPS 2013). In the southern portion of lynx range, snow depths are deeper, with deepest snows in the mountains of southern Colorado. A landscape density of greater than 0.5 hare per ha (0.2 hare per ac.) is required to sustain lynx within their home ranges (NPS 2013). Geographic and habitat barriers of the Canada lynx include roads and other human-made structures used as boundaries for critical habitat where they clearly delineated areas with confirmed records of lynx and the presence of the physical and biological features essential to lynx. If climate change results in landscape-scale reductions in hare densities, some areas that currently support lynx populations may become less capable of doing so, and lynx could decline or disappear from these areas regardless of the diversity or abundance of alternate prey species. Such climate-induced impacts to hare habitats and populations could be accompanied by projected reductions in snow quantity. Climate-induced impacts to hare habitats and populations could be accompanied by projected reductions in snow quantity, quality, and duration, thereby reducing the competitive advantage lynx have over other hare predators in the areas that currently support lynx populations (79 FR 54782).

**Dispersal/Migration**

**Motility/Mobility**

Adult: High mobility.

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory (NatureServe 2015)

**Dispersal**

Adult: High

**Immigration/Emigration**

Adult: Immigrates/emigrates.

**Dependency on Other Individuals or Species for Dispersal**

Adult: Large, boreal forested landscape with adequate snowshoe hare density (USFWS 2005).

**Dispersal/Migration Narrative**

Adult: Canada lynx have high mobility and are nonmigratory feline species. They have been observed to have moderate dispersal and moderate immigration/emigration. In North America, the distribution of lynx is nearly coincident with that of snowshoe hares. Immigration of lynx from Canada plays a vital role in sustaining lynx in the contiguous United States (USFWS 2005). Dispersal of Canada lynxes typically occurs when juveniles disperse from their natal area, or when lynxes disperse as a response to snowshoe hare declines. Kittens remain with their mother through their first winter, and natal dispersal occurs from late April to early May. Maximum natal dispersal distance for females is 9.7 km (6.0 mi.). Canada lynxes are capable of long-range exploratory movements of up to 1,000 km (600 mi.) (USFS 2013). Individual lynxes maintain large home ranges between 31 to 216 km<sup>2</sup> (12 to 83 sq. mi.). Because lynxes require a large home range, the population can only persist in large boreal forested areas that contain appropriate forest types, snow depths, and high densities of snowshoe hare. Northeast lynxes are most likely to occur in areas that support deep snow (greater than 268 cm [106 in.] annual snowfall). This area is associated with regenerating boreal forests in landscapes with an area of 100 km<sup>2</sup> (40 sq. mi.) or greater. Areas with smaller patches of boreal forest are unlikely to provide suitable habitat to support a lynx population. It has been reported that female lynxes tend to establish home ranges adjacent to their mother (USFWS 2005). When prey is scarce, the highly mobile lynx has a propensity to disperse long distances. Little is known about preferred habitat outside of the home range. Dispersing lynx may colonize suitable but unoccupied habitats, augment existing resident populations, or disperse to unsuitable or marginal habitats where they cannot survive. Numerous lynx mortality records exist from anomalous habitats or habitats where no records support evidence (either current or historical) of a reproducing population (USFWS 2005).

**Additional Life History Information**

Adult: Dispersal of Canada lynxes typically occurs when juveniles disperse from their natal area, or when lynxes disperse as a response to snowshoe hare declines. Kittens remain with their mother through their first winter, and natal dispersal occurs from late April to early May. Maximum natal dispersal distance for females is 9.7 kilometers (km) (6.0 miles [mi.]). Canada lynxes are capable of long-range exploratory movements of up to 1,000 km (600 mi.) (USFS 2013). Individual lynxes maintain large home ranges between 31 to 216 km<sup>2</sup> (12 to 83 sq. mi.). Because lynxes require a large home range, the population can only persist in large boreal

forested areas that contain appropriate forest types, snow depths, and high densities of snowshoe hare. Northeast lynxes are most likely to occur in areas that support deep snow (greater than 268 cm [106 in.] annual snowfall). This area is associated with regenerating boreal forests in landscapes with an area of 100 km<sup>2</sup> (40 sq. mi.) or greater. Areas with smaller patches of boreal forest are unlikely to provide suitable habitat to support a lynx population. It has been reported that female lynxes tend to establish home ranges adjacent to their mother (USFWS 2005).

***Population Information and Trends*****Population Trends:**

Stable (USFWS 2005)

**Species Trends:**

Species level trends are stabilizing, but are highly dependent on the availability of snowshoe hares (USFWS 2005).

**Resiliency:**

Moderate

**Representation:**

Low

**Redundancy:**

Moderate

**Population Growth Rate:**

Stable (NPS 2013)

**Number of Populations:**

Fourteen extant contiguous population (USFWS 2005).

**Resistance to Disease:**

Moderate

**Adaptability:**

Low

**Additional Population-level Information:**

Lynx survivorship, productivity, and population dynamics are closely related to snowshoe hare density in all parts of its range. A minimum density of snowshoe hares (greater than 0.5 hare per ha [1.2 hares per ac.]) distributed across a large landscape is necessary to support survival of lynx kittens and recruitment into and maintenance of a lynx population (USFWS 2005).

**Population Narrative:**

The current Canada lynx population is stable, but exact populations numbers remain undescribed. The contiguous United States population is composed of 14 extant populations, in Colorado, Idaho, Maine, Michigan, Minnesota, New Hampshire, New York, Oregon, Montana,

Utah, Vermont, Washington, Wisconsin, and Wyoming (USFWS 2005). The cyclic or fluctuating nature of lynx populations provides an additional element of uncertainty in assessing population trends. The entire population is experiencing a stabilizing trend despite adverse interactions with humans, loss of habitat, challenges of global warming, inadequacy of regulatory mechanisms, and incidental mortalities caused by legal trapping. With the lynx's moderate resiliency and redundancy, with low representation, the population remains stable and could possibly increase with attention to existing regulatory mechanisms (USFWS 2013). Lynx in the contiguous United States and neighboring Canadian provinces interact as metapopulations; therefore, assessments of population viability must be made at this larger scale and not solely based on populations in the contiguous United States. Immigration of lynx into the contiguous United States is believed important to sustaining persistent lynx populations in core areas adjacent to Canada; therefore, contiguous United States lynx populations might be negatively affected if trapping reduces the numbers of emigrating lynx. Lynx survivorship, productivity, and population dynamics are closely related to snowshoe hare density in all parts of its range. A minimum density of snowshoe hares (greater than 0.5 hare per ha [1.2 hares per ac.]), distributed across a large landscape, is necessary to support survival of lynx kittens and maintenance of a lynx populations (USFWS 2005). Although snowshoe hare populations in the southern portion of the range in the contiguous United States may fluctuate, they do not show strong, regular population cycles as in the north in the contiguous United States. The degree to which regional local lynx population fluctuations are influenced by local snowshoe hare population dynamics is unclear (USFWS 2005). However, studies found that lynx responded to the increase in hare numbers with approximately four-fold increases in their population sizes, followed by a three- to four-fold decrease during the decline phase of the cycle. During a cyclic decline in hare numbers, lynx demonstrate lower survival than during any other phase in the cycle (NPS 2013).

### ***Threats and Stressors***

**Stressor:** Habitat or range destruction, modification, or curtailment

**Exposure:** Indirect

**Response:** Decreased availability of suitable habitat, leading to increased interactions between lynxes and humans.

**Consequence:** Increased mortality.

**Narrative:** Human alteration of the distribution and abundance, species composition, successional stages, and connectivity of forests results in changes in the forest's capacity to sustain lynx populations. People change forests through timber harvest, fire suppression, and conversion of forest lands to agriculture. Forest fragmentation may eventually become severe enough to isolate habitat into small patches, thereby reducing the viability of wildlife that are dependent on larger areas of forest habitat. Destruction, modification, or curtailment of habitat or range has a large effect on the success of Canada lynx populations. In all regions within the range of lynx in the contiguous United States, timber harvest, recreation, and their related activities are the predominant land use affecting lynx habitat. The primary factor that caused the Canada lynx to be listed was the lack of guidance for conservation of lynx and snowshoe hare habitat in National Forest Land and Resource Plans and Bureau of Land Management (BLM) Land Use Plans, given that a substantial amount of lynx habitat in the contiguous United States is federally managed. This lack of guidance allowed the continued degradation of lynx habitat on federal lands through timber management and other federal activities. Except for lynx habitat management plans on some private and state lands in Washington, in the remainder of the

contiguous United States range there are no management plans that specifically address lynx conservation (65 FR 16052; USFWS 2005). This stressor has resulted in decreased availability of suitable habitat, leading to increased interactions between lynxes and humans.

**Stressor:** Incidental trapping

**Exposure:** Direct; trapping, snaring, and hunting.

**Response:** See narrative.

**Consequence:** Mortality

**Narrative:** The final rule (65 FR 16052) and remanded final rule (72 FR 1186) found that despite concerns that overtrapping had severely depressed the United States populations of lynx, low numbers of lynx in the contiguous United States compared to northern Canada occur not as a result of historical overtrapping in the United States, but because lynx and their prey are naturally limited by the amount of habitat, topography, and climate. Precautions have been implemented by states to restrict lynx trapping since the 1980s, but legal trapping, snaring, and hunting for bobcat, coyote, wolverine, and other furbearers create a potential for incidental capture or shooting of lynx (USFWS 2005). Lynx persist throughout their range despite the incidental catch that presumably has occurred throughout the past, probably at higher levels than presently. Although there is concern about the potential for death of lynxes that are incidentally captured, there is no information to indicate that the loss of these individuals has negatively affected the overall ability of lynx in the contiguous United States to persist. Individuals may be lost, which could affect small local populations (USFWS 2005). Lynx trapping in Canada, where lynx are a legally harvested furbearer, may affect rates of lynx immigration into the contiguous United States. Immigration of lynx into the contiguous United States is believed important to sustaining persistent lynx populations in core areas adjacent to Canada; therefore, contiguous United States lynx populations might be negatively affected if trapping reduces the numbers of emigrating lynx (USFWS 2005).

**Stressor:** Inadequacy of existing regulatory mechanisms

**Exposure:** Indirect

**Response:** Poor regulations, habitat disturbance, and interactions with humans.

**Consequence:** Mortality

**Narrative:** As a result of federal, state, and tribal regulations and plans that conserve lynx, in particular the Forest Service and BLM Lynx Conservation Agreements and the revision of some Forest Plans, the threats to lynx from the inadequacy of existing regulatory mechanisms have been reduced since the lynx was listed. However, establishment of consistent guidance that provides adequate regulatory mechanisms over the longer term is needed throughout the range of the lynx. Similarly, plans to conserve lynx habitat and provide long-term conservation of lynx in the Northeast are currently lacking. The Maine Forest Practices Act has significantly changed silvicultural practices from clearcutting to partial harvesting, which may not create conditions that are beneficial to lynx and snowshoe hares (USFWS 2005).

**Stressor:** Roads and snow trails

**Exposure:** Direct; caused by interactions with humans along roadside during wintering season.

**Response:** Reduced fitness; and inability to compete for resources, find mates, and defend territory.

**Consequence:** Increased mortality.

**Narrative:** It is critical that landscape connectivity between lynx habitats and populations in Canada and the contiguous United States be maintained. Lynx movements may be adversely

influenced by high traffic volume on roads that bisect suitable lynx habitat, such as in the Southern Rockies. At this time, there is no evidence that competition between lynx and potential competitors such as coyotes (*Canis latrans*) and bobcats (*Lynx rufus*), if it exists, exerts a population-level impact on lynx (USFWS 2005). The theory that compacted snow trails and roads that are maintained for winter recreation and forest management facilitate competition by giving other species, particularly coyotes, access to lynx winter habitat has neither been proven nor disproven at this time (USFWS 2005).

**Stressor:** Hybridization

**Exposure:** Bobcat and Canada lynx interface.

**Response:** Hybridization; see narrative.

**Consequence:** Unknown; hybridization.

**Narrative:** The ranges of lynx and bobcat naturally interface in the contiguous United States. The range of bobcats is limited by snow conditions that provide a competitive advantage to lynx. In 2003, lynx-bobcat hybridization was first documented in Minnesota and has since been documented elsewhere in the Great Lakes and the Northeast. Whether lynx-bobcat hybridization has implications for lynx conservation is unknown at this time (USFWS 2005).

**Stressor:** Climate change

**Exposure:** Direct and indirect.

**Response:** Decrease in suitable habitat and prey availability.

**Consequence:** Population decline.

**Narrative:** Scientific evidence has demonstrated that the climate has been warming globally, as evidenced by changes in the amount of snow cover, among other indicators. Continued warming temperatures are likely to negatively affect the cold climatic conditions that create and maintain the boreal forest ecosystem for which lynx are highly adapted. As a result, continued warming trends may eventually cause the boreal forests in the contiguous United States to recede north and/or recede to higher, colder elevations, which would likely result in adverse effects to the contiguous United States population of lynx (USFWS 2005).

## ***Recovery***

### **Reclassification Criteria:**

Reclassification/uplisting criteria have not been established for this species. A final recovery plan for the Canada lynx has not been published.

### **Delisting Criteria:**

Specific recovery objectives, delisting criteria, and actions will be developed in the course of the formal recovery planning process and as additional data become available for analysis. The U.S. Fish and Wildlife Service has concluded that it is not practicable at this time to establish demographic criteria for delisting the species (USFWS 2005). Recovery of the Canada lynx will be achieved when conditions have been attained that will allow lynx populations to persist long-term in each (of the identified) core areas (USFWS 2005).

### **Recovery Actions:**

- A final recovery plan for the Canada lynx has not been published. The Canada lynx final recovery outlines provide a preliminary recovery strategy for the species. The recovery of the Canada lynx will be achieved when conditions have been attained that will allow lynx

populations to persist long-term in each of the identified core areas. The preliminary recovery objectives and measures for calculating progress toward the recovery goal of delisting the lynx—as well as the recommended 12 recovery actions to attain that goal—are presented below, with the understanding that all are subject to change as new information is gathered (USFWS 2005).

- Establish management commitments in core areas that will provide for adequate quality and quantity of habitat, so that there is a reasonable expectation that persistent lynx populations can be supported in each of the core areas for at least the next 100 years (USFWS 2005).
- Maintain baseline inventories of lynx habitat in each core area, monitoring changes in structure and the distribution of habitat components (USFWS 2005).
- Monitor lynx use in lynx analysis units or other appropriate management unit at least once every 10 years to determine distribution and occupancy in the core area (USFWS 2005).
- Identify habitat facilitating movement between each core area and lynx populations in Canada (USFWS 2005).
- Ensure that habitat in secondary areas remains available for occupancy by lynx (USFWS 2005).
- Identify population and habitat limiting factors for lynx in the contiguous United States (USFWS 2005).
- Identify population and habitat limiting factors for lynx in the contiguous United States (USFWS 2005).
- Develop a post-delisting monitoring plan that will be in place and ready for implementation prior to delisting (USFWS 2005).

***Conservation Measures and Best Management Practices:***

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***Additional Threshold Information:***

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## SPECIES ACCOUNT: *Martes caurina* (Humboldt marten)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Threatened

#### **Physical Description**

Medium-sized carnivore that has a long and narrow body type typical of the mustelid family (e.g., weasels, minks, otters, and fishers): Overall brown fur with distinctive coloration on the throat and upper chest that varies from orange to yellow to cream, large and distinctly triangular ears, and a bushy tail that is proportionally equivalent to about 75 percent of the head and body length.

#### **Taxonomy**

We consider for listing the (previously-classified) subspecies Humboldt marten (*Martes americana humboldtensis*), or the (now-recognized) subspecies Humboldt marten (*M. caurina humboldtensis*), or the Humboldt marten DPS of the Pacific marten (*M. caurina*). Populations of marten from coastal Oregon (considered members of *M. a. caurina*) are proposed as more closely related to *M. a. humboldtensis* than to *M. a. caurina* in the Cascades of Oregon (citing Dawson 2008, Slauson et al. 2009a), justifying that the range of the subspecies or DPS of the Humboldt marten should be expanded to include coastal Oregon populations of martens. However, the species and its nomenclature has remained under review and the current 12-month finding relies on the April 7, 2015, DPS analysis (80 FR 18742) concluding that Pacific martens in coastal Oregon and northern coastal California were both discrete and significant and constituted a listable entity referred to collectively as the “coastal DPS of the Pacific marten.” The 12-month finding and the associated SSA reflect analysis of that DPS. Preliminary results of genetic evaluation of the Pacific marten indicate that coastal Oregon and northern coastal California marten populations likely represent a single subspecies (Slauson et al. 2009a, pp. 1338-1339; Schwartz et al. 2016, unpublished report) but the taxonomic change has not yet been published. In this case, our listable entity may be a subspecies, but the analysis maintains its validity. (83 FR 50574) The preferred nomenclature in the SSA (FWS July 2018) is Coastal Marten (*Martes caurina*). The SSA and the listing proposal synthesizes the biology and status of the distinct population segment (DPS) of the Pacific marten (*Martes caurina*) in coastal Oregon and northern coastal California, commonly referred to as the coastal marten.

#### **Historical Range**

Historically occurred throughout the coastal forests of northwestern California and Oregon.

#### **Current Range**

Coastal marten historically ranged throughout coastal Oregon and coastal northern California, but the species has not recently been detected throughout much of the historical range, despite extensive surveys. The species currently exists in four small (<100) populations and is absent from the northern and southern ends of its historical range. This current range is approximately 7.3 percent of its known historical range, with two populations in Oregon and two populations in California. The species has been extirpated from Sonoma and Mendocino Counties, CA, and largely from Humboldt, Del Norte, and Siskiyou Counties, CA. In Oregon, coastal martens have been largely extirpated from much of the inland counties within the historical range and are known to currently occur in Coos, Curry, Josephine, Douglas, Lane, and Lincoln Counties.

**Critical Habitat Designated**

Yes;

***Life History*****Food/Nutrient Resources****Food Source**

Adult: Opportunistic

**Reproductive Strategy**

Adult: Polygamous

**Lifespan**

Adult: 5 year in wild; various reports of up to 7 years

**Breeding Season**

Adult: June through August, with delayed implantation

**Reproduction Narrative**

Adult: North American martens are polygamous, with females solely responsible for raising young. Information on timing of marten mating is largely known from the behavior of captive animals, but is believed to occur from late June to early August, with a peak in July (Markley and Bassett 1942, pp. 606–607). Females typically give birth in March and April (Strickland et al. 1982, p. 602) (Figure 2.4). Females do not mate until 15 months of age and, due to delayed implantation, will not produce their first litters until they are at least 24 months old (Strickland et al. 1982, p. 601). However, not all yearling females produce ova. Thompson and Colgan (1987, p. 831) reported less than 25 percent of yearlings produced ova, and Fortin and Cantin (2004, pp. 228– 229) reported a range of 44–76 percent ovulation rate (females >1.5 years) for greater than 183 American martens over a decade. Not all females of reproductive age give birth in any given year. In Ontario, Thompson and Colgan (1987, p. 831) reported a 50 percent pregnancy rate during years of environmental stress. However, in the Sierra Nevada Mountains of California, from 2009–2011, of 20 females that were at least 2 years old, all were lactating annually, indicating that they were all actively involved in attempting to raise litters (Slauson 2017, p. 13); this time period included one of the top ten snowfall years in the Sierra Nevada Mountains over the last century, suggesting adult females in the Sierras attempt to produce litters annually, regardless of winter conditions and its effects on prey populations. The limited data available for coastal martens show 75 percent of females reproducing, with a mean litter size of 1.8 (0.6) kits (Moriarty 2018, pers. comm.). (USFWS 2018)

**Habitat Type**

Adult: Forest

**Spatial Arrangements of the Population**

Adult: The coastal marten needs to have multiple resilient populations distributed throughout its range to provide for redundancy. The more populations, and the wider the distribution of those populations, the more redundancy the species exhibits. Based on the distributions of

current verifiable marten detections and adjacent suitable habitat, we identified four extant population areas (EPAs) within coastal Oregon and northern coastal California: (1) Central Coastal Oregon Extant Population Area; (2) Southern Coastal Oregon Extant Population Area; (3) Oregon-California Border Extant Population Area; and (4) Northern Coastal California Extant Population Area. Additional detections of coastal martens have occurred outside of the current EPAs but they did not meet the criteria of a population (most likely, they represent transient individuals in search of new territories) according to methods used in the Humboldt Marten Conservation Strategy and Assessment

**Habitat Narrative**

Adult: Martens tend to select older forest stands (e.g., late-successional, old-growth, large-conifer, mature, late-seral, structurally complex). These forests have a mixture of old and large trees, multiple canopy layers, snags and other decay elements, dense understory development, and biologically complex structure and composition.

***Dispersal/Migration*****Motility/Mobility**

Adult: While some adult male and female martens leave their home ranges during periods of low prey densities (Thompson and Colgan 1987, pp. 830–831), overall the prevalence of adults leaving their established home ranges is low.

**Dispersal**

Adult: While dispersal distances of more than 70 km (43 mi) have been reported for martens (e.g., Fecske and Jenks 2002, p. 310), this is rare and most studies find that the majority of juvenile martens dispersed <15 km (9.3 mi) (<15 km in Maine by Phillips 1994, pp. 73–75; <5 km in Ontario by Broquet et al. 2006, p. 1694; 15.5 km in Alaska and British Columbia by Pauli et al. 2012, p. 393; 10.8 km in California by Slauson 2017, p. 143). The limited data we have for dispersal events of coastal marten suggest that dispersal distances are similar (K. Slauson 2018, pers. comm.). Habitat conditions greatly influence dispersal.

***Population Information and Trends*****Population Trends:**

Declining

**Number of Populations:**

4: Central Coastal Oregon, Southern Coastal Oregon, CA-OR Border, Northern Coastal CA

**Population Size:**

Central Coastal Oregon, 71; Southern Coastal Oregon, 12-<100; CA-OR Border, 12-<100; Northern Coastal CA, 80-100

***Threats and Stressors***

**Stressor:** Habitat destruction

**Exposure:** Increasing potential interactions and subsequent marten injury, mortality, or predation.

**Response:****Consequence:**

**Narrative:** Our analysis of the past, current, and future influences on what the coastal marten needs for long-term viability revealed that two factors pose the largest risk to future viability of the species. These risks are primarily related to habitat loss and associated changes in habitat quality and distribution and include: (1) A decrease in connectivity between populations; and (2) habitat conversion from that suitable for martens to that suitable for generalist predators and competitors, thereby increasing potential interactions and subsequent marten injury, mortality, or predation. These factors are all influenced by vegetation management, wildfire, and changing climate.

**Stressor:** Rodenticides used in Marijuana cultivation

**Exposure:****Response:****Consequence:**

**Narrative:** Exposure to rodenticides (Factor E) through direct ingestion or the consumption of exposed prey has lethal and sub-lethal effects on coastal martens. Illegal marijuana cultivation sites on public, tribal, and private forest lands are implicated as the likely source of these rodenticides. In a similar carnivore species, 85% of carcasses tested were exposed to rodenticides, with the exposure in 13% being the direct cause of death.

**Stressor:** Road corridors

**Exposure:****Response:****Consequence:**

**Narrative:** Since 1980, 19 mortalities of coastal martens caused by vehicles (Factor E) have been documented, all in Oregon and mostly along U.S. Highway 101. We expect that some unknown amount of marten roadkills go undetected, so this is likely an underestimate of the number of martens killed by cars.

**Stressor:** Disease

**Exposure:****Response:****Consequence:**

**Narrative:** Certain diseases (Factor C) are also a concern to martens and other carnivore populations, including canine distemper viruses (CDV), rabies viruses, parvoviruses, and the protozoan (single-celled organism) *Toxoplasma gondi*.

***Recovery******Conservation Measures and Best Management Practices:***

- Forest Management Practices:

**References**

Threatened Species Status for Coastal Distinct Population Segment of the Pacific Marten (Proposed Rule) October 9, 2018. USFWS. (2018). Species Status Assessment for the Coastal Marten (*Martes caurina*) Version 2.0. July 2018 U.S. Fish and Wildlife Service Region 8 Arcata, CA (141p)

Threatened Species Status for Coastal Distinct Population Segment of the Pacific Marten (Proposed Rule). October 9, 2018 USFWS. (2018). Species Status Assessment for the Coastal Marten (*Martes caurina*) Version 2.0. July 2018 U.S. Fish and Wildlife Service Region 8 Arcata, CA (141p)

83 FR 50574: Endangered and Threatened Wildlife and Plants

Threatened Species Status for Coastal Distinct Population Segment of the Pacific Marten (Proposed Rule). October 9, 2018

Threatened Species Status for Coastal Distinct Population Segment of the Pacific Marten. October 9, 2018.

## **SPECIES ACCOUNT: *Microtus californicus scirpensis* (Amargosa vole)**

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### ***Species Taxonomic and Listing Information***

**Listing Status:** Endangered; November 15, 1984 (49 FR 45160).

### **Physical Description**

The Amargosa vole (*Microtus californicus scirpensis*) is a stout-bodied, almost cylindrical, compact mouse. The comparatively short tail, small rounded ears, and short legs easily distinguish it from most other small mouse-like rodents. The Amargosa vole averages 20.3 centimeters (cm) (8 inches [in.]) in total length. Tail length averages about 6.3 cm (2.5 in.). Observed weights for male and female Amargosa voles average 72 grams (g) (2.54 ounces [oz.]) and 59.73 g (2.11 oz.), respectively. Coloration is bright brown, ranging from cinnamon buff to buckthorn brown. Distinguishing characteristics include the bright pelage coloration, and a small skull with a comparatively large zygomatic width (USFWS 1997; USFWS 2009).

### **Taxonomy**

The Amargosa vole is one of 17 subspecies of the California vole (*M. californicus*). The taxon was originally described as a distinct species, *Microtus scirpensis*, based on seven specimens collected near Shoshone. Kellogg (1918), in revising the *californicus* species group in and adjacent to California, reassigned the scientific name *Microtus californicus scirpensis*, and provided the vernacular name Amargosa meadow mouse. This is the currently recognized scientific name in the literature (USFWS 1997). Characteristics that distinguishing the Amargosa vole from other voles include the bright fur coloration, and a small skull with a comparatively large zygomatic width (USFWS 2009).

### **Historical Range**

The historical range of the Amargosa vole was limited to wetland pockets extending from the desert community of Shoshone, Inyo County; to the Amargosa Canyon, Inyo County, California. The largely subterranean Amargosa River and an associated series of small tributary springs maintain an isolated 16-kilometer (10-linear-mile) stretch of perennial surface water (USFWS 1997).

### **Current Range**

Occurs in highly localized and isolated wetlands along the Amargosa River in Inyo County California, from the vicinity of Shoshone to the upper end of Amargosa Canyon near Tecopa (NatureServe 2015; USFWS 2009). It has been estimated that no more than one square kilometer (km<sup>2</sup>) (247 acres [ac.]) of uneven patchy habitat for this species remains (CDFW 2016).

### **Distinct Population Segments Defined**

No

### **Critical Habitat Designated**

Yes; 11/15/1984.

### **Legal Description**

On November 15, 1984, the Service determined endangered status and critical habitat for the Amargosa vole.

**Critical Habitat Designation**

The critical habitat of the Amargosa vole falls within an overall zone of 4,520 acres in southeastern Inyo County, California. Within this zone, the critical habitat consists of marshes and associated land and water along the Amargosa River, from just north of Tecopa Hot Springs to the Amargosa Canyon, just south of Tecopa.

**Primary Constituent Elements/Physical or Biological Features**

The areas designated as critical habitat satisfy all known criteria for the ecological, behavioral, and physiological requirements of the species. The marsh vegetation (primarily bulrush) provides sufficient cover for escape from predators as also serves as a food source.

Within these areas, the major constituent elements that are known to require special management considerations or protection are marsh vegetation (primarily bulrushes of the genus *Scirpus*), springs, and some open water along the Amargosa River, which provide escape cover and an adequate food supply.

**Special Management Considerations or Protections**

In the case of the Amargosa vole, such activities that may adversely modify habitat include burning or otherwise removing marsh vegetation, overgrazing of marsh or adjacent vegetation, pumping of ground water supplies, diverting or channelizing springs or the Amargosa River, road repair work, off-road vehicle use in or adjacent to marsh areas, use of herbicides or rodenticides, introduction of exotic plant or animal species, and exploration for and exploitation of geothermal resources.

***Life History*****Feeding Narrative**

Adult: The Amargosa vole is a granivore and herbivore that eats various forbs and grasses, preferring developing seeds and tender leaves over mature seeds. When seasonally available, green emergent vegetation comprises the bulk of the diet; grass seeds predominate in the diet during the summer and autumn (USFWS 1997). Activity rates of the Amargosa vole are moderate; activity usually occurs in daylight hours during winter months, although animals may become crepuscular and nocturnal through the summer (USFWS 1997). Cyclic vole population explosions may result in intensive intraspecific competition for available resources, as observed with California vole populations in Contra Costa County, California. However, there is not enough available information to determine whether these types of cyclic population eruptions and crashes occur in the Amargosa vole subspecies (USFWS 2009). The Amargosa vole is dependent on desert wetland vegetation dominated by chairmaker's bulrush (*Schoenoplectus americanus*). These wetlands usually exist as small, isolated patches, rarely exceeding 2 ha (5 ac.) (CDFW 2016). Additionally, California voles require regular intake of large amounts of water, meeting or exceeding 10 percent of body weight per day (USFWS 1997, 2009). Voles are primary consumers and often the principal herbivores in occupied habitats. They may excavate an extensive underground network of runways and tunnels, and in dense cover frequently develop extensive surface runways (USFWS 2009).

**Reproduction Narrative**

Adult: Little is known about the biology and the life history of the Amargosa vole. Most of the information provided is based on information about the other California vole subspecies. Social systems of subspecies of California vole reportedly range from monogamy to polygamy (USFWS 2009). Reproductive maturity is reached when females attain a weight of 25 to 31.1 g (0.9 to 1.1 oz.) and males a weight of 34 to 39.6 g (1.2 to 1.4 oz.) (USFWS 1997). California voles may produce as many as five litters per year, and their gestation period is 21 days (BLM 2014). In central California, litter sizes increase from about three at the beginning of the breeding season in the fall, to a peak of about six in the spring; the mean litter size is 4.7 (USFWS 2009). The young are weaned after 14 days (USFWS 1997). The life expectancy for most California vole subspecies is short; the average longevity of adult males and adult females observed in east San Francisco Bay was 8 weeks and 12.5 weeks, respectively (USFWS 1997). However, some observations indicate that Amargosa voles may live longer than just a few months, and researchers have observed one Amargosa vole that lived for at least 1 year. Reproduction may occur at any time of year, but it is primarily influenced by factors such as temperature and precipitation that determine availability of food and water. A greater percentage of males and females were observed in reproductive condition in June than in November. The age structure of Amargosa vole populations changes significantly throughout the year, with the proportion of adults increasing and the proportion of subadults and juveniles decreasing between June and November (USFWS 2009).

**Geographic or Habitat Restraints or Barriers**

Adult: Limited to wetland pockets that are not subjected to regular inundation during heavy summer thunderstorms (USFWS 1997).

**Spatial Arrangements of the Population**

Adult: Clumped

**Environmental Specificity**

Adult: Narrow/specialist.

**Tolerance Ranges/Thresholds**

Adult: Low

**Site Fidelity**

Adult: High

**Dependency on Other Individuals or Species for Habitat**

Adult: Closely associated with marshes dominated by chairmaker's bulrush (*Schoenoplectus americanus*) (BLM 2014).

**Habitat Narrative**

Adult: Amargosa voles occur in isolated herbaceous wetland habitats where chairmaker's bulrush (*Schoenoplectus americanus*) is a dominant perennial overstory species subjected to regular inundation during heavy summer thunderstorms. These wetlands form discontinuous narrow bands along the Amargosa River, broken by more characteristic desert vegetation dominated by creosote bush (*Larrea tridentata*), burrobush (*Ambrosia dumosa*), and desert holly (*Atriplex hymenelytra*) (USFWS 1997). Fragmentation of the isolated wetland pockets creates a habitat barrier (USFWS 1997). Key resources that are needed for habitat include marsh

vegetation (primarily bulrushes of the genus *Schoenoplectus* sp.), springs, and some open water along the Amargosa River (49 FR 45160). The Amargosa vole constructs burrows, and frequently moves around near their entrances (NatureServe 2015). The species' primary association is with bulrush in wet or lightly-flooded substrates, and most areas of high vole abundance occur at the interface between bulrush and saltgrass (*Distichlis spicata*) habitats (USFWS 2009).

***Dispersal/Migration*****Motility/Mobility**

Adult: Low

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Low

**Immigration/Emigration**

Adult: Unlikely

**Dependency on Other Individuals or Species for Dispersal**

Adult: No

**Dispersal/Migration Narrative**

Adult: Mobility and dispersal of the Amargosa vole are limited by the fragmented pockets of suitable wetland habitat that exist along the Amargosa River. Additionally, California vole home range size is typically small. The species has a tendency to remain in a restricted area. Dispersal distances of up to 120 m (400 ft.) were recorded for a comparatively small proportion of the marked California voles during a 1966 study (USFWS 1997; USFWS 2009).

**Additional Life History Information**

Adult: California vole home range size is typically small. The species has a tendency to remain in a restricted area. Dispersal distances of up to 120 meters (m) (400 feet [ft.]) were recorded for a comparatively small proportion of the marked California voles during a 1966 study (USFWS 1997).

***Population Information and Trends*****Population Trends:**

Since 2012, the population trend for the Amargosa vole at Marsh 1 has declined overall. The density of Marsh 1 has decreased 80 percent (February) and 81 percent (May) from 2012 to 2015, respectively. Consistently in 2013-2015, the greatest population sizes are in June-September, while the lowest population numbers are in February-May. These data indicate that the population continues to decline and shows an annual pattern of greater numbers in the summer than winter. A population viability analysis of the Marsh 1 population suggests a mean expected time to extinction of 4 years, with a 25 percent probability of population loss within 12 months (Foley and Foley 2016). Because all other known marshes support far fewer voles than what used to be present at Marsh 1, loss of this population dramatically increases the

extirpation probability of the entire species. A population viability analysis conducted for the entire species (Foley and Foley 2016) predicted a time to extinction of 20 to 24 years, with a 4-5 percent probability of extinction within 12 months due to environmental stochasticity. In this study the more common California vole was used as a surrogate for the Amargosa vole due to a lack of long-term, multi-year demographic data for the species (USFWS 2019).

**Species Trends:**

Unknown (USFWS 2009)

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Number of Populations:**

Unknown (USFWS 2009)

**Population Size:**

Klinger et al. (2014) indicated that the population size of Marsh 1 ranged between 5 and 116 individuals in 2013 and 2014 depending on the season (Table 1), with the estimate for overall population size (all marshes) for 2013 and 2014 ranging from a low of 69 individuals in March 2013 to a high of 426 individuals in March 2014 (USFWS 2019).

**Resistance to Disease:**

Recent trapping efforts in September 2010 revealed a high prevalence of deformities resulting from necrotic skin disease carried by tiny parasitic mites. This is the first documented disease for the Amargosa vole: a host of other parasites as well as viral and bacterial pathogens are potentially transmitted from extant and invasive animals in the Amargosa ecosystem (CDFW 2016).

**Adaptability:**

Low (CDFW 2016)

**Additional Population-level Information:**

It is difficult to accurately assess the current status of the species with the limited information available on its distribution and abundance. There are few data available on the relative abundance of Amargosa voles. Only six reports on trapping surveys are available, and only four of these trapped in more than a handful of sites. In addition, not all investigators used a common trapping season, so any population cycling in a given year is likely to confound conclusions regarding real population trends (USFWS 2009).

**Population Narrative:**

It is difficult to accurately assess the current status of the species with the limited information available on its distribution and abundance. There are few data available on the relative

abundance of Amargosa voles. Only six reports on trapping surveys are available, and only four of these trapped in more than a handful of sites. In addition, not all investigators used a common trapping season, so any population cycling in a given year is likely to confound conclusions regarding real population trends. Impacts have likely resulted in the loss of the species from its type locality in Shoshone, California; another population (site 40) in the Tecopa area appears to have been lost since the time of listing. However, it also appears likely that there are three new populations of Amargosa vole that were not present at the time of listing. Although only a few surveys have been conducted since listing, eight additional populations have been consistently occupied (USFWS 2009). Recent trapping efforts in September 2010 revealed a high prevalence of deformities resulting from necrotic skin disease carried by tiny parasitic mites. This is the first documented disease for the Amargosa vole: a host of other parasites as well as viral and bacterial pathogens are potentially transmitted from extant and invasive animals in the Amargosa ecosystem (CDFW 2016).

### ***Threats and Stressors***

**Stressor:** Burning of marsh habitat

**Exposure:** Burning on private land, and wildfires.

**Response:** Loss of habitat.

**Consequence:** Risk of local extirpation of Amargosa vole populations.

**Narrative:** Burning of marsh habitat for pasture land was noted as a factor for this subspecies' listing in 1984. Since that time, there has been a decrease in widespread use of fire to clear marsh vegetation. However, burning on private lands near Tecopa Hot Springs has occurred as recently as 2008, and continues to be a localized threat (USFWS 2009).

**Stressor:** Diversion of spring flows, and alteration of historical marsh configuration

**Exposure:** Water diversion and groundwater pumping.

**Response:** Habitat degradation and removal.

**Consequence:** Population decline.

**Narrative:** Diversion of spring flows and alteration of historical marsh configuration due to new and/or unresolved threats associated with groundwater pumping, salt cedar invasion, and diversions and other manmade barriers to natural spring flow persist. The diversion of spring flow directly reduces the herbaceous wetland habitat that the Amargosa vole requires. This may further contribute to declines in the remaining populations (USFWS 2009).

**Stressor:** Establishment of salt cedar (*Tamarix* sp.)

**Exposure:** Salt cedar (*Tamarix* spp.).

**Response:** Habitat degradation.

**Consequence:** Population decline.

**Narrative:** Establishment of salt cedar (*Tamarix* spp.) in the Amargosa River drainage has continued to diminish available Amargosa vole habitat through replacement of bulrush marshes. This may contribute to further population decline (USFWS 2009).

**Stressor:** Predation

**Exposure:** Domestic cats.

**Response:** Mortality

**Consequence:** Population decline.

**Narrative:** Predation by domestic cats in the Tecopa and Tecopa Hot Springs area is a potential source of mortality to the Amargosa vole (USFWS 2009).

**Stressor:** Inadequacy of existing regulatory mechanisms

**Exposure:** Spring diversions and marsh burning.

**Response:** Reduced quantity and quality of available habitat.

**Consequence:** Population decline.

**Narrative:** Spring diversions and marsh burning on private land are not within the regulatory control of National Environmental Policy Act or Federal Land Policy and Management Act. These factors continue to affect Amargosa vole habitat, causing reductions in both the quality and quantity of available habitat (USFWS 2009).

**Stressor:** Climate change

**Exposure:** Warmer air temperatures, more intense precipitation events, and increased summer continental drying.

**Response:** May alter habitat.

**Consequence:** Population decline.

**Narrative:** Climate change may affect the Amargosa vole's wetland habitat as a result of prolonged drought. This could place increasing stress on remaining Amargosa vole populations, and could lead to further declines in remaining populations (USFWS 2009).

**Stressor:** Disease threats

**Exposure:** Mites that are a larval trombiculid in the genus *Neotrombicula* are prevalent in wild voles, and can cause severe skin lesions and deformities.

**Response:** While the broader effects on vole populations are unknown, infection may have negative consequences such as reduced individual health and fitness.

**Consequence:** Population decline.

**Narrative:** Research since 2009 has demonstrated that disease may be a potential threat that could diminish the health of Amargosa voles (USFWS 2019).

**Stressor:** Inbreeding depression

**Exposure:** Genetic consequences of small, fragmented populations and low genetic diversity and a limited amount of gene flow.

**Response:** Low genetic diversity may similarly reduce the fitness of the Amargosa vole.

**Consequence:** Population decline.

**Narrative:** Loss of genetic diversity in small populations may decrease the potential for persistence in the face of long-term environmental change (Shaffer 1981, Shaffer 1987, Primack 1998). Loss of genetic diversity can also result in decline in fitness from expression of deleterious recessive alleles (Meffe and Carroll 1994). (USFWS 2019).

### **Recovery**

#### **Reclassification Criteria:**

The number, configuration, size, and quality of available habitat patches are conserved and managed and support at least two independently functioning populations in core areas within the range (USFWS 2019)

Habitat where each population from criterion A.1 occurs has a secure water source for its exclusive use. The independently functioning populations (i.e., those in core areas) are separated to a sufficient degree, so that they are buffered from stochastic threats (e.g. wildfire) and other threats that could transfer between marshes (e.g. disease). (USFWS 2019)

Disease dynamics and potential effects to the Amargosa vole are well understood at the individual and population scale such that the threat of disease is sufficiently managed or ameliorated (USFWS 2019)

The mean winter population size is stable or increasing for a period of at least 10 years. A minimum winter (December-January) density of 11.4 voles per hectare in core area #1 and 2.5 voles per hectare in either core area #2 or #3 is maintained during this time (USFWS 2019).

**Delisting Criteria:**

The number, configuration, size, and quality of available habitat patches is conserved and managed to support at least three independently functioning populations in core areas (as described in A1) within the range for 20 years (USFWS 2019)

New habitat is established that is geographically disconnected from the Tecopa marshes, with successful translocation of Amargosa voles to the new marshes. This will reduce the likelihood that a single catastrophic event (e.g., fire, drought) will extirpate the species. Translocated populations will be monitored for a minimum of 5 years to establish average density. This established vole density will serve as the minimum limit for translocated populations to maintain for a period of at least 20 years. This follows the rationale used in A.3 and E.2 for measuring long-term population stability (USFWS 2019)

A management plan is implemented for invasive plant control to protect habitat for the Amargosa vole and is prioritized in core management areas (as identified in A.1) and newly established areas outside of the Tecopa Marshes (USFWS 2019).

Impacts from invasive predators (e.g., domestic cats) are reduced or managed to levels that do not pose a threat to the persistence of the Amargosa vole (USFWS 2019).

At least three independently functioning populations are naturally reproducing and are stable or increasing in numbers over a period of 20 years. A minimum winter (December-January) density of 11.4 voles per hectare in core area #1 and 2.5 voles per hectare in both core area #2 and #3 is maintained during this time (USFWS 2019).

A regional water conservation management plan is developed and implemented, and the water level in marshes where the Amargosa vole occurs, is considered to be stable or increasing for a minimum of 20 years, through multiple drought cycles (USFWS 2019).

**Recovery Actions:**

- Secure all extant wetland habitats, while focusing priority on upland areas containing core vole populations. Secured lands will be managed to maintain viable vole populations and maximize habitat conditions through protection of spring sources and control of exotic and/or competitive species and incompatible uses (USFWS 1997).
- Survey the population and obtain basic life history information (USFWS 1997).

- Quantify habitat characteristics and determine temporal and spatial patterns of use (USFWS 1997).
- Complete genetic analyses. Genetic information is needed from contemporary and historic populations (USFWS 1997).
- Enhance Amargosa vole populations and habitat. This may include reintroduction of the vole into historic habitat (USFWS 1997).
- Monitor habitat trends (USFWS 1997).
- The U.S. Fish and Wildlife Service (USFWS) should work with the California Department of Fish and Wildlife (CDFW), the Bureau of Land Management (BLM), The Nature Conservancy, and the Amargosa Conservancy to continue acquisition of fee title or conservation easements on private lands that contain Amargosa vole habitat (USFWS 2009).
- BLM, CDFW, and USFWS should continue to participate in interstate forums regarding the groundwater issues in the Death Valley Regional Flow System, and monitor county and municipal plans for groundwater development in southern Nevada (especially Pahrump Valley) (USFWS 2009).
- USFWS and CDFW should identify private landowners in the Tecopa and Tecopa Hot Springs area who have Amargosa vole habitat on their lands, and notify them of the potential adverse effects from burning of bulrush vegetation (USFWS 2009).
- USFWS, CDFW, Inyo County, and BLM should investigate the magnitude of the impact associated with domestic cat predation, and take steps to educate residents in Tecopa and Tecopa Hot Springs of the threat that domestic cats pose to this species. If necessary, Inyo County should explore the need for an ordinance to prohibit free-roaming cats in this area (USFWS 2009).
- USFWS, CDFW, and BLM should develop a spring discharge monitoring program for all springs that support Amargosa vole habitat to track changes in water volume (USFWS 2009).
- BLM, CDFW, and USFWS should work with their partners to remove salt cedar from all existing Amargosa vole habitat and from areas where its removal may result in regeneration of historic bulrush marsh habitats (USFWS 2009).
- USFWS and CDFW should initiate a range-wide monitoring effort for this species of sufficient intensity to determine trends in occupancy for all suitable Amargosa vole habitat (USFWS 2009).
- BLM, CDFW, and USFWS should investigate the potential for restoring bulrush corridors and/or intermediate habitats between existing occupied habitats to promote increased gene flow between these fragmented populations (USFWS 2009).
- USFWS and CDFW should investigate the magnitude of the impact related to interspecific competition with house mice and take steps to control it, if necessary (USFWS 2009).
- BLM, CDFW, and USFWS should investigate the potential impacts that the Tonopah and Tidewater Railroad Grade has on bulrush marsh distribution, and determine whether breaching of this barrier would create additional Amargosa vole habitat (USFWS 2009).
- Construct mega marshes (connection of multiple marshes or patches of habitat to create larger areas of suitable habitat) to provide additional habitat for Amargosa voles (USFWS 2019).
- Enhance habitat quality in existing marshes (USFWS 2019).
- Conduct environmental education and public outreach (USFWS 2019).
- Secure local water sources (USFWS 2019).
- Proceed in land acquisition and/or development of conservation easements with key, willing landowners that have critical marshes, spring sources, or flow paths (USFWS 2019).

- More effectively use and distribute water from spring and well outflows to increase their utility in supporting marsh habitat quantity and quality (USFWS 2019).
- Maintain a captive breeding colony (USFWS 2019).
- Implement population augmentation in existing and future marsh habitat (USFWS 2019).
- Develop a regional water use management plan (USFWS 2019).

***Conservation Measures and Best Management Practices:***

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***Additional Threshold Information:***

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## **SPECIES ACCOUNT: *Microtus pennsylvanicus dukecampbelli* (Florida salt marsh vole)**

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### ***Species Taxonomic and Listing Information***

**Commonly-used Acronym:** FSMV

**Listing Status:** Endangered; January 14, 1991; Southeast Region (R4)

### **Physical Description**

The Florida salt marsh vole is a small (178-198 millimeters [7.0-7.9 inches] in length), short-tailed rodent with a blunt head and short ears. The fur is black-brown dorsally and dark gray ventrally. *Microtus pennsylvanicus dukecampbellii* is a subspecies of the widespread meadow vole (*M. pennsylvanicus*). *Microtus pennsylvanicus dukecampbelli* differs from *M. pennsylvanicus* in its larger size, darker coloration, relatively smaller ears, and certain skull characteristics (USFWS, 1997).

### **Taxonomy**

Southeastern-most of more than 25 subspecies; disjunct from other subspecies by 500 km (NatureServe, 2015).

### **Historical Range**

See Current Range/Distribution

### **Current Range**

Known from one marsh (Island Field Marsh) along the shore of Waccasassa Bay on Florida's Gulf coast (Levy County) (Woods 1992) (NatureServe, 2015).

### **Critical Habitat Designated**

Yes;

### ***Life History***

### **Feeding Narrative**

Adult: They feed on a variety of plant matter, including bark, grass, roots, and seeds. Few individuals live longer than 6 months (USFWS, 1997).

### **Reproduction Narrative**

Adult: Voles have a high reproductive rate and breed throughout most of the year with a peak of breeding activity occurring in the spring (Golley 1962). Woods et al. (1982) also reported breeding activity for Florida salt marsh voles during the spring. The gestation period for voles is 21 days, and the average litter size is 5 young. The young mature rapidly and start breeding when about 2 months of age (Golley 1962). The lifespan of voles is short, typically, few individuals live longer than 6 months (Golley 1960) (USFWS, 1997).

### **Environmental Specificity**

Adult: High (inferred from USFWS, 1997)

**Tolerance Ranges/Thresholds**

Adult: Low (inferred from USFWS, 1997)

**Site Fidelity**

Adult: High (inferred from USFWS, 1997)

**Habitat Narrative**

Adult: The vole is known to occur only at the type locality in salt marsh habitat where the vegetation is dominated by salt grass (*Distichlis spicata*) with smooth cordgrass (*Spartina alterniflora*) and glasswort (*Salicornia* spp.) also present (Woods et al. 1982). During a 1996 survey at the type locality, this vegetation was dense and 45-60 centimeters (17.7-23.6 inches) high in most places (T. Doonan pers. Comm.). Only a few areas along the Gulf coast salt marshes sampled supported habitat with similar characteristics. Florida salt marsh vole may need the dense, matted ground-level vegetation that these areas provide (T. Doonan pers comm.) (USFWS, 1997). High environmental specificity, ecological integrity and site fidelity are inferred based on specific habitat needs and is low tolerance range (USFWS, 1997).

***Dispersal/Migration*****Motility/Mobility**

Adult: High (NatureServe, 2015)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migrant (NatureServe, 2015)

**Dispersal**

Adult: Low (Inferred from USFWS, 1997)

**Immigration/Emigration**

Adult: Unlikely (Inferred from USFWS, 1997)

**Dispersal/Migration Narrative**

Adult: Mice and voles are highly mobile species. NatureServe (2015) notes that this species is non-migratory and low dispersal and unlikely immigration/emigration are inferred based on this species specific habitat requirements (USFWS, 1997).

***Population Information and Trends*****Population Trends:**

Not available

**Resiliency:**

Low (inferred from USFWS, 1997 and NatureServe, 2015)

**Representation:**

Low (inferred from USFWS, 1997 and NatureServe, 2015)

**Redundancy:**

Low (inferred from USFWS, 1997 and NatureServe, 2015)

**Number of Populations:**

One (NatureServe, 2015)

**Population Size:**

Unknown (NatureServe, 2015)

**Population Narrative:**

There is one known population and population size is unknown (NatureServe, 2015). Low resiliency, representation and redundancy are inferred based on having a single population and specific habitat requirements.

***Threats and Stressors***

**Stressor:** Climate change (USFWS, 2008)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Long-term climate change will continue to affect the extent of salt marsh available to the FSMV along the central Gulf coast of Florida. A future sea level rise of 13 - 30 cm due to climate change is predicted by the end of the century, which would significantly change the coastal marshes and adjacent uplands of Florida (USFWS, 2008).

**Stressor:** Habitat destruction or modification (USFWS, 2008)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** While much of the coastal marshes near the FSMV locations are publicly owned and managed for conservation, effective habitat management options are not understood. There is potential habitat located on privately owned lands, which are under pressure to develop along coastal areas near Cedar Key and along Florida's Gulf coast. The FSMV is at risk from extinction from catastrophic storm events that impact habitat, and may not maintain densities necessary to persist through storm events and seasonal fluctuations of resources (USFWS, 2008).

**Stressor:** Storms and hurricanes (USFWS, 2008)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Catastrophic weather events, hurricanes, or other strong storm systems that cause extreme high water events are primary threats to the FSMV. The Gulf coast salt marshes are very vulnerable to flooding because they are low lying and the adjacent Gulf waters are very shallow. While the FSMV has been able to maintain adult populations under the extreme conditions of tropical storm systems, the FSMV continues to be at risk of extinction due to the natural cyclic nature of vole populations and its limited distribution (USFWS, 2008).

***Recovery***

**Reclassification Criteria:**

Prevent extinction by maintaining the existing population (USFWS 2008).

**Delisting Criteria:**

The one (1) FSMV metapopulation exhibits a stable or increasing population trend for multiple generations, and natural recruitment (Factor A and E) (USFWS 2018).

When, in addition to the above criteria, it can be demonstrated that despite sea level rise and other environmental influences, sufficient suitable habitat remains for FSMV to remain viable for the foreseeable future. (Factor A and E) (USFWS 2018).

***Conservation Measures and Best Management Practices:***

- Acquire and protect the FSMV habitat at the original type locality and adjacent salt marsh habitats (USFWS, 2008).
- Establish an ongoing monitoring program at the two known locations of occurrence (USFWS, 2008).
- Conduct presence/absence surveys in potential FSMV habitat identified on LSNWR and other areas (Raabe and Gauron 2005) (USFWS, 2008).
- Map potential FSMV habitat on Chassahowitzka NWR, St. Marks NWR, and other public lands between these refuges. Conduct presence/absence surveys in potential FSMV habitat identified (USFWS, 2008).
- Update recovery plan and develop objective, measurable recovery criteria (USFWS, 2008).
- Develop management techniques to manage for, optimize existing and create FSMV habitat (i.e. saltgrass marsh) (USFWS, 2008).
- Evaluate establishing a captive breeding program for augmentation of existing populations and reintroductions as a recovery tool. Partner with land managers of publicly owned lands with potential FSMV habitat as potential reintroduction sites (USFWS, 2008).

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## SPECIES ACCOUNT: *Mustela nigripes* (Black-footed ferret)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered/Experimental Population (entire WY); 03/11/1967, 08/21/1991; Mountain-Prairie Region (R6) (USFWS, 2016)

### **Physical Description**

The black-footed ferret is a slim-bodied, medium-sized weasel (mustelid), typically weighing 645 to 1,125 grams (1.4 to 2.5 pounds) and measuring 47.9 to 60.0 centimeters (19 to 24 inches) in total length. Upper body parts are yellowish buff and occasionally whitish; feet and tail tip are black; and a black “mask” occurs across the eyes (USFWS 2013).

### **Taxonomy**

The black-footed ferret is in the order Carnivora, family Mustelidae, genus *Mustela*, and subgenus *Putorius*. The species is one of four members of the genus *Mustela* in North America, and the only ferret species native to the Americas (USFWS 2013).

### **Historical Range**

The black-footed ferret is endemic to North America. The historical habitat of the black-footed ferret coincided with the ranges of the black-tailed prairie dog (*Cynomys ludovicianus*), Gunnison’s prairie dog (*Cynomys gunnisoni*), and white-tailed prairie dog (*Cynomys leucurus*). These prairie dog species collectively occupied approximately 40.5 million ha (100 million ac.) of intermountain and prairie grasslands, extending from Canada into Mexico and including the states of Arizona, Colorado, Kansas, Montana, Nebraska, New Mexico, North Dakota, Oklahoma, South Dakota, Texas, Utah, Wyoming; the provinces of Alberta and Saskatchewan; and, through extrapolation, northern Mexico (USFWS 2013). A desktop analysis of the likely distribution of prairie dog habitat where the black-footed ferret probably occurred historically in the United States suggests that 85 percent of all ferrets may have occurred in black-tailed prairie dog habitat, 8 percent in Gunnison’s prairie dog habitat, and 7 percent in white-tailed prairie dog habitat, reinforcing the assumption that most ferrets probably occurred in black-tailed prairie dog habitat, based on the more expansive extent of their distribution (USFWS 2013).

### **Current Range**

The black-footed ferret was considered extremely rare before a small population was located in Mellette County, South Dakota, in 1964. Breeding attempts with a few captured animals failed to produce surviving young. The last wild animals in the Mellette population were observed in the field in 1974. The last captive animal from the Mellette population died at Patuxent Wildlife Research Center in 1979, and the ferret was presumed extinct. In 1981, a remnant population was discovered near Meeteetse, Wyoming. Following disease outbreaks at Meeteetse in the early 1980s, all surviving wild ferrets at Meeteetse were removed between 1985 and 1987. These ferrets were used to initiate a captive breeding program. Of the 18 remaining ferrets captured from Meeteetse, 15 individuals, representing the genetic equivalent of seven distinct founders, produced a captive population lineage that is the foundation of present recovery efforts. Extant populations, both captive and reintroduced, descend from these “founder” animals. No wild populations of black-footed ferrets have been found following the final capture of the last known Meeteetse ferret in 1987, despite extensive and intensive searches throughout the historic range of the ferret. It is very unlikely that any undiscovered wild

populations remain. To date, current black-footed ferret reintroduction sites (listed in chronological order of implementation) include: (1) Shirley Basin, Wyoming (1991); (2) Badlands National Park, South Dakota (1994); (3) UL Bend National Wildlife Refuge (NWR), Montana (1994); (4) Conata Basin, South Dakota (1996); (5) Aubrey Valley, Arizona (1996); (6) Fort Belknap Indian Reservation, Montana (1997); (7) Coyote Basin, Utah (1999); (8) Cheyenne River Indian Reservation, South Dakota (2000); (9) Wolf Creek, Colorado (2001); (10) BLM “40 Complex,” Montana (2001); (11) Janos, Chihuahua, Mexico (2001); (12) Rosebud Indian Reservation, South Dakota (2004); (13) Lower Brule Indian Reservation, South Dakota (2006); (14) Wind Cave National Park, South Dakota (2007); (15) Espee Ranch, Arizona (2007); (16) Logan County, Kansas (2007); (17) Northern Cheyenne Indian Reservation, Montana (2008); (18) Vermejo Ranch (black-tailed prairie dog habitat), New Mexico (2008); (19) Grasslands National Park, Saskatchewan, Canada (2009); and (20) Vermejo Ranch (Gunnison’s prairie dog habitat), New Mexico (2012) (USFWS 2013).

**Distinct Population Segments Defined**

None

**Critical Habitat Designated**

Yes;

***Life History*****Feeding Narrative**

Adult: The black-footed ferret is a carnivore that specializes in prairie dogs (>90 percent) and other small mammals (e.g. ground squirrels, cottontail rabbits, and deer mice) and birds (USFWS 2013; NatureServe 2015). Ferrets are nocturnal predators active especially during the nighttime after the moon is above the horizon (USFWS 2013). Given their preferred prey base, black-footed ferrets select for areas in prairie dog colonies that contain high burrow densities and therefore high densities of prairie dogs required for food (prey base) and shelter (burrows) (USFWS 2013). Young ferrets (kits) reach adult weight after 125 days (USFWS 2013).

**Reproduction Narrative**

Adult: Black-footed ferrets depend on prairie dogs and their dens, modifying and occupying existing burrows for food and shelter (USFWS 2013). Ferrets reach sexual maturity at approximately 1 year of age, and begin breeding between mid-March and early April. Litters averaging 3.5 individuals (range 1 to 5) are born below ground within 45 days and require parental care (Hillman and Clark 1980). Ferret young (kits) appear above ground in July, and disperse from their mothers in September or October. Life expectancy information is sparse, but the mean life expectancy of free-ranging ferrets in the Meeteetse population was 0.9 year. In the wild, females have reached 5 years of age, and males have reached 4 years. The Association of Zoos and Aquariums recommends an optimum sex ratio for recovery planning of 90 male to 150 female, equivalent to a sex ratio of 3:5 (USFWS 2013).

**Geographic or Habitat Restraints or Barriers**

Adult: The majority of ferret populations historically occurred in black-tailed prairie dog habitat, because the ferret is nearly completely dependent on prairie dogs, for both food and shelter (USFWS 2013).

**Spatial Arrangements of the Population**

Adult: Clumped (in association with prairie dog habitat).

**Environmental Specificity**

Adult: Narrow/specialist.

**Site Fidelity**

Adult: High

**Dependency on Other Individuals or Species for Habitat**

Adult: Ferrets select areas in prairie dog colonies that contain high burrow densities and therefore high densities of prairie dogs required for food (prey base) and shelter (burrows) (USFWS 2013).

**Habitat Narrative**

Adult: The black-footed ferret inhabits grasslands, steppe, and shrub steppe habitats in the Great Plains, mountain basins, and semi-arid grasslands of North America, in areas with prairie dog colonies that contain high burrow densities and therefore high densities of prairie dogs required for food (prey base) and shelter (burrows) (NatureServe 2015, USFWS 2013). In high-density black-tailed prairie dog habitat, the home ranges of female ferrets average approximately 60 ha (148 ac.), whereas males average approximately 130 ha (321 ac.). Territories—defended areas within an animal's home range—average 13 ha (32 ac.) for females and 36 ha (89 ac.) for males, and contain higher burrow densities than the rest of the home range (USFWS 2013).

***Dispersal/Migration*****Motility/Mobility**

Adult: Moderate—constrained by range of prairie dogs.

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Low

**Immigration/Emigration**

Adult: Unlikely

**Dependency on Other Individuals or Species for Dispersal**

Adult: Ferrets select areas in prairie dog colonies that contain high burrow densities and therefore high densities of prairie dogs required for food (prey base) and shelter (burrows) (USFWS 2013).

**Dispersal/Migration Narrative**

Adult: Black-footed ferrets select areas in prairie dog colonies that contain high burrow densities and therefore high densities of prairie dogs required for food (prey base) and shelter (burrows) (USFWS 2013), and are typically nonmigratory. Dispersal among young from natal areas (and

occasionally adults) occurs predominantly in the fall months, at distances of between 49 km (30 mi.) and 20 km (12 mi.). Males tend to move and disperse more than females (USFWS 2013). At present, all wild ferret populations are the result of reintroduction projects. In two cases (Badlands National Park in South Dakota and Conata Basin in South Dakota; and Coyote Basin in Utah, and Wolf Creek in Colorado), two different reintroductions subsequently merged into a single biological population (USFWS 2013).

**Additional Life History Information**

Adult: Dispersal among young from natal areas (and occasionally adults) occurs predominantly in the fall months, at distances of between 49 kilometers (km) (30 miles [mi.]) and 20 km (12 mi.). Males tend to move and disperse more than females (USFWS 2013). In two cases (Badlands National Park in South Dakota and Conata Basin in South Dakota; and Coyote Basin in Utah, and Wolf Creek in Colorado), two different reintroductions subsequently merged into a single biological population (USFWS 2013).

***Population Information and Trends*****Population Trends:**

Unknown

**Species Trends:**

Unknown

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Moderate

**Population Growth Rate:**

Unknown

**Number of Populations:**

There have been 20 specific black-footed ferret reintroduction projects, beginning in 1991; they have had varying levels of success. In two cases, two different reintroductions have since merged into one biological population: Badlands National Park in South Dakota, reintroduced in 1994, with Conata Basin in South Dakota, reintroduced in 1996; and Coyote Basin in Utah, reintroduced in 1999, with Wolf Creek in Colorado, reintroduced in 2001 (USFWS 2013).

**Population Size:**

250 to 1,000 individuals (NatureServe 2015). Approximate minimum population sizes at known release sites (as of 2011) are as follows: (1) Shirley Basin, Wyoming (203); (2) Badlands National Park, South Dakota (33); (3) UL Bend NWR, Montana (20); (4) Conata Basin, South Dakota (72); (5) Aubrey Valley, Arizona (75); (6) Fort Belknap Indian Reservation, Montana (0); (7) Coyote Basin, Utah (3); (8) Cheyenne River Indian Reservation, South Dakota (25); (9) Wolf Creek,

Colorado (No Data); (10) BLM “40 Complex,” Montana (No Data); (11) Janos, Chihuahua, Mexico (No Data); (12) Rosebud Indian Reservation, South Dakota (No Data); (13) Lower Brule Indian Reservation, South Dakota (12); (14) Wind Cave National Park, South Dakota (46); (15) Espee Ranch, Arizona (No Data); (16) Logan County, Kansas (38); (17) Northern Cheyenne Indian Reservation, Montana (No Data); (18) Vermejo Ranch (black-tailed prairie dog habitat), New Mexico (5); (19) Grasslands National Park, Saskatchewan, Canada (12); and (20) Vermejo Ranch (Gunnison’s prairie dog habitat), New Mexico (No Data) (USFWS 2013).

**Minimum Viable Population Size:**

Unknown

**Resistance to Disease:**

Low

**Adaptability:**

Low

**Additional Population-level Information:**

Black-footed ferret populations are difficult to enumerate due to their remote locations, difficult accessibility, nocturnal habits, and logistical problems and costs associated with the requisite field work. Accordingly, ferret populations at some reintroduction sites are not regularly or even accurately assessed. The U.S. Fish and Wildlife Service (USFWS) views ferret population estimates at most sites as minimum numbers because of the aforementioned issues, and because additional variables such as weather, intensity of search effort, and length of search effort may provide different perspectives (USFWS 2013).

**Population Narrative:**

There have been 20 specific black-footed ferret reintroduction projects, beginning in 1991; they have had with varying levels of success. In two cases, two different reintroductions have since merged into one biological population: Badlands National Park in South Dakota, reintroduced in 1994, and Conata Basin in South Dakota, reintroduced in 1996; and Coyote Basin in Utah, reintroduced in 1999, and Wolf Creek in Colorado, reintroduced in 2001. Black-footed ferret populations are difficult to enumerate due to their remote locations, difficult accessibility, nocturnal habits, and logistical problems and costs associated with the requisite field work. Accordingly, ferret populations at some reintroduction sites are not regularly or even accurately assessed. The USFWS views ferret population estimates at most sites as minimum numbers because of the aforementioned issues, and because additional variables such as weather, intensity of search effort, and length of search effort may provide different perspectives (USFWS 2013). Approximate minimum population sizes at known release sites (as of 2011) are as follows: (1) Shirley Basin, Wyoming (203); (2) Badlands National Park, South Dakota (33); (3) UL Bend NWR, Montana (20); (4) Conata Basin, South Dakota (72); (5) Aubrey Valley, Arizona (75); (6) Fort Belknap Indian Reservation, Montana (0); (7) Coyote Basin, Utah (3); (8) Cheyenne River Indian Reservation, South Dakota (25); (9) Wolf Creek, Colorado (No Data); (10) BLM “40 Complex,” Montana (No Data); (11) Janos, Chihuahua, Mexico (No Data); (12) Rosebud Indian Reservation, South Dakota (No Data); (13) Lower Brule Indian Reservation, South Dakota (12); (14) Wind Cave National Park, South Dakota (46); (15) Espee Ranch, Arizona (No Data); (16) Logan County, Kansas (38); (17) Northern Cheyenne Indian Reservation, Montana (No Data); (18) Vermejo Ranch (black-tailed prairie dog habitat), New Mexico (5); (19) Grasslands National Park,

Saskatchewan, Canada (12); and (20) Vermejo Ranch (Gunnison's prairie dog habitat), New Mexico (No Data) (USFWS 2013). Given the difficulty in enumerating ferret populations throughout their range, population level and species level trends are unknown, but the overall population is estimated to be between 250 and 1,000 individuals. Overall, the black-footed ferret has a low resilience to withstand stochastic events, due to the populations' small sizes and vulnerability to health risks; a low representation to adapt to changing environmental conditions across the landscape, due to the limited breadth of genetic diversity within and among populations; a moderate redundancy to withstand catastrophic events, due to the duplication and distribution of the species across 20 populations; a low resistance to disease, due to small population sizes and the limited breadth of genetic diversity; and low adaptability, due to the specialized nature of the ferret's association with prairie dogs and prairie dog habitat, and their vulnerability to disease and climate change. Information on the species' population growth rates and minimum viable population sizes are unknown and under investigation.

### ***Threats and Stressors***

**Stressor:** Recreational purposes

**Exposure:** Indirect, direct.

**Response:** Decrease in prey (prairie dog) availability, decrease in prey (prairie dog) density, decrease in shelter (burrow) availability, decrease in connectivity and dispersal/movement corridors, increased rates of emigration, increased instances of lead exposure to scavengers/predators.

**Consequence:** Reduction in population numbers, decreased reproductive success, reduction in immigration/emigration, increased genetic effects of population bottleneck, higher susceptibility to mortality/extirpation.

**Narrative:** Recreational prairie dog shooting has increased over the past decade at some black-footed ferret reintroduction sites. Depending on its intensity, shooting can negatively impact local prairie dog populations, and the resulting loss in prey base likely affects black-footed ferret reintroduction sites. Recreational shooting of prairie dogs can lead to emigration, and also causes direct mortality to prairie dog associated species. Thus, incidental take of black-footed ferrets by prairie dog shooters is also a potential, but as yet undocumented, source of ferret mortality. Recreational shooting of prairie dogs also contributes to the problem of lead accumulation in wildlife food chains that include prairie dogs. Killing large numbers of animals, not removing carcasses from the field, and using expanding bullets containing lead may present potentially dangerous amounts of lead to scavengers and predators of prairie dogs (USFWS 2013).

**Stressor:** Disease

**Exposure:** Direct

**Response:** Removal of prime (young/healthy) breeding individuals, decrease in number of breeding individuals.

**Consequence:** Reduction in population numbers, decreased reproductive success, reduction in immigration/emigration, increased genetic effects of population bottleneck, higher susceptibility to mortality/extirpation.

**Narrative:** Given the black-footed ferret's small population sizes, several diseases threaten the species' survival. These includes Sylvatic plague, which can be transmitted by fleas that have acquired the bacterium by biting infected animals; pneumonically (via the respiratory system) among infected animals; or by the consumption of contaminated tissues. Recovery efforts for the black-footed ferret are hampered because both ferrets and prairie dogs are extremely

susceptible to plague. Plague can impact ferrets directly via infection and subsequent mortality. It can also indirectly impact ferrets through the disease's effects on prairie dogs and the potential for dramatic declines in the ferret's primary prey base (USFWS 2013).

**Stressor:** Inadequacies of existing regulatory mechanisms

**Exposure:** Indirect

**Response:** Fewer protective regulations.

**Consequence:** Impairment of ferret recovery efforts.

**Narrative:** Few protective regulations are in place for prairie dogs (on which the black-footed ferret depends for food and shelter) in comparison to the ferret. Although inadequate regulatory mechanisms are not likely to cause black-tailed prairie dog, white-tailed prairie dog, and Gunnison's prairie dog to become threatened or endangered in the foreseeable future, prairie dogs appear able to persist in smaller, more fragmented populations than were common historically; and most prairie dog populations are no longer large and stable enough to support recovery of the ferret. The existing regulatory mechanisms are inadequate to support the large prairie dog populations that ferrets require. More protective regulations for prairie dogs, particularly those related to poisoning and maintenance of large prairie dog complexes, could improve opportunities for ferret recovery at sites that currently have marginal potential. Ferret recovery is biologically possible; however, the restoration of adequate prairie dog habitats will take more time, patience, and commitment by federal, state, local, tribal, and private land managers than has occurred to date (USFWS 2013).

**Stressor:** Poisoning

**Exposure:** Indirect, direct.

**Response:** Decrease in prey (prairie dog) availability, decrease in prey (prairie dog) density, decrease in shelter (burrow) availability, decrease in connectivity and dispersal/movement corridors, increased rates of emigration, increased instances of secondary poisoning to nontarget wildlife such as black-footed ferret.

**Consequence:** Reduction in population numbers, decreased reproductive success, reduction in immigration/emigration, increased genetic effects of population bottleneck, higher susceptibility to mortality/extirpation.

**Narrative:** Poisoning of prairie dogs is a major factor in the historical declines of prairie dogs and black-footed ferrets. Poisoning can affect the ferret directly, through inadvertent secondary poisoning when the ferret consumes poisoned prairie dogs; or indirectly, through the loss of the prairie dog prey base. Primary poisons used for prairie dogs include strychnine, Compound 1080, Zinc phosphide, and the anticoagulant rodenticides chlorophacinone (Rozol®) and diphacinone (Kaput®). Poisoning, if thorough enough, may result in permanent loss of prairie dogs, which can preclude ferret recovery opportunities (USFWS 2013).

## ***Recovery***

### **Reclassification Criteria:**

To reclassify the black-footed ferret from endangered to threatened status, the following criteria, originally established in the 1988 Recovery Plan and expanded (as noted in italics) must be met:

Conserve and manage a captive breeding population of black-footed ferrets with a minimum of 280 adults (105 males, 175 females) distributed among at least three facilities.

Establish free-ranging black-footed ferrets totaling at least 1,500 breeding adults, in 10 or more populations, in at least 6 of 12 states within the historical range of the species, with no fewer than 30 breeding adults in any population.

Maintain these population objectives for at least 3 years prior to downlisting.

Maintain approximately 100,000 ha (247,000 ac.) of prairie dog occupied habitat at reintroduction sites by planning and implementing actions to manage plague and conserve prairie dog populations.

**Delisting Criteria:**

Delisting may occur when the following recovery criteria are met:

Conserve and manage a captive breeding population of black-footed ferrets with a minimum of 280 adults (105 males, 175 females) distributed among at least three facilities.

Establish free-ranging black-footed ferrets totaling at least 3,000 breeding adults, in 30 or more populations, with at least one population in each of at least 9 of 12 states within the historical range of the species, with no fewer than 30 breeding adults in any population, and at least 10 populations with 100 or more breeding adults, and at least five populations within colonies of Gunnison's and white-tailed prairie dogs.

Maintain these population objectives for at least 3 years prior to delisting.

Maintain a total of approximately 200,000 ha (494,000 ac.) of prairie dog occupied habitat at reintroduction sites by planning and implementing actions to manage plague and conserve prairie dogs.

Complete and implement a post-delisting monitoring and management plan, in cooperation with the states and tribes, to ensure that recovery goals are maintained.

**Recovery Actions:**

- Conserve and manage a captive ferret population of sufficient size and structure to support genetic management and reintroduction efforts (USFWS 2013).
- Identify prairie dog habitats with the highest biological potential for supporting future free-ranging populations of ferrets (USFWS 2013).
- Establish free-ranging populations of ferrets to meet downlisting and delisting goals (USFWS 2013).
- Ensure sufficient prairie dog habitat to support a wide distribution of ferret populations over the long term, considering social, political, and economic concerns of local residents (USFWS 2013).
- Reduce disease-related threats in wild populations of ferrets and associated species (USFWS 2013).
- Support partner involvement and conduct adaptive management through cooperative interchange (USFWS 2013).

***Conservation Measures and Best Management Practices:***

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***Additional Threshold Information:***

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## SPECIES ACCOUNT: *Myotis grisescens* (Gray bat)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; 04/28/1976; Great Lakes-Big Rivers Region (R3) (USFWS, 2016)

### **Physical Description**

A small bat with unicolored dorsal fur (gray after the mid-summer molt, at other times sometimes chestnut brown or russet); paler below, with hairs darker basally; wing membrane (gray) connects to the foot at the ankle; calcar is unkeeled; total length is 80 - 105 mm; forearm length is 40 - 46 mm; ear length is 14 - 16 mm; tail length is 33 - 45 mm; hind foot is 9 - 12 mm; mass is 7 - 16 g (usually 8 - 10 g); wingspread is 275 - 300. Distinct sagittal crest on skull. Teeth 38; dentition I 2/3, C 1/1, P 3/3, M 3/3 (Barbour and Davis 1969, Sealander 1979) (NatureServe, 2015).

### **Taxonomy**

The gray bat is one of the largest species in the genus *Myotis* in eastern North America (Decher and Choate 1995) (USFWS, 2009).

### **Historical Range**

See current range/distribution.

### **Current Range**

The range extends from southeastern Kansas and central Oklahoma east to western Virginia and western North Carolina, and from Missouri, Illinois, and Indiana south to southern Alabama and northern Florida (Decher and Choate 1995). The primary range is concentrated in the cave regions of Alabama, Arkansas, Kentucky, Missouri and Tennessee, with smaller populations found in adjacent states, including a growing population in a quarry in Clark County, Indiana (USFWS 2009). Summer and winter ranges are essentially the same (NatureServe, 2015).

### **Distinct Population Segments Defined**

No

### **Critical Habitat Designated**

No;

### ***Life History***

### **Feeding Narrative**

Adult: Adults and immatures are invertivores. Feeds mostly upon flying insects, including mayflies and beetles (Tuttle et al., Lacki et al. 1995); diet may vary with local resources and habitat. Foraging is generally parallel to streams, over the water at heights of 2 to 3 m (LaVal et al. 1977) or in forests (Caire et al. 1989). The energy demands on adult females are tremendous during lactation, and individual females sometimes feed continuously for seven or more hours per night (Tuttle and Stevenson). Individuals enter hibernation from October to November and emerge from hibernation between March and May (Tuttle 1976). Forages at night in loose groups, but become territorial when insect numbers decrease; territories seem to be controlled by reproductively-active females (Tuttle et al.) (NatureServe, 2015). Foraging of gray bats in

summers is strongly correlated with open water of rivers, streams, lakes or reservoirs. Gray bats are highly dependent on aquatic insects, especially mayflies, caddisflies, and stoneflies. The species is an opportunistic forager, however, and also consumes beetles and moths (Harvey 1994; Tuttle and Kennedy 2005) (USFWS, 2009).

### **Reproduction Narrative**

Adult: Mating occurs in September - October. Adult females store sperm through the winter and become pregnant soon after emergence from hibernation (Gutherie and Jeffers 1938, Harvey 1994, Tuttle and Kennedy 2005). One young is born late in May or in early June. Most young are able to fly in 20 - 35 days, depending on colony size. Individual females typically do not produce young until their second year. Recorded maximum longevity approximately 14 - 17 years but may be longer (Harvey 1992, Tuttle and Kennedy 2005). Maternity caves often have a stream flowing through them and are separate from the caves used in summer by males. Although mothers continue to nurse young for a period after the young are flying, juveniles are apparently left to learn how to hunt on their own (Tuttle and Stevenson). Elder and Gunier (1981) determined that the mean annual survival rate is about 70% in males and 73% in females. Stevenson and Tuttle (1981) found that the after-first-year survival rate is about 55 to 85% in relatively undisturbed colonies, and 57 to 66% in disturbed colonies (NatureServe, 2015). Average gestation is approximately 64 days and a single pup is born in late May or early June. Nursery colonies typically form on domed ceilings that are capable of trapping the combined body heat from clustered individuals and where the temperature ranges between 14° and 25°C (Harvey 1992; Harvey 1994; Tuttle and Kennedy 2005; Martin 2007) (USFWS, 2009).

### **Spatial Arrangements of the Population**

Adult: Colonial (NatureServe, 2015)

### **Environmental Specificity**

Adult: Narrow (inferred from USFWS, 2009)

### **Site Fidelity**

Adult: High (NatureServe, 2015)

### **Habitat Narrative**

Adult: Roost sites are nearly exclusively restricted to caves throughout the year (Hall and Wilson 1966, Barbour and Davis 1969, Tuttle 1976). Winter roosts are in deep vertical caves with domed halls. Large summer colonies utilize caves that trap warm air and provide restricted rooms or domed ceilings. Occasionally non-cave roost sites are used, such as storm sewers and buildings. Hibernation sites often have multiple entrances, good air flow (Martin 2007), and temperatures of approximately 5 - 9°C, though 1 - 4°C may be preferred (Tuttle and Kennedy 2005). Undisturbed summer colonies may contain up to 250,000 bats, and average 10,000 to 25,000 (Tuttle 1979). Summer caves are nearly always located within 1 km of a river or reservoir over which the bats forage (Tuttle 1979). Tuttle (1979) showed that forested areas along the banks of streams and lakes provide important protection for adults and young. Gray bats show strong philopatry to both summering and wintering sites (Tuttle 1976; Tuttle 1979; Kennedy and Tuttle 2005; Martin 2007) (NatureServe, 2015). Because of their highly specific roost and habitat requirements, only about 5% of available caves are suitable for occupancy by gray bats (Tuttle 1979; Harvey 1994) (USFWS, 2009).

***Dispersal/Migration*****Motility/Mobility**

Adult: High (inferred from NatureServe, 2015)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Migrates between summer and winter roosting sites (NatureServe, 2015)

**Dispersal**

Adult: High (inferred from NatureServe, 2015)

**Dispersal/Migration Narrative**

Adult: Wintering caves often are hundreds of kilometers from summer range. Individuals regularly migrate 17 - 437 kilometers between summer maternity sites and winter hibernacula, with some individuals moving as much as 689 - 775 kilometers (Hall and Wilson 1966, Tuttle 1976; Tuttle and Kennedy 2005). Although individuals may travel up to 35 kilometers between prime feeding areas over lakes or rivers and occupied caves (LaVal et al. 1977, Tuttle and Stevenson 1977, Tuttle and Kennedy 2005), most maternity colonies are 1 - 4 kilometers from foraging locations (Tuttle 1976) (NatureServe, 2015).

***Population Information and Trends*****Population Trends:**

Decline of 10-50% (NatureServe, 2015)

**Species Trends:**

Increasing (NatureServe, 2015)

**Resiliency:**

High (inferred from NatureServe, 2015)

**Redundancy:**

High (inferred from NatureServe, 2015)

**Number of Populations:**

Unknown (NatureServe, 2015)

**Population Size:**

3.4 million (NatureServe, 2015)

**Population Narrative:**

The area of occupancy, number of subpopulations/locations, and abundance likely have decreased over the past 200 years, but the degree of decline is uncertain, probably 10 - 50%. Area of occupancy, number of subpopulations/locations, and abundance have increased over the past several decades. According to USFWS (2009), Michael Harvey reported that the species increased from approximately 1,575,000 to roughly 2,678,000 in 2002 and to around 3,400,000 in 2004 (Ellison et al. 2003; Martin 2007). Total population was estimated at 1.5 million in the early 1980s, approximately 3.4 million in 2005-2007 (Harvey and Currie 2007, Martin 2007). The

number of distinct occurrences (subpopulations) has not been determined using standardized criteria. Martin (2007) listed the species as occurring in 384 caves scattered across 11 states (his analysis did not include Indiana) (NatureServe, 2015).

### ***Threats and Stressors***

**Stressor:** Human disturbance (NatureServe, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Decline began with cave disturbance associated with saltpeter production during the Civil War. Some of the largest colonies were lost as a result of cave commercialization. Some caves were improperly gated. Cave disturbance was the major factor in the historical decline. Cave protection efforts have greatly reduced this threat. However, human disturbance is the main reason for the continued decline of gray bats in caves that are not protected (USFWS 2009). The species is especially vulnerable due to its high fidelity to particular favored caves, and it is very sensitive to disturbance, including the mere presence of humans with lights; disturbance may result in bats moving to less favorable roosting places (NatureServe, 2015).

**Stressor:** White nose syndrome (USFWS, 2009)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** In February 2006, an unknown fungus was documented on a number of hibernating bats at Howes Cave near Albany, New York (Blehert et al. 2009). The unknown growth was labeled as white-nose syndrome due to the presence of a visually striking whitish covering on the muzzles, ears, or wing membranes of affected bats. Although WNS has not yet been documented in any population of *Myotis grisescens*, the recent discovery of the condition in Smyth County (Hancock Cave), Virginia is approximately 11 miles from a bachelor colony of ca. 2000 gray bats in the same county and ca. 22 miles from a bachelor colony of 2000 gray bats in Russell County (Ferrell's Cave) (David Kampwerth, USFWS, Conway, AR FO, pers. comm. 13 Aug 2009). Many bat experts predict that WNS will continue to spread south and west (Zimmerman 2009). Al Hicks, a wildlife biologist for the New York Department of Environmental Conservation and who has been a leader in learning more about WNS, stated that "all of our hibernating bats are in trouble" (in Cohn 2008). This would obviously include gray bats, and given that WNS is apparently not species specific and has already been confirmed adversely impacting three other species in the genus *Myotis*, white-nose syndrome should be viewed as a new threat to *M. grisescens* (USFWS, 2009).

**Stressor:** Habitat degradation (NatureServe, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The use of forestry insecticides and crop pesticides in areas adjacent to riparian corridors where gray bats forage may reduce the prey base or kill bats that ingest contaminated insects (Northern Prairie Wildlife Research Center). Pesticide contamination remains a concern but currently is not known to be causing declines (USFWS 2009). Other threats include deforestation and impoundment of waterways (and subsequent cave inundation). Natural and

human-caused flooding remains a secondary threat at some gray bat sites (USFWS 2009) (NatureServe, 2015).

**Stressor:** Climate change (NatureServe, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Climate change could have a significant impact on gray bats. It is projected that a rise in ambient temperature could make traditional and currently occupied hibernacula and maternity sites unsuitable for roosting gray bats and cause a shift in the species' range northward. This could adversely affect the species' food supply, or affect the ability of bats to adequately deposit important fat reserves that are critical for winter survival. However, no documentation of such effects currently exists (USFWS 2009; NatureServe, 2015)

### ***Recovery***

#### **Reclassification Criteria:**

1. Documentation of permanent protection of 90% of Priority 1 hibernacula (USFWS, 2009).
2. Documentation of stable or increasing populations at 75% of Priority 1 maternity caves for 5 years (USFWS, 2009).

#### **Delisting Criteria:**

1. Documentation of permanent protection of 25% of Priority 2 caves in each state and reclassification criteria have been met (USFWS, 2009).
2. Documentation of stable or increasing populations of 25% of Priority 2 caves in each state for 5 years and reclassification criteria have been met (USFWS, 2009).

#### **Recovery Actions:**

- Prevent disturbance to important roost habitat (USFWS, 1982).
- Maintain, protect, and restore foraging habitat (USFWS, 1982).
- Monitor population trends (USFWS, 1982).

### ***Conservation Measures and Best Management Practices:***

- During the next review period, the following priority actions should be undertaken: 1) continued monitoring of the spread of WNS, 2) continued efforts to prevent human disturbance to roosting gray bats by placement of various protective measures at maternity sites, 3) reestablishment of natural air flow at sites where improperly installed cave gates or other structures have impeded air circulation and adversely impacted sensitive roosting temperatures, 4) ongoing monitoring of gray bat populations at Priority 1 hibernacula, and selected maternity sites (see further comments below), and 5) at sites where cave protection is not possible through the use of gates, fences, or signs, continue efforts to prevent human intrusion through the use of conservation easements, Safe Harbor agreements, private landowner agreements, or other mechanisms (USFWS, 2009).
- The U.S. Fish and Wildlife Service (1982) listed "stable or increasing populations for 5 years" at Priority 1 and 2 maternity sites as part of its recommended reclassification and delisting criteria for gray bat. As pointed out, however, by Tuttle (1979, 2003), Sabol and Hudson (1995), and Elliott (2008), there are numerous problems associated with assessing population trends of roosting bats,

especially at some maternity sites where population estimates fluctuate widely from year to year due to movement of bats among sites (Traci Hemberger, Kentucky Dept. of Fish and Wildlife Resources, pers. comm. 22 May 2006, 25 January 2009). Sasse et al. (2007) questioned the usefulness of including five-year population trend analyses at maternity sites as reclassification and delisting criteria for gray bat because they believed that five years was insufficient time to examine trends and the large number of sites precluded an adequate evaluation. To assess gray bat population trends, we recommend: 1) the continued censusing of Priority 1 hibernacula and 2) determine the effectiveness of obtaining annual estimates of select gray bat maternity sites by using technologically advanced equipment such as NIR or TIR videography with computer and statistical software packages as recommended by Sabol and Hudson (1995), Kunz (2003), Martin (2007), Sasse et al. (2007) and Elliott (2008). The maternity sites selected for annual monitoring should be established through cooperation between the USFWS and its many partners. As pointed out by Sasse et al. (2007) and others (e.g., Bill Gates, USFWS - Wheeler National Wildlife Refuge, pers. comm. 31 Aug 2009; peer reviewers - see appendices B&C below), monitoring should be greater than five years because a longer time period will be necessary to adequately assess population trends of this species. The Service recommends that the species be monitored for a minimum of 10-20 years (USFWS, 2009).

- As noted in sections 2.3.2.3 and 2.4, the spread of WNS to gray bat populations would be catastrophic and significantly increase the threat of extinction. To address the possible spread of the fungus to gray bat populations, a WNS response and action plan should be developed by USFWS in close coordination and cooperation with our many Federal, state, and private partners. This plan should include: 1) protocols for documenting the presence of the fungus, 2) suggested methods to hopefully curtail the spread of WNS, 3) protocols for dealing with contaminated bats, 4) protocols for maintaining and protecting gray bat populations (e.g., prioritizing cave closures, the possible quarantine of affected bats, any treatment measures if available), and 5) guidelines for public outreach, and other issues as identified. The spread of WNS into gray bat populations would necessitate a revision to the species' recovery plan (USFWS, 2009).

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## SPECIES ACCOUNT: *Myotis septentrionalis* (Northern long-eared bat)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Threatened; 05/04/2015; Great Lakes-Big Rivers Region (R3) (USFWS, 2016)

#### **Physical Description**

The northern long-eared bat is a medium-sized bat about 3 to 3.7 inches in length but with a wingspan of 9 to 10 inches. As its name suggests, this bat is distinguished by its long ears, particularly as compared to other bats in its genus, *Myotis*, which are actually bats noted for their small ears (*Myotis* means mouse-eared) (USFWS, 2016). Pelage (fur) colors include medium to dark brown on its back; dark brown, but not black, ears and wing membranes; and tawny to pale-brown fur on the ventral side (Nagorsen and Brigham 1993, p. 87; Whitaker and Mumford 2009, p. 207) (USFWS, 2015).

#### **Taxonomy**

*Myotis septentrionalis* formerly was regarded as conspecific with *Myotis keenii*; van Zyll de Jong (1979, 1985) and Jones et al. (1992) regarded *Myotis keenii* and *M. septentrionalis* as separate species; Koopman (in Wilson and Reeder 1993) included *septentrionalis* in *Myotis keenii*, noting that they may be separate species. Baker et al. (2003) and Simmons (in Wilson and Reeder 2005) recognized *M. septentrionalis* and *M. keenii* as distinct species. Most older literature using the name *Myotis keenii* actually pertains to *Myotis septentrionalis*. No subspecies are recognized. No genetically distinctive subpopulations have been identified (Johnson et al. 2014) (NatureServe, 2015).

#### **Historical Range**

Prior to the incidence of white-nose syndrome, this species was regarded as more common in the northern part of the range than in the south (Harvey 1992), and it was rare in the northwestern portion of the range (Nagorsen and Brigham 1993, Caceres and Barclay 2000). It was reported as very rare in Alabama (Best, pers. comm.), uncommon in Indiana, Kentucky, Tennessee, and Wisconsin (Mumford and Cope 1964, Harvey 1991, Jackson 1961), more common in northern Michigan than in southern Michigan (Kurta 1982), and quite common in New York (Hamilton and Whitaker 1979). (NatureServe, 2015). Historically, the northern long-eared bat was widely distributed in the eastern part of its range (Caceres and Barclay 2000, p. 2) (USFWS, 2015).

#### **Current Range**

This bat is widely but patchily distributed in the eastern and northcentral United States and adjacent southern Canada, from eastern British Columbia and southern Yukon eastward across southern Canada to eastern Quebec, Prince Edward Island, and Newfoundland, and southward to southern Texas (one old record), Louisiana, Alabama, Georgia, and Florida (one old record from panhandle), and westward in the United States generally to the eastern margin of the Great Plains region (Barbour and Davis 1969, Harvey 1992, van Zyll de Jong 1985, Hall 1981, Crnkovic 2003, Wilson and Reeder 2005, Amelon and Burhans 2006, Marks and Marks 2006, Henderson et al. 2009, Ammerman et al. 2012, Park and Broders 2012). The overall summer and winter ranges are essentially the same (Barbour and Davis 1969) (NatureServe, 2015).

#### **Distinct Population Segments Defined**

No

**Critical Habitat Designated**

Yes;

***Life History*****Feeding Narrative**

Adult: This species evidently is an opportunistic insectivore (Kunz 1973); prey composition varies widely among sites and seasons; diet includes Lepidoptera, Coleoptera, Neuroptera, Diptera, Hymenoptera, Homoptera, and Hemiptera (Whitaker 1972, LaVal and LaVal 1980, Griffith and Gates 1985, Dodd et al. 2012; see also Ammerman et al. 2012r for a review of other recent information). These bats capture flying insects and also glean prey from plants or the forest floor. Foraging occurs within forests, along forest edges, over forest clearings, and occasionally over ponds (Ammerman et al. 2012). Hibernation occurs from late summer/early fall to spring. In summer, an activity peak generally occurs 1 - 2 hours after sunset, with a secondary peak 7 - 8 hours after sunset (NatureServe, 2015). Arachnids are also being a common prey item (Feldhamer et al. 2009, p. 45) (USFWS, 2015).

**Reproduction Narrative**

Adult: Copulation occurs in the late summer and early fall, during the swarming period when large numbers of bats congregate in and near certain caves (Baker 1983, Kurta 1980). Females store sperm during hibernation, though some may copulate again at spring emergence (Guthrie 1933, Racey 1982). Females ovulate at the time of emergence and parturition occurs 50 - 60 days later (Baker 1983). Females bear a single young. Young-of-the-year may reproduce in their first fall, but the proportion of the cohort doing so is unknown (Kurta, pers. comm.). Most nursery colonies are in cavities or beneath loose bark in trees or snags in upland forests, with roost entrances generally below or within the tree canopy (Mumford and Cope 1964, Sasse and Perkins 1996, Lacki and Schwierjohann 2001, Menzel et al. 2002, Owen et al. 2002, Carter and Feldhamer 2005, Perry and Thill 2007, Lacki et al. 2009, Timpone et al. 2010, Silvis et al. 2012). The disparity in the sex ratio (male-biased) appears to be quite consistent among studies, seasons, and sites. Although age structure is not known for any population, potential longevity is at least two decades (NatureServe, 2015).

**Spatial Arrangements of the Population**

Adult: Solitary (NatureServe, 2015); colonial, small groups (USFWS, 2015)

**Site Fidelity**

Adult: High (NatureServe, 2015)

**Habitat Narrative**

Adult: Individuals usually roost solitarily. This bat generally is associated with old-growth forests composed of trees 100 years old or older. It relies on intact interior forest habitat, with low edge-to-interior ratios. Relevant late-successional forest features include a high percentage of old trees, uneven forest structure (resulting in multilayered vertical structure), single and multiple tree-fall gaps, standing snags, and woody debris. However, recent studies indicate that these bats can exploit relatively isolated and small forest fragments (Caceres and Barclay 2000, Henderson et al. 2008, Johnson et al. 2008). Hibernation occurs primarily in caves, mines, and

tunnels, typically those with large passages and entrances, relatively constant and cool temperatures, high humidity, and no air currents (Griffin 1940, Jackson 1961, Mumford and Cope 1964, Kurta 1982, Raesly and Gates 1987, Caceres and Pybus 1997, USFWS 2013). A lack of suitable hibernacula may prevent occupancy of areas that otherwise have adequate habitat (Kurta 1982). There appears to be a high degree of philopatry in hibernaculum use. Caves, mines, and quarry tunnels are used as night roosts, typically by males, but also by non-reproductive females (Clark et al. 1987, Jones et al. 1967) (NatureServe, 2015). Northern long-eared bats actively form colonies in the summer (Foster and Kurta 1999, p. 667) and exhibit fission-fusion behavior (Garroway and Broders 2007, p. 961), where members frequently coalesce to form a group (fusion), but composition of the group is in flux, with individuals frequently departing to be solitary or to form smaller groups (fission) before returning to the main unit (Barclay and Kurta 2007, p. 44) (USFWS, 2015).

***Dispersal/Migration*****Motility/Mobility**

Adult: High (NatureServe, 2015)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Seasonal movements between winter and summer sites (USFWS, 2015)

**Dispersal**

Adult: Moderate (USFWS, 2015)

**Dispersal/Migration Narrative**

Adult: In West Virginia, foraging home ranges of seven females averaged 61.1 hectares (Menzel et al. 1999) (NatureServe, 2015). While the northern long-eared bat is not considered a long-distance migratory species, short regional migratory movements between seasonal habitats (summer roosts and winter hibernacula) have been documented between 56 km (35 mi) and 89 km (55 mi) (Nagorsen and Brigham 1993 p. 88; Griffin 1940b, pp. 235, 236; Caire et al. 1979, p. 404) (USFWS, 2015).

***Population Information and Trends*****Population Trends:**

Decline of >70% (NatureServe, 2015)

**Species Trends:**

>70% decline (NatureServe, 2015)

**Resiliency:**

Very high (inferred from USFWS, 2016)

**Representation:**

High (inferred from NatureServe, 2015)

**Redundancy:**

High (inferred from NatureServe, 2015)

**Number of Populations:**

Unknown (NatureServe, 2015)

**Population Size:**

10,000 - 100,000 individuals (NatureServe, 2015)

**Resistance to Disease:**

Low (inferred from NatureServe, 2015; see threats)

**Population Narrative:**

Recent genetic data indicate that movements and genetic interchanges among populations may be considerable. The range-wide trend over the long term is uncertain, but the number of subpopulations as well as the overall population size clearly have declined (>70%). Range-wide trend over the past 10 years or three generations is uncertain, but the number of subpopulations as well as the overall population size clearly have declined to a large degree (>70%). Total adult population size is unknown but presumably at least 10,000 and perhaps greater than 100,000. Although there are hundreds of hibernating colonies rangewide, these colonies rarely comprise even as many as 50 individuals (very exceptionally 300), suggesting that the overall population (even before the incidence of white-nose syndrome) was relatively small. The number of distinct occurrences has not been determined using standardized criteria. More than 780 hibernacula have been identified throughout the species' range in the United States, although many hibernacula contain only a few (1 to 3) individuals (Barbour and Davis 1969, Whitaker and Hamilton 1998, USFWS 2013). Missouri, Pennsylvania, and West Virginia each have greater than 100 known hibernacula (USFWS 2013) (NatureServe, 2015). The species' range includes 37 states (USFWS, 2016).

***Threats and Stressors***

**Stressor:** White nose syndrome (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The most serious threat is white-nose syndrome (WNS), an often (but not always) lethal condition caused by a fungal pathogen (*Pseudogymnoascus destructans*). WNS was first noticed in 2006 in New York. Since its initial discovery, WNS has spread rapidly (confirmed in more than 100 bat hibernacula) and now has been documented throughout northeastern North America and as far west as Missouri and Arkansas, and south to northern Alabama and northern Georgia (as of May 2014; [www.whitenosesyndrome.org](http://www.whitenosesyndrome.org)). WNS affects *Myotis septentrionalis* and several other bat species (Gargas et al. 2009) and has resulted in several million bat deaths in the northeastern United States in recent years. Though *M. septentrionalis* was not common in surveys in the northeastern United States before the recognition of WNS, counts of this species subsequently have declined to zero in many caves since the advent of the disease (Hicks et al. 2008). As of 2013, WNS was still spreading and was documented in 22 of the 39 states in which the species occurs. As of early 2015, WNS was still spreading but was confined primarily to areas east of the Mississippi River (plus several locations in Arkansas and Missouri, with suspected instances in Iowa and Minnesota). The vast majority of known hibernacula are in regions where WNS has been confirmed. USFWS (2013) found no information to indicate that there are areas

within the species' range that will not be impacted by the disease or that similar rates of decline (to what has been observed in the East, where the disease had been present for at most 8 years) will not occur throughout the species' range (NatureServe, 2015).

**Stressor:** Habitat modification and degradation (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Loss, degradation, and fragmentation of mature forest habitat (associated with various kinds of human activities, such as logging; oil, gas, and mineral development; and wind energy development) also may be a significant threat (Center for Biological Diversity 2010, USFWS 2011). However, the general lack of genetic structure at both watershed and regional scales indicates that forest disturbances such as prescribed fire or timber harvest at watershed scales do not appear to disrupt northern myotis gene flow across the landscape (Johnson et al. 2014). Mortality caused directly by wind turbines may pose a significant threat in some areas (USFWS 2011). Closures of mines used for hibernation are a potential threat, but there is no evidence that mine closures are currently affecting *Myotis septentrionalis* populations (USFWS 2011; NatureServe, 2015). Anthropogenic modifications to cave and mine entrances, such as the addition of restrictive gates or other structures intended to exclude humans, may not only alter flight characteristics and access (Spanjer and Fenton 2005, p. 1110), but may change airflow and alter internal microclimates of the caves and mines, eliminating their utility as hibernacula (Service 2007, p. 71). Northern long-eared bats likely evolved with fire in their habitat, and thus may benefit from fire-created habitat. However, there are potential negative effects from prescribed burning, including direct mortality (USFWS, 2015).

**Stressor:** Human disturbance (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** This species is sensitive to disturbance during hibernation (Garner, pers. comm., Thomas 1995); frequently aroused bats may deplete their energy reserves. Nursery colonies are very sensitive to disturbance by humans; bats may move to an alternate roost after a single disturbance, even if no attempt is made to capture the bats (Layne 1978) (NatureServe, 2015).

**Stressor:** Climate change (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Climate change may also affect this species, as northern long-eared bats are particularly sensitive to changes in temperature, humidity, and precipitation. Impacts from climate change may also indirectly affect the northern long-eared bat due to changes in food availability, timing of hibernation, and reproductive cycles, along with other factors, all of which may contribute to a shift in suitable habitat (USFWS, 2015).

**Stressor:** Contaminants (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Environmental contaminants, in particular insecticides, pesticides, and inorganic contaminants, such as mercury and lead, may also have detrimental effects on northern long-eared bats. Contaminants may bioaccumulate (become concentrated) in the tissues of bats, potentially leading to a myriad of sublethal and lethal effects (USFWS, 2015).

### **Recovery**

**Reclassification Criteria:**

Not available - this species does not have a recovery plan.

**Delisting Criteria:**

Not available - this species does not have a recovery plan.

**Recovery Actions:**

- Not available - this species does not have a recovery plan.

### **Conservation Measures and Best Management Practices:**

- Direct protection of caves and mines can be accomplished through installation of bat-friendly gates that allow passage of bats while reducing disturbance from human entry as well as changes to the cave microclimate from air restrictions. The NPS has proactively taken steps to minimize effects to underground bat habitat resulting from vandalism, recreational activities, and abandoned mine closures (Plumb and Budde 2011, unpublished data). In addition, the NPS is properly gating abandoned coal mine entrances, using a “bat-friendly” design, as funding permits (Graham 2011, unpublished data). All known hibernacula within national grasslands and forestlands of the Rocky Mountain Region of the USFS are closed during the winter hibernation period, primarily due to the threat of WNS, although this will reduce disturbance to bats in general inhabiting these hibernacula (USFS 2013, unpaginated). Many States are also taking a proactive stance to conserve and restore forest and riparian habitats with specific focus on maintaining forest patches and connectivity. Many States are undertaking research and monitoring efforts to gain more information about habitat needs of and use by northern long-eared bat (USFWS, 2015).
- In 2011, the Service, in partnership with several other State, Federal, and Tribal agencies, finalized a national response plan for WNS to provide a common framework for the investigation and management of WNS (Service 2011, p. 1). In 2012, a sister plan was finalized for the national response to WNS in Canada, allowing for a broader coordinated response to the disease throughout the two countries. The multi-agency, multiorganization WNS response team, under the U.S. National Plan and in coordination with Canadian partners, has and continues to develop recommendations, tools, and strategies to slow the spread of WNS, minimize disturbance to hibernating bats, and improve conservation strategies for affected bat species. In 2009, the Service also issued a recommendation for a voluntary moratorium on all caving activity in States known to have hibernacula affected by WNS, and all adjoining States, unless conducted as part of an agency-sanctioned research or monitoring project (Service 2009, entire). The NPS is currently updating their cave management plans (for parks with caves) to include actions to minimize the risk of WNS spreading to uninfected caves. Research is also under way to develop control and treatment options for WNS-infected bats and environments (USFWS, 2015).
- Due to the known impacts from wind energy development, in particular to listed (and species currently being evaluated to determine if listing is warranted) bird and bat species in the Midwest, the Service, State natural resource agencies, and wind energy industry representatives are developing the MSHCP. The planning area includes the Midwest Region of the Service, which

includes all of the following States: Illinois, Indiana, Iowa, Michigan, Minnesota, Missouri, Ohio, and Wisconsin. The MSHCP would allow permit holders to proceed with wind energy development, which may result in “incidental” taking of a listed species under section 10 of the Act, through issuance of an incidental take permit (77 FR 52754; August 30, 2012). Currently, the northern long-eared bat is included as a covered species under the MSHCP. The MSHCP will address protection of covered species through avoidance, minimization of take, and mitigation to offset “take” (e.g., habitat preservation, habitat restoration, habitat enhancement) to help ameliorate the effect of wind development (77 FR 52754; August 30, 2012). In some cases, the USFS has agreed to limit or restrict burning in the central hardwoods from mid- to late April through summer to avoid periods when bats are active in forests (Dickinson et al. 2010, p. 2200) (USFWS, 2015).

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## SPECIES ACCOUNT: *Myotis sodalis* (Indiana bat)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; 03/11/1967; Great Lakes-Big Rivers Region (R3) (USFWS, 2015)

### **Physical Description**

A medium-sized bat. Its forearm length is 35-41 mm (13 /8- 15 /8 in), and the head and body length ranges from 41-49 mm (15 /8-17 /8 in). The Indiana bat usually has a distinctly keeled calcar. The ears and wing membranes have a dull appearance and flat coloration that does not contrast with the fur, and the fur lacks luster compared with that of little brown bats (Barbour and Davis 1969, Hall 1981). The nose of an Indiana bat is lighter in color than that of a little brown bat. The skull of an Indiana bat has a small sagittal crest, and the braincase tends to be smaller, lower, and narrower than that of the little brown bat (Barbour and Davis 1969, Hall 1981) (USFWS, 2007).

### **Taxonomy**

The Indiana bat was first described as a species by Miller and Allen (1928), based on museum specimens collected in 1904 from Wyandotte Cave in Crawford County, Indiana. Before that time, specimens of the Indiana bat often were confused with those of other *Myotis*, especially the little brown bat. The Indiana bat is monotypic, indicating there are no recognized subspecies. Alternative common names for the species are Indiana myotis, social bat, pink bat, and little sooty bat (Bailey 1933, Osgood 1938, Nason 1948, Mumford and Whitaker 1982) (USFWS, 2007).

### **Historical Range**

Historically, the Indiana bat had a winter range restricted to areas of cavernous limestone in the karst regions of the east-central United States (Miller and Allen 1928, Hall 1962, Thomson 1982, Figure 1). Prior to and during much of the European settlement of the eastern United States, winter populations of Indiana bats likely occurred in karst regions of what would eventually become Alabama, Arkansas, Georgia, Illinois, Indiana, Iowa, Kentucky, Maryland, Massachusetts, Missouri, New Jersey, New York, North Carolina, Oklahoma, Pennsylvania, Tennessee, Vermont, Virginia, and West Virginia. The historic summer distribution and range for this species is poorly documented. The historic summer range almost certainly included areas where the bats have now been locally extirpated due to extensive loss and fragmentation of summer habitat (e.g., forests, woodlands, wetlands) (USFWS, 2007).

### **Current Range**

The overall range extends west to the western Ozark region in eastern Oklahoma (Saugey et al. 1990) and Iowa (Clark et al. 1987), north and east to southern Wisconsin and Michigan (Evers 1992, Kurta and Teramino 1994, Kurta 1995), New York, New England, and northern New Jersey, and south to northern Alabama and Arkansas, with accidental or nonregular occurrences outside this range (e.g., Florida, Marks and Marks 2006). The species has disappeared from or greatly declined in most of its former range in the northeastern United States (e.g., Trombulak et al. 2001). Most capture records of reproductively active females and juveniles have occurred in glaciated portions of the Midwest including southern Iowa, northern Missouri, much of Illinois, most of Indiana, southern Michigan, and western Ohio, and in Kentucky, with a growing number of maternity records documented in New York, New Jersey, and Vermont in recent

years (USFWS 2009). Maternity colonies also exist to the south in Arkansas (Brandebura et al. 2011) and in heavily forested regions to at least eastern Tennessee and western North Carolina (Britzke et al. 2003). However, the geographic locations of the majority of Indiana bat maternity colonies remain unknown (USFWS 2009). Northern populations migrate south to Alabama, Tennessee, Kentucky, Indiana, Missouri, and West Virginia for winter. In winter, the species is apparently absent from Michigan, Ohio, and northern Indiana where suitable caves and mines are unknown. About 42 percent of the total population hibernates in southern Indiana (USFWS 2013). (NatureServe, 2015). In general, the spatial distribution of winter habitat/hibernacula has changed little since the Indiana bat was first listed. However, in at least three known cases, the species has expanded its current winter range beyond its historic winter limits as a result of occupying man-made hibernacula (e.g., mines, tunnels, a dam) in relatively recent times (USFWS, 2009).

**Distinct Population Segments Defined**

No

**Critical Habitat Designated**

Yes; 9/24/1976.

**Legal Description**

On September 24, 1976, the U.S. Fish and Wildlife Service designated critical habitat for the Indiana Bat (*Myotis sodalis*) pursuant to Section 7 of the Endangered- Species Act of 1973 (41 FR 41914 - 41916). This Final Rule was corrected and augmented on September 22, 1977 (42 FR 47840 - 47845).

**Critical Habitat Designation**

Approximately 75 percent of the known population hibernates at the sites designated below. The bats are entirely dependent on the shelter provided by there caves and mines during the winter. Their loss or subjection to excessive disturbance or modification would lead to the near or t3tal extinction of the species.

The Following areas (exclusive of those existing man-made structures or settlements which are not necessary to the normal needs or survival of the species) are critical habitat for the Indiana bat (*Myotis sodalis*):

- (1) Illinois. The Blackball Mine, La Salle County.
- (2) Indiana. Big Wyandotte Cave, Crawford County; Bay's Cave, Greene County.
- (3) Kentucky. Bat Cave, Carter County; Coach Cave, Edmonson County.
- (4) Missouri. Cave 021, Crawford County; Cave 009, Franklin County; Cave 017, Franklin County; Pilot Knob Mine, Iron County; Bat Cave, Shannon County; Cave 029, Washington County (numbers assigned by Division of Ecological Services, U.S. Fish and Wildlife Service, Region 6).
- (5) Tennessee. White Oak Blowhole Cave, Blount County.
- (6) West Virginia. Hellhole Cave, Pendleton County.

**Primary Constituent Elements/Physical or Biological Features**

Not described. From the critical habitat designation, it can be inferred that shelter provided by caves and mines during the winter is a primary constituent element for this species.

***Life History*****Feeding Narrative**

Adult: Flying insects are the typical prey items; diet reflects prey present in available foraging habitat. Foraging habitats include riparian areas, upland forests, ponds, and fields (Menzel et al. 2001), but forested landscapes are the most important habitat in agricultural landscapes (Menzel et al. 2005). Females begin hibernation soon after mating, whereas males often remain active through mid-October to November (Cope and Humphrey 1977). Most individuals are in hibernation by late November although some are still active until December (Barbour and Davis 1969). Activity is resumed generally in April, with few bats still in the hibernation caves by mid-May (NatureServe, 2015). Consistent use of moths, flies, beetles, and caddisflies throughout the year at various colonies suggests that Indiana bats are selective predators to a certain degree, but incorporation of ants into the diet also indicates that these bats can be opportunistic (Murray and Kurta 2002). The Indiana bat is a nocturnal insectivore (USFWS, 2007).

**Reproduction Narrative**

Adult: Mating occurs from late August to early October prior to hibernation, or in spring. Bats assemble at cave entrances at dusk and dawn in late August and September. Such staging is believed to facilitate breeding and reduce the chances of inbreeding in small summer colonies (Humphrey and Cope 1977). Ovulation takes place after the bats arouse in spring. Delayed fertilization (from sperm stored during the autumn matings) occurs in most reproductively active females (Guthrie 1933). Solitary females or small maternity colonies bear their offspring in hollow trees or under loose bark of living or dead trees (Humphrey et al. 1977, Garner and Gardner 1992). Young are born in June - July. Litter size is 1. Young first fly at 25 - 37 days. Maximum longevity is about 15 years (NatureServe, 2015). Female Indiana bats, like most temperate vespertilionids, give birth to one young each year (Mumford and Calvert 1960, Humphrey et al. 1977, Thomson 1982). The age of reproductive maturity is highly variable in vespertilionids, ranging from 3 to 16 months in both sexes (Tuttle and Stevenson 1982). This species has low fecundity. The sex ratio of the Indiana bat is generally reported as equal or nearly equal, based on early work by Hall (1962), Myers (1964), and LaVal and LaVal (1980) (USFWS, 2007).

**Geographic or Habitat Restraints or Barriers**

Adult: Large open areas (USFWS, 2007)

**Spatial Arrangements of the Population**

Adult: Forms hibernation clusters of 500 - 1,000 (NatureServe, 2015); females colonial year round (USFWS, 2007)

**Environmental Specificity**

Adult: Narrow (NatureServe, 2015)

**Tolerance Ranges/Thresholds**

Adult: Low - specific thermoregulatory requirements (NatureServe, 2015)

**Site Fidelity**

Adult: High (NatureServe, 2015)

**Habitat Narrative**

Adult: *Myotis sodalis* hibernates primarily in caves (about 70 percent of population), also in mines and in one dam and one tunnel (USFWS 2009). Maternity sites generally are behind loose bark of dead or dying trees or in tree cavities (Menzel et al. 2001). In hibernation, limestone caves with pools are preferred. Roosts usually are in the coldest part of the cave. Preferred sites have a mean midwinter air temperature of 4 - 8°C (tolerates much broader range) (Hall 1962, Henshaw and Folk 1966), well below that of caves that are not chosen (Clawson et al. 1980). Hibernation in the coldest parts of the cave ensures a sufficiently low metabolic rate so that the fat reserves last through the six-month hibernation period (Henshaw and Folk 1966, Humphrey 1978). Relative humidity in occupied caves ranges from 66 to 95% and averages 87% throughout the year (Barbour and Davis 1969, Clawson et al. 1980). Because of these requirements, *M. sodalis* is highly selective of hibernacula. During the fall, when these bats swarm and mate at their hibernacula, males roost in trees nearby during the day and fly to the cave during the night. In summer, habitat consists of wooded or semi-wooded areas, often but not always along streams. Humphrey et al. (1977) determined that dead trees are preferred roost sites and that trees standing in sunny openings are attractive because the air spaces and crevices under the bark are warmer. Hibernating individuals characteristically form large, compact clusters of as many as 5,000 individuals (averaging 500 to 1,000 bats per cluster; Hall 1962); the clusters may average 300 individuals per square foot (LaVal and LaVal 1980). Clusters form in the same area in a cave each year, with more than one cluster possible in a particular cave (Hall 1962, Engel et al. 1976). Clustering may perform certain functions, such as protecting the central individuals from temperature changes (Twente 1955), reducing the sensitivity of most bats to external disturbance (Hall 1962), or rapid arousal and escape from predators (Humphrey 1978). Strong homing tendencies are reflected in fidelity to hibernacula (NatureServe, 2015). In order to meet their energy, thermoregulation, and social needs, adult females are colonial year-round. As a rule, Indiana bats do not cross large open areas and will follow tree lines or fencerows to reach foraging areas despite increased energy expenditures and commuting distances (Murray and Kurta 2004, Winhold et al. 2005), although exceptions to this have been noted. Adequate habitat connectivity is needed to allow for movement of bats (USFWS, 2007).

***Dispersal/Migration*****Motility/Mobility**

Adult: High (inferred from NatureServe, 2015)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Migratory or seasonal movements (NatureServe, 2015)

**Dispersal**

Adult: High (inferred from NatureServe, 2015)

**Dispersal/Migration Narrative**

Adult: Northern breeding populations migrate south to limestone cave area in Alabama, Tennessee, Kentucky, Indiana, Missouri, and West Virginia. Winter and summer habitats may be as much as 480 km apart (Layne 1978). Migrants leave hibernation sites in late March and April. Some males migrate while most remain in the general geographic vicinity of the hibernaculum throughout the summer (Hall 1962). Migration from nursery roosts occurs during late summer; arrival at hibernacula occurs from late August to early September (Barbour and Davis 1969) (NatureServe, 2015).

**Additional Life History Information**

Adult: Migration occurs in late summer (NatureServe, 2015)

***Population Information and Trends*****Population Trends:**

Decline from 1965 to 2001 (USFWS, 2007); increase from 2003 - 2007 (USFWS, 2009); 30 - 50% decline (NatureServe, 2015)

**Species Trends:**

Stable (NatureServe, 2015)

**Resiliency:**

Very high (inferred from USFWS, 2009)

**Redundancy:**

Very high (inferred from USFWS, 2009)

**Number of Populations:**

269 maternity colonies (USFWS, 2009)

**Population Size:**

468,184 (USFWS, 2009)

**Resistance to Disease:**

Low (USFWS, 2009)

**Population Narrative:**

The range-wide population estimate in 2005, 2007, 2009, and 2013 was relatively stable (ranged from 534,239 in 2013 to 590,875 in 2007), though abundance declined significantly between 2005 and 2013 in some states (e.g., New York and West Virginia) (USFWS 2013). This species has experienced a long term population decline of 30 - 50%; the population estimate in 2013 was about 60 percent of the 1960s estimate (USFWS 2013) (NatureServe, 2015). When the 2007 Plan was released, the Service had records of extant winter populations at approximately 281 hibernacula in 19 states and 269 maternity colonies in 16 states and the rangewide. The revised 2007 overall population estimate is 468,184. Since the Indiana bat's original listing and since standardized winter surveys began in the early 1980's, the Indiana bat's overall population decreased precipitously until an increasing population trend began in 2003 and continued through 2007. WNS poses a significant new threat to the species' status and may quickly reverse recent population gains (USFWS, 2009).

***Threats and Stressors***

**Stressor:** Disease and parasites (NatureServe, 2015; USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** White-nose syndrome (WNS) has quickly and significantly raised the degree of threat against the species and has lowered the species overall recovery potential (USFWS 2009). A model developed by Thogmartin et al. (2013) projected that WNS will cause a severe range-wide decline (> 86 percent) in the *M. sodalis* population over the next decade, with few of the remaining wintering populations exceeding 250 females (NatureServe, 2015). Rabies can be fatal to bats, although antibody evidence suggests that some bats may recover from the disease (Messenger et al. 2003). Rabies has never been reported in Indiana bats (Thomson 1982, Whitaker and Douglas in press), although relative to many other species few have been tested. Butchkoski and Hassinger (2002) observed hair loss in a maternity colony of Indiana bats roosting in an abandoned church in Pennsylvania. Similar atypical loss of hair occurred in little brown bats using the same roost, suggesting that the hair loss was somehow environmentally induced or perhaps caused by an unknown parasite. Although they did not observe mortality related to the hair loss, they discussed thermoregulatory implications (USFWS, 2007).

**Stressor:** Degradation of hibernation habitat (USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** There are well-documented examples of modifications to Indiana bat hibernation caves that affected the thermal regime of the cave, and thus the ability of the cave to support hibernating Indiana bats. Reasons for modifications include (but are not limited to) alterations to accommodate tourists, erection of physical barriers (e.g., doors, gates) to control cave access, and mining (particularly saltpeter). Generally, threats to the integrity of hibernacula have decreased since the time that Indiana bats were listed as endangered. Increasing awareness of the importance of cave microclimates to hibernating bats and regulatory authorities under ESA have both helped to alleviate this threat. However, the threat of collapse in mines where Indiana bats hibernate, and the threat of inadvertent modifications to caves or natural catastrophes that can impact hibernacula remain (USFWS, 2007).

**Stressor:** Loss and degradation of summer, migration, and swarming habitats (USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Loss of forest cover and degradation of forested habitats have been cited as part of the decline of Indiana bats (U.S. Fish and Wildlife Service 1983, Gardner et al. 1990, Garner and Gardner 1992, Drobney and Clawson 1995, Whitaker and Brack 2002). In some areas, such as northern Indiana, up to 97 percent of the landscape has been cleared of trees, and the absence of woodlands on the landscape certainly equates to less habitat than in prehistoric and early historic periods. It is reasonable to conclude that Indiana bat reproductive rates would be affected by alterations which lowered the quality of their maternity habitat or forced females to search for new habitat. In highly fragmented landscapes, the loss of connectivity among

remaining forest patches may degrade the quality of the habitat for Indiana bats. Conversion to agriculture has been the largest single cause of forest loss. The conversion of floodplain and bottomland forests, recognized as high quality habitats for Indiana bats, has been a particular cause of concern (Humphrey 1978). Dredging and channelization of riverine habitats to provide for agricultural drainage and flood control has also been cited as a specific threat to Indiana bat summer habitat (Humphrey et al. 1977, Humphrey 1992, Drobney and Clawson 1995). Currently, the greatest single cause of conversion of forests within the range of the Indiana bat is urbanization and development (Wear and Greis 2002; U.S. Forest Service 2005, 2006). Indiana bats are known to use forest-agricultural interfaces for foraging. In contrast, Indiana bats appeared to avoid foraging in highly developed areas. Modifications of the surface habitat around the hibernacula can impact the integrity, and in turn the microclimate, of the hibernacula. Areas surrounding hibernacula also provide important summer habitat for those male Indiana bats that do not migrate, which is thought to be a large proportion of the male population. Loss or degradation of habitat within this area has the potential to impact a large proportion of the total population (USFWS, 2007).

**Stressor:** Disturbance during hibernation (USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The primary forms of human disturbance to hibernating bats result from cave commercialization (cave tours and other commercial uses of caves), recreational caving, vandalism, and research-related activities. There are well-documented examples of disturbance resulting in declines in populations of hibernating bats (Barbour and Davis 1969). Disturbance causes the bats to arouse and use fat reserves essential for successful hibernation. Thomas et al. (1990) demonstrated that arousal from hibernation is metabolically expensive for bats; little brown bats used as much fat during a typical arousal from hibernation as would be used during 67 days of torpor. Few major hibernacula are still threatened by commercial use during the hibernation period. Impacts of recreational caving on hibernating bats are more difficult to assess and to control compared with commercial uses because commercial caves are generally gated, or have some effective means of controlling access. Increased awareness and voluntary cooperation of cavers who belonged to organized cave groups likely resulted in reduced levels of disturbance. However, it is more difficult to address visitors who are not associated with organized groups and are less likely to appreciate the sensitive nature of the cave environment and cave fauna. Disturbance of hibernating bats by cavers remains a threat in many hibernacula. Direct killing of hibernating Indiana bats by vandals has been documented throughout the species' range (Greenhall 1973, Humphrey 1978, Murphy 1987). Since the early 1980s, biennial hibernacula surveys constitute the major research-related disturbance of hibernating Indiana bats throughout most of the species range. Efforts are made to minimize the disturbance associated with these surveys (USFWS, 2007).

**Stressor:** Disturbance of summering bats (USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Research-related disturbance of summering Indiana bats has been observed. Marking-related injuries have also been reported, particularly injuries related to bat banding (Baker et al. 2001), but some researchers have concluded that the risk of banding injuries and associated

mortality of Indiana bats is slight (LaVal and LaVal 1980). Several researchers have also reported that impacts related to radiotagging of bats are minor. Mohr (1972) noted that handling of pregnant female bats may cause abortion. Generally, current procedures being used by researchers to capture, mark, and track Indiana bats during summer appear to result in minimal mortality, but continued caution and evaluation are warranted (USFWS, 2007).

**Stressor:** Predation (USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Munson and Keith (1984) conservatively estimated that an average of 1,150 bats per year were consumed by raccoons over the past 1,500 years based on raccoon feces collected in Wyandotte Cave, noting that the true predation rate is possibly several times that figure. Bat bones are routinely observed in raccoon feces in mines used as Indiana bat hibernacula in New York and the feces are often found far from the hibernacula entrance, suggesting that the raccoons may be penetrating into hibernacula specifically to seek hibernating bats (A. Hicks, pers. comm., 2006). Observations or evidence of predation by raccoons, mink (*Mustela vison*), snakes, owls, and feral and domestic cats in or at the entrance of hibernacula have been reported (Goodpaster and Hoffmeister 1950, Thomson 1982, Brack 1988, Butchkoski 2003). Evidence that hibernating Indiana bats were consumed by mice (*Peromyscus* sp.) has been observed on numerous occasions in Indiana caves, with one incident involving 13 dead Indiana bats (V. Brack, pers. comm., 2006). Indiana bats roosting under bark are susceptible to predation, both within the roost and when they depart at dusk. Humphrey et al. (1977) observed an unsuccessful attack on a foraging Indiana bat by a screech owl (*Otus asio*) near the bat's roost (USFWS, 2007).

**Stressor:** Competition (USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Researchers have observed that the overlap in roosting niches between Indiana bats and northern long-eared bats could lead to interspecific competition, particularly in habitats where roosts are not abundant (Foster and Kurta 1999), but Carter et al. (2001) reported no evidence of competition for roosts between these two species on their study area. Butchkoski and Hassinger (2002) noted no antagonistic behavior between Indiana bats and little brown bats that formed maternity roosts in the same abandoned church in Pennsylvania. Competition for roosts with other taxa has been noted. Kurta and Foster (1995) observed temporary takeover of an Indiana bat maternity roost by a pair of brown creepers (*Certhia americana*). Indiana bats temporarily abandoned a primary maternity roost tree that was being used by nesting pileated woodpeckers (*Dryocopus pileatus*) in Indiana. Competition for prey is more commonly cited than competition for roosts but is also not well documented. Whitaker (2004) studied food habits among eight species of bats in a single community and showed that main foods were most similar for the Indiana bat, little brown bat, and northern long-eared bat. The impact of the competition on populations will be exacerbated by habitat fragmentation. Loss and degradation of habitat will force more individuals of sympatric bat species (as well as other taxa with similar habitat requirements) into smaller and potentially lower quality patches of habitat (USFWS, 2007).

**Stressor:** Inadequacy of existing regulatory mechanisms (USFWS, 2007)

**Exposure:****Response:****Consequence:**

**Narrative:** Generally, existing regulatory mechanisms are more effective at protecting Indiana bat hibernacula than summer habitat. Even in situations where a maternity colony is known to be present, it is seldom known what the range extent of the colony is. Further, the conservation value of protecting a hibernaculum is easier to demonstrate and quantify compared with the value of protecting summer habitat. ESA protection extends to hibernacula that are privately owned, but recovery options are often limited on private lands. The location of most Indiana bat maternity colonies is not known; the U.S. Fish and Wildlife Service estimates that the location of approximately 270 maternity colonies has been identified, representing perhaps 6 to 9 percent of all colonies. Monitoring and management of maternity colonies on private lands can only be achieved through effective outreach to private landowners. Current regulatory mechanisms, or the manner in which those mechanisms have been implemented, have thus far not been effective in providing for this type of outreach on a broad scale (USFWS, 2007).

**Stressor:** Natural factors (USFWS, 2007)

**Exposure:****Response:****Consequence:**

**Narrative:** Natural catastrophes in hibernacula have the potential to kill large numbers of Indiana bats. Flooding events that killed large numbers of hibernating Indiana bats were reported by DeBlase et al. (1965) in Wind Cave, Breckinridge County, Kentucky (thousands of bats killed in 1964); T. Hemberger (Kentucky Department of Fish and Wildlife Resources, pers. comm., 2006) in Bat Cave, Carter County, Kentucky (3,000 bats killed in 1997); Johnson et al. (2002) in Batwing Cave, Crawford County, Indiana (several hundred bats killed in 1996); and Hicks and Novak (2002) in Haile's Cave, Albany County, New York (several hundred bats killed in 1996). Indiana bats have also frozen to death in hibernacula (Humphrey 1978). Cool temperatures also reduce the food supply for Indiana bats (Humphrey et al. 1977, Belwood 1979). The extent to which temperatures inside maternity roosts impact productivity of Indiana bats is not known. However, cold spring temperatures could further stress pregnant females, already stressed by energy demands of hibernation and migration (USFWS, 2007).

**Stressor:** Environmental contaminants (USFWS, 2007)

**Exposure:****Response:****Consequence:**

**Narrative:** By the late 1970s and early 1980s, bat mortalities caused by organochlorine pesticides (dieldrin, heptachlor epoxide) were documented in several Missouri caves (Clark et al. 1978, 1980, 1983). Although the historic studies of bat/organochlorine poisonings documented lethality, there is still no understanding of the long-term health effects of sub-lethal doses of organochlorine pesticides to individual longevity and reproductive fitness. More than 70 analytical data sets or subsets exist for analytical samples of bat carcasses, bat guano, and bat hair from caves throughout the range of the Indiana bat, including Missouri, Kentucky, New York, Indiana, Illinois, Ohio, Oklahoma, Tennessee, West Virginia, and Virginia (Martin 1992; Ryan et al. 1992; Hudgins 1993; McFarland 1998; New York State Department of Environmental Conservation et al. 2004; O'Shea and Clark 2002; BHE 2004, 2005; Adornato 2005; Sparks 2006; USFWS, Bloomington, Indiana Field Office, unpublished data, 1997-2006; USFWS, Cookeville,

Tennessee Field Office, unpublished data, 1997-2001). From this incomplete literature review and data mining effort, it is clear that there are still potentially significant organochlorine contaminant problems in several Missouri caves. Other site specific organochlorine contaminant problems may be adversely impacting Indiana bats. For example, Stansley et al. (2001) documented recent bat mortalities in localized areas where chlordane had historically been used. In the limited studies of PCBs impacts to bats (Clark and Prouty 1976, Clark and Lamont 1976, Clark 1978, Clark and Krynsky 1978) there is evidence of reproductive failures in bats. PCB transfer from the female to its young through nursing is the most important exposure route in prevalent bats. Juvenile bats typically contain the highest concentrations of PCBs in studied populations (Clark and Prouty 1976). Adult male bats may continue to bioaccumulate PCBs throughout their life and will generally have higher concentrations than adult females (Clark et al. 1975). Thousands of miles of rivers and streams throughout the range of the Indiana bat have fish consumption advisories due to PCB contamination. Many known maternity colonies are located in corn-producing areas. It is unknown whether or not this is cause for concern, yet, recent improvements in analytical chemistry techniques for monitoring the persistent organochlorine pesticides and PCBs have found low levels of chlorpyrifos in almost every recently analyzed Indiana bat carcass and guano sample (Sparks 2006). BHE (2004, 2005) also detected low levels of chlorpyrifos in several surrogate bat samples from Fort Leonard Wood and from nearby controls. This confirms that exposure to OP pesticides is routinely occurring in at least parts of the Indiana bat's range. In addition, several bats from Indiana that died under suspicious circumstances (i.e., cause of death unknown) were tested for contaminants. The following OP pesticides were detected in 3 of 9 submitted samples: diazinon, methyl parathion, and chlorpyrifos (Sparks 2006). In guano samples recently evaluated from several Indiana caves (Coon, Grotto and Wyandotte Caves), the OP pesticide dichlorvos was detected (Sparks 2006). The greatest risk to bats from pyrethroids is indirect; the significant reduction or loss of the insect prey base near a maternity colony could have an adverse impact on survival. The residual contamination from lead mining in southwestern Missouri could be sufficient to cause adverse effects to Indiana bats on the western limits of its range. In 1992 and 1993, oil pits in the oil production well fields of southwestern Indiana were surveyed for dead animals. Hundreds of dead birds and bats were found in oil pits in counties with Indiana bat summer habitat (USFWS, Bloomington, Indiana, Field Office, unpublished data, 1993-1994). Identification of oiled bat carcasses was done by the Ashland, Oregon, Forensics Laboratory, but most bats were only identified to *Myotis* spp. Spills of petroleum and crude oil can have significant short-term impacts to occupied summer habitats and likely result in take of some individual Indiana bats (USFWS, 2007).

**Stressor:** Climate change (USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Humphries et al. (2002) used climate change models to predict a northern expansion of the hibernation range of the little brown bat; such modeling would likely result in predictions of range shifts for Indiana bats as well. Potential impacts of climate change on hibernacula can be compounded by mismatched phenology in food chains (e.g., changes in insect availability relative to peak energy demands of bats) (V. Meretsky, pers. comm., 2006). Changes in maternity roost temperatures may also result from climate change, and such changes may have negative or positive effects on development of Indiana bats, depending on the location of the maternity

colony. The effect of climate change on Indiana bat populations is a topic deserving additional consideration (USFWS, 2007).

**Stressor:** Collisions with man-made objects (USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Johnson (2005) reviewed bat mortality due to collisions with turbines at wind-energy developments in the United States. Eleven species of North American bats have been recorded among the mortalities; species within the genus *Lasiurus* form a large proportion of the bats killed. No documented mortality of Indiana bats at wind farms has occurred to date. However, there is growing concern regarding the potential for bat kills given the rapid proliferation of wind farming and the large-scale mortality that has occurred at some facilities. Wind-energy developments, particularly near hibernacula or along potential migration routes where large numbers of Indiana bats could be impacted, should be evaluated as a potential threat. Bat collision mortalities have also been associated with communication towers and other manmade structures (Johnson 2005). Like collisions with wind turbines and communication towers, strikes with aircraft occur most often during the fall migration. Russell et al. (2002) verified that an Indiana bat was killed by collision with a vehicle on a Pennsylvania road. There is no implication to date that Indiana bats are particularly susceptible to such collisions, but they may represent a threat to local populations under certain conditions (USFWS, 2007).

## ***Recovery***

### **Reclassification Criteria:**

1. Permanent protection of 80% of all Priority 1 hibernacula in each Recovery Unit (USFWS, 2009).
2. A minimum overall population estimate equal to the 2005 population estimate of 457,000 bats (USFWS, 2009).
3. Predicted continued positive population growth rate at each of the most populous hibernacula in each RU (using a linear regression with 90% confidence interval through 5 most recent population estimates as a means of predicting trend over the next 10-year period) (USFWS, 2009).

### **Delisting Criteria:**

1. Protection of a minimum of 50% of Priority 2 hibernacula in each Recovery Unit (USFWS, 2009).
2. A minimum overall population estimate equal to the 2005 population estimate of 457,000 bats (USFWS, 2009).
3. Positive population growth rates at a minimum of 80% of all Priority 1A hibernacula/complexes as evidenced by a positive slope of a linear regression through the 5 most recent population estimates post-reclassification (USFWS, 2009).

### **Recovery Actions:**

- Develop and implement public information and outreach program (USFWS, 2007).
- Conserve and manage hibernacula and their winter populations (USFWS, 2007).
- Conserve and manage summer habitat to maximize survival and fecundity (USFWS, 2007).
- Plan and conduct research essential for recovery (USFWS, 2007).

***Conservation Measures and Best Management Practices:***

- Within the next year (and prior to the next 5-year review), the Service plans to approve and finalize a revision to the Indiana Bat Recovery Plan. The revised plan will certainly need to address the newly emerging threat of WNS at whatever level is possible given the knowledge base at that time. Although WNS was not identified/addressed as a threat in the 2007 Plan, the population-based recovery criteria in the 2007 Plan are likely to remain as one of the most effective means of assessing the WNS-related mortality and recovery from WNS in the future. While we have a successful means of monitoring WNS in Indiana bat hibernacula, additional actions are necessary to help minimize the impacts of WNS on Indiana bats if possible. Additional research to understand the causes and potential spread of WNS should be initiated immediately. Research of potential management actions aimed at minimizing the potential spread of WNS must continue to be supported and effective actions implemented and adapted as we learn more about the cause(s) of WNS-related mortalities. As we understand more about the cause(s) and vectors of WNS, management actions to help minimize mortalities should be investigated and implemented (i.e., an adaptive management approach will be taken). Public education/outreach efforts about WNS must also continue (USFWS, 2009).
- It is also apparent from this Review that additional attention should be placed on securing permanent/long-term protection of both Priority 1 and Priority 2 hibernacula. Several Priority 1 hibernacula would satisfy Reclassification Criterion 1 if their cave/mine entrances were gated or if appropriate buffer zones were delineated and protected (USFWS, 2009).
- We also recommend that the Service pursue some of the highest priority recovery actions identified within the 2007 Plan that would improve our understanding of the Indiana bat's population status and progress towards recovery. In particular, actions 2.4.1, 3.1.2 - 3.1.6, and 3.2.2.1 should ideally be completed prior to the next 5-year review (USFWS 2007). These specific actions involve estimating population biology parameters and demographics such as juvenile and adult survivorship, reproductive success, and developing population models for the Indiana bat. We recommend a population viability analysis be conducted that would model the population impacts of discrete catastrophes and/or variable WNS mortality scenarios across the species' range. The Service is currently planning such a population analysis to be conducted in 2009-2010 (USFWS, 2009).
- Finally, it is apparent from conducting this Review that the Service will need to continue to improve and maintain a significant, ongoing level of coordination with bat surveyors, the caving community, and other conservation and research partners in order to maintain the Service's hibernacula and population databases and in order to successfully coordinate and implement the recovery actions outlined in the 2007 Plan across the species' wide range (USFWS, 2009).

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Bloomington, Indiana.

## **SPECIES ACCOUNT: *Neotoma floridana smalli* (Key Largo woodrat)**

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### ***Species Taxonomic and Listing Information***

**Listing Status:** Endangered; Southeast Region (R4) (USFWS, 2015)

### **Physical Description**

The color of the Key Largo woodrat is described as sepia or grey-brown above shading into cinnamon on the sides, with cream or white ventral coloration. The forefeet are white to the wrist and the hindfeet are primarily white to the ankles. The Key Largo woodrat has large ears, protuberant eyes, and a hairy tail. The head-and-body-length of the Key Largo woodrat ranges from 120 to 230 mm, their tail length ranges from 130 to 190 mm, and their hindfoot length ranges from 32 to 39 mm. Males, on average, weigh 258 g, while the females tend to be much smaller, weighing only 210 g (Hersh 1981) (USFWS, 1999).

### **Taxonomy**

Based on mtDNA data, subspecies *smalli* of Key Largo, Florida, has diverged little from subspecies *floridana* of peninsular Florida, but Hayes and Harrison (1992) suggested the retention of *smalli* as a subspecies based on behavioral characteristics and the disjunct distribution. (NatureServe, 2015)

### **Current Range**

North Key Largo, Monroe County, Florida. Formerly found throughout Key Largo, but range has been reduced by habitat destruction. Introduced on Lignum Vitae Key. Disjunct from other populations in peninsular Florida by a gap of 240 km (NatureServe, 2015). Until Hurricane Irma, there had been no recent, significant changes in the known distribution of the KLWR since the previous review. (Further assessment is needed to determine if/how the distribution of the KLWR has changed since the storm.) While the proportion of the area occupied has fluctuated, this subspecies remains distributed throughout the tropical hardwood hammock habitat on public lands in the northern third of Key Largo. Recent surveys and ongoing monitoring efforts suggest a spatial trend of KLWR persisting at higher densities within the central core of the CLNWR and DJSP, and lower densities at the northern and southern extremities. Without implementation of recovery actions on public lands, such as invasive species control, this trend is expected to continue and increase in severity. For instance, the northern extent of the KLWR's currently known range is threatened by a large feral cat colony, and development and free roaming and feral cats encroach at the southern extent. No KLWR have been captured during limited survey efforts on lands adjacent to known occupied areas; however, KLWR may be persisting at low densities. Feral cat colonies are prevalent in these areas within Key Largo which may reduce KLWR densities and nest building activities, thereby reducing detectability (USFWS 2018).

### **Critical Habitat Designated**

Yes;

### ***Life History***

### **Feeding Narrative**

Adult: Eats leaves, but, seeds, and fruit (Layne 1978).; Food Habits: Herbivore (Adult, Immature), Granivore (Adult, Immature), Frugivore (Adult, Immature) (NatureServe, 2015)

### **Reproduction Narrative**

Adult: Litter size 1-4 (average 2); average of 2 litters per year (Layne 1978).; Density was estimated at 7.6/ha over 851 ha at Key Largo (Humphrey 1988). Abundance is poorly correlated with abundance of sign (Humphrey 1988). Solitary (except females with young).; (NatureServe, 2015)

### **Habitat Narrative**

Adult: Mature, undisturbed subtropical hardwood (hammock) forest. Optimal habitat: dominant trees must be at least 25-30 cm in diameter (Layne 1978); rat abundance increases with hammock maturity. Builds and nests within a large stick house on the ground; houses may remain in use for many years and often are built around a stump, log, boulder, or other similar object; may occupy old buildings (Layne 1978). (NatureServe, 2015)

### ***Dispersal/Migration***

### **Motility/Mobility**

Adult: Nonmigrant: Y; Local migrant: N; Distant migrant: N; (NatureServe, 2015)

### **Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigrant: Y; Local migrant: N; Distant migrant: N; (NatureServe, 2015)

### **Dispersal**

Adult: Nonmigrant: Y; Local migrant: N; Distant migrant: N; (NatureServe, 2015)

### **Dispersal/Migration Narrative**

Adult: Nonmigrant: Y; Local migrant: N; Distant migrant: N; (NatureServe, 2015)

### **Additional Life History Information**

Adult: Nonmigrant: Y; Local migrant: N; Distant migrant: N; (NatureServe, 2015)

### ***Population Information and Trends***

### **Population Trends:**

Recent surveys (2012 to present) appeared to indicate an increasing and currently stable population trend prior to Hurricane Irma (Cove et al. 2017a). However, according to a preliminary post-Hurricane Irma assessment (2017/2018), there is evidence of a significant population reduction (Cove 2018). Further assessment is needed to determine if the population has declined or is stable (USFWS 2018). Recent analyses of trapping data in the periods 1995 to 1996, 2000 to 2001, 2005, and 2007 suggest that the KLWR population had a positive growth rate ( $\lambda=1.070$ ) in all but the 2005 study period. Abundance estimates from 2008 to 2011 suggest a steep decline from 2008 (estimate of 693 individuals) to 2010 (estimate of 78 individuals), but an increase in 2011 (estimate of 256 individuals). From 2013 to 2015, KLWR distribution was assessed by sampling supplemental nests throughout KLWR habitat. The probability of nests being occupied increased annually from 23% in 2013 to 30% in 2014 and 37% in 2015. While observations of stick nests steadily increased from 2007 to 2012, beginning in 2015, there

appeared to be a notable increase in natural stick nests, large stick nests, and stick-stacking on most supplemental 6 nest structures within Crocodile Lake National Wildlife Refuge (CLNWR) and Dagny Johnson Key Largo Hammock Botanical State Park (DJSP). This perceived increase in stick nest building behavior is related to an increasing use of these nests (e.g., increased abundance). In 2017, trapping surveys yielded a success rate of 4.14 KLWR per 100 trapnights, with KLWR present on 21 of 32 grids (66 percent). This far exceeds rates documented less than 10 years ago (1.5 KLWR per 100 trapnights; 0.87 KLWR per 100 trapnights), and signifies an increase in trap success to rates not documented since the 1980s. Preliminary data from a post-Hurricane Irma population assessment have indicated a significant decline (USFWS 2018).

**Number of Populations:**

1 - 20 (NatureServe, 2015)

**Population Size:**

1 - 1000 individuals (NatureServe, 2015)

**Population Narrative:**

Humphrey estimated 6500 individuals in 1987, 10 times previous estimates. Ca. 850 ha. of occupied habitat on north Key Largo; 20 small occurrences formerly were one big one prior to habitat fragmentation. Introduced to Lignum Vitae Key. (NatureServe, 2015)

**Threats and Stressors****Stressor:****Exposure:****Response:****Consequence:**

**Narrative:** Destruction of tropical hammocks for development is cause of population decline. Predation by introduced pythons. (NatureServe, 2015)

**Stressor:** Disease

**Exposure:** Raccoon roundworm, New Guinea flatworm and *Toxoplasma gondii*

**Response:** While the broader effects on KLWR populations are unknown, infection may have negative consequences such as reduced individual health and fitness.

**Consequence:** Population decline.

**Narrative:** While no evidence of raccoon roundworm was found in Key Largo, surveillance for this parasite should not be considered obsolete. Even low levels of this parasite on the landscape can impact woodrat populations. The New Guinea flatworm (*Platydemus manokwari*) is a large predatory flatworm. This species is highly invasive, named one of the "World's 100 Worst Invaders", and primarily affects populations of land snails. In 2017, several specimens have been collected within KLWR habitat at DJSP incidentally during a herpetological coverboard survey. This species is a vector for the rat lungworm, which can impact rodent populations and cause meningitis in humans. No testing for this parasite in KLWR has been conducted and no surveillance for this threat is underway at this time. *Toxoplasma gondii* is a protozoan that causes toxoplasmosis. Cats are the only definitive host known for *T. gondii*, and the parasite is spread to other warm-blooded animals through the cats' feces (or coming into contact with water, food, or soil contaminated by cat feces). Consequently, this disease is a concern for all mammals in the Keys due to the presence of feral cat feeding colonies and high density of outdoor cats. This

parasite is well documented as a public health concern and often lethal in rats and mice. Furthermore, *Toxoplasma* infection has been shown to make rats and mice bolder and even attracted to cat urine, which could indirectly strengthen free-roaming cat impacts on woodrats. The threat of disease to such a limited population is high, and investigations regarding the occurrence and potential impact of these possible vectors are underway (USFWS 2018).

**Stressor:** Predation

**Exposure:** Raptors, corn snakes (*Elaphe guttata*), diamondback rattlesnakes (*Crotalus adamanteus*), Florida black racers (*Coluber constrictor priapus*), Keys rat snakes (*Elaphe obsoleta deckerti*), owls, and raccoons (*Procyon lotor*). Nonnative predators include free-roaming domestic cats (*Felis catus*), fire ants (*Solenopsis invicta*), and Burmese pythons (*Python molurus bivittatus*)

**Response:** Mortality.

**Consequence:** Population decline.

**Narrative:** One of the largest feral cat colonies is operated adjacent to the Dagny Johnson Key Largo Hammocks State Botanical Site, yet there have not been comprehensive or continuous free-roaming cat control efforts in place within the range of the KLWR. Limited cat control has been undertaken in the past on Crocodile Lake National Wildlife Refuge and Dagny Johnson Key Largo Hammocks State Botanical Site. However, it was usually instituted on a small scale, and only targeted a few individual cats. To aid recovery efforts of both KLWR and Key Largo cotton mice, the Service funded a successful largescale control effort that was conducted in the winter of 2004. Raccoons, while a natural predator, are attracted to areas with feral cat colonies due to regular feedings. This factor, in addition to the general attraction of raccoons to garbage, has likely led to elevated densities of raccoons in North Key Largo. Recent research has found that 22 percent of feral cats sampled in Key Largo had a diet primarily consisting of wild prey (over 50 percent wild prey). Considering the wildlife community in Key Largo, and cameras documenting cats with rodents in their mouths, these cats are likely preying upon endangered small mammals, including KLWR. Predator management efforts continue, but are concentrated on the CLNWR half of the KLWR's range, which limits overall effectiveness. Sharing efforts and responsibility for this issue among landowners and agencies would greatly advance enforcement efforts and outreach messaging. Seven non-native Burmese pythons have been captured in Key Largo since April 2007, and predation of KLWRs by Burmese pythons was documented in 2007. An eradication program for this non-native predator is in place, but largely relies on reports from the public. Intra-agency partnerships have developed to assess ecological risks, encourage responsible pet ownership, organize exotic pet amnesty days and media campaigns, and form a rapid response team. To specifically protect the KLWR, the Service has funded a USGS project that includes a multi-faceted effort to detect and control Burmese pythons on Key Largo using visual surveys and several types of experimental traps to capture pythons. Over 25 Burmese pythons have now been captured in Key Largo since 2007. Three 18-inch hatchling Burmese pythons were found and removed from northern Key Largo, near DJSP, in 2016. This was the first sighting of pythons of this age and size in Key Largo, and presents evidence of a breeding population of pythons in Key Largo. Four adult pythons were captured from the remains of the Nike missile base in 2017. The three, 8-foot pythons and one, 16-foot python were found within missile bunkers. An additional adult python was captured during a debris removal/ habitat restoration project at the Nike site in 2016. Since Hurricane Irma, land managers have anecdotally seen an increase in both feral cat and python observations. Cats may have been abandoned by owners evacuating the Keys, or upon their return. The storm may also have served as a dispersal event for the Burmese pythons, aiding their travel from the Everglades region. With

relatively mild habitat impacts from the storm, a possible population increase of both of these known KLWR predators may pose significant indirect effects of the storm on the wildlife in Key Largo. Predation of KLWRs where recruitment is sufficient and suitable habitat is available is not a concern. Conversely, increased predation pressure on isolated populations from natural and non-native predators can have a substantial impact. The drastic decline of Allegheny woodrats (*N. magister*) in Pennsylvania was attributed primarily to predation by great horned owls (*Bubo virginianus*) and exposure to raccoon roundworms. In addition, due to their moderate size and mostly terrestrial mode of life, KLWRs may be particularly vulnerable to predation. In light of the increased level of native predators, the addition of non-native predators, and the direct relation of this threat to mortality, the severity and scope of this threat are high. The threat of both disease and predation (by non-native, invasive species) is increasing. The needs to survey for, manage, and understand the population level effects of potential disease agents and predators are significant (UFWFS 2018).

**Stressor:** Inadequacy of existing regulatory mechanisms.

**Exposure:** FEMA flood insurance consultation; State and county regulations.

**Response:** Mortality.

**Consequence:** Population decline.

**Narrative:** FEMA flood insurance consultation. On August 25, 1994, the United States District Court for the Southern District of Florida directed the Federal Emergency Management Agency (FEMA) to consult with the Service to determine whether implementation of the National Flood Insurance Program in Monroe County was likely to jeopardize the continued existence of federally listed species (Case No. 90-10037-CIV-MOORE). In 2003, the Service issued a jeopardy biological opinion with reasonable and prudent alternatives that required Monroe County to consult with the Service before issuing building permits in suitable habitat for listed species. Thus, in recent years, the Service provided technical assistance on pertinent projects (virtually all building applications on private parcels throughout the range of the KLWR, excluding Coastal Barrier Resource Act zones). On September 9, 2005, the Court ordered an injunction against FEMA issuing flood insurance on any new developments in suitable habitat of federally listed species, and required the Service to submit a revised biological opinion within nine months (deadline later extended to August 9, 2006). Because the Court ruled that the 2003 reasonable and prudent alternatives were invalid, Monroe County was no longer required to consult with the Service before issuing building permits in suitable habitat and the Service suspended technical assistance on building permit applications. The Service finalized its reanalysis of the National Flood Insurance Program in Monroe County, and provided a biological opinion to the Court on August 8, 2006 (Service 2006). The biological opinion provides a revised strategy for implementing regulatory actions pertaining to threatened and endangered species. This strategy includes clarification of FEMA's oversight role and a more comprehensive strategy of evaluating potential impacts. The latter incorporates a lot-by-lot assessment of potential impacts that takes into account the limitations on development imposed by the County's Rate of Growth Ordinance (ROGO) system with its new designations of geographical tiers. In the biological opinion, the Service concluded that continued administration of the National Flood Insurance Program in the Keys was not likely to jeopardize the continued existence of the KLWR. The Court will determine whether to accept the biological opinion and whether to lift the prohibition on FEMA's issuance of flood insurance in Monroe County. State and county regulations. The KLWR is listed by the FWC as endangered (Chapter 39-27, Florida Administrative Code). This legislation prohibits take, except under permit, but does not provide any direct habitat protection. Wildlife habitat is protected on FWC wildlife management areas and wildlife environmental areas according to

Florida Administrative Code 68A-15.004. Florida Park Service regulations prohibit take of specimens and destruction of vegetation (i.e., habitat) on park property without a permit. The State of Florida has compelled the Monroe County Board of Commissioners to strengthen controls on land use since at least 1975 when the Keys were designated an Area of Critical State Concern. A critical regulatory factor is the level of service on U.S. Highway 1 as it relates to hurricane evacuation time. The County developed a (ROGO) that, as of March 2006, incorporated a land tier system that specifically designates areas of native habitat for listed species, including the KLWR. The process made it more costly to destroy habitat and now discourages development in unfragmented habitat, steers available permit allocations to disturbed areas that are poor habitat for native fauna, and implements a land acquisition program for areas with native vegetation, including KLWR habitat. Monroe County's Comprehensive Land Use Plan (March 2007) states that development within hammock "shall be reviewed to ensure the functional integrity of the entire hammock" and development proposals within this habitat type "shall identify the extent to which the area is habitat for threatened or endangered species" and adverse impacts to "the functional integrity of the hammock or pineland in which development is to be undertaken, the developer shall provide for mitigation in an amount greater than the area disturbed in the form of replanting disturbed areas with native species or by the acquisition and preservation, including donations, of land containing comparable quality and character of vegetation as the area disturbed." Pressure to develop remaining residential and commercial land within the range of the KLWR continues. However, development is subject to regulatory oversight by Monroe County (e.g., the ROGO), the State (e.g., designated an Area of Critical State Concern), and the Service (e.g., ESA consultation, presumably including continued consultation with FEMA regarding administration of the National Flood Insurance Program). Regulatory mechanisms have reduced the threat of further habitat loss in north Key Largo. The Monroe County Animal Control Ordinance does not restrict free-roaming cats. This ordinance could be changed, or a Keys-wide ordinance could be added to limit cats to living indoors (or otherwise contained), particularly in sensitive environmental areas (USFWS 2018).

**Stressor:** Competitors.

**Exposure:** Black or Norway rats.

**Response:** Mortality.

**Consequence:** Population decline.

**Narrative:** The presence of competitors, particularly non-native species, is a significant influence on habitat suitability. Trash dumping occurs throughout the KLWR's range and attracts human commensals. In the past, black rats (*Rattus rattus*) were captured at equal or greater numbers as KLWRs on hammock study sites (Hersh 1981) and thought to be a serious competitor, but subsequent trapping sessions yielded very few captures of black or Norway rats (*Rattus norvegicus*) (Barbour and Humphrey 1982; Goodyear 1985). In a 2013 camera trap survey of KLWR supplemental nests, black rats were only detected at two of the nest structures, compared to KLWR detected at 65 nests (Cove et al. 2017a). However, more recent live-trapping surveys resulted in the capture of more black rats ( $n = 108$ ) than woodrats ( $n = 98$ ) in spring 2017 (Cove 2017). These black rats were evenly distributed throughout the protected habitats with the highest densities at the northern and southern boundaries of the CLNWR and DJBSP. There is growing evidence that black rat populations have increased in recent years, possibly in response to exotic predator management, which should warrant further examination of direct and indirect effects of black rats on KLWR. Gambian giant pouch rats (*Cricetomys gambianus*), the largest murids, were unintentionally released in Marathon, Florida in 1999. Possible sightings on Key Largo have not been confirmed with trapping (Engeman et al. 2006), but due to their large size,

high fecundity, and similar food and nest site requirements, their impact on KLWR would be extensive. An eradication program initiated in Marathon appears to have been successful, though the pouch rats could emigrate by several means (Engeman et al. 2006). Furthermore, the hurricanes of 2005 may have assisted in their dispersal to nearby islands. The severity of this threat is high, while the scope remains moderate. Over 190 free-ranging Gambian pouch rats have been documented in the Florida Keys, with the majority in Grassy Key, where a captive-breeding colony still exists. Confirmed accounts range from Islamorada to Key West, and the most recent account was in 2017 (USFWS 2018).

**Stressor:** Severe weather events.

**Exposure:** Hurricanes

**Response:** Damage to habitat.

**Consequence:** Population decline.

**Narrative:** Hurricanes influence vegetational succession in the Florida Keys. Undisturbed hammocks are presumably more resistant to storms than hammocks that have been fragmented or have had surrounding mangrove and transitional vegetation removed. Damage to habitat from past hurricanes has included windshear, significant canopy loss, uprooting of large trees, understory damage, and significant soil disturbance. Extensive damage represents habitat loss to KLWR, but some disturbance serves to open habitat and allow for greater plant diversity. The severity and scope of this threat are variable and stochastic (USFWS 2018).

**Stressor:** Climate change.

**Exposure:** Sea level rise.

**Response:** May alter habitat.

**Consequence:** Population decline.

**Narrative:** Sea level rise has been shown to affect conversions of upland communities with low soil and moisture salinities to communities comprised of more salt tolerant plant species and higher soil and groundwater salinities (Ross et al. 1994). This phenomenon may potentially result in the loss of suitable KLWR habitat through inundation or vegetative species composition changes. The general effects of sea level rise within the range of the KLWR will depend upon the rate of rise and landform topography. However, the specific effects across the landscape will be affected by complex interactions between geomorphology, tides, and fluctuations in energy and matter. These effects have yet to be simulated and projected for the range of the KLWR. The imminence of this threat is low, but the severity remains unknown. The KLWR's distribution appears to be undergoing constriction due to expanding mangrove areas and inland human infrastructure (FWC 2017). This constriction may be pushing ideal habitat inland toward roads which could lead to vehicle-related mortalities. Recent evidence of KLWR using mangrove pods for nesting material, instead of the typical hardwood sticks, may constitute evidence of these habitat changes. Recent climate change modeling suggests that the tropical hardwood hammock in Key Largo is less vulnerable to sea level rise than other areas in the Florida Keys (FWC 2017). However, at three to four feet of sea level rise, water levels fragment habitat and several habitat bottlenecks materialize; effects further exacerbated by the highway running through the KLWR's range (CR905). This level of sea level rise is forecasted to occur in 43 to 80+ years (2060-2100; NOAA 2017). Additionally, a portion of hammock areas affected by sea level rise will likely transition into mangrove forest, causing further reduction of KLWR habitat (USFWS 2018).

### ***Recovery***

**Reclassification Criteria:**

The criteria included in the approved recovery plan to reclassify the KLWR from endangered to threatened are: 1) further loss, fragmentation, or degradation of suitable, occupied habitat must be prevented; 2) native and nonnative nuisance species must be reduced by 80 percent; 3) all suitable, occupied habitat on priority acquisition lists on Key Largo must be protected either through land acquisition or cooperative agreements; 4) tropical hardwood hammocks that form the habitat of the Key Largo woodrat must be managed on protected lands to eliminate trash and control exotics; and 5) stable (rate of increase equal or greater than 0.0 as a 3-year running average for 6 years) populations of the Key Largo woodrat must be distributed throughout north Key Largo and three additional, stable, populations established elsewhere within the historic range. These criteria have not been met. Habitat degradation and loss has continued and threats from nonnative invasive species have increased. A working group has been developed to address new issues and persistent threats. While population monitoring has indicated a population increase since the last review, the threat of nonnative predators has also increased. Several projects restoring hardwood hammock have been initiated and currently are underway, but further habitat loss and degradation has occurred (some due to Hurricane Irma). No additional populations have been established. Therefore, we have not met the criteria to reclassify KLWR (USFWS 2018).

**Delisting Criteria:**

The Key Largo woodrat will be considered for delisting when all the following criteria have been met: 1. Five (5) additional populations are established or discovered that exhibit a stable or increasing population trend for multiple generations, and natural recruitment (Factor A). 2. The five (5) new populations should be located outside of Dagny Johnson Botanical Preserve State Park and Crocodile Lake National Wildlife Refuge and be connected to the extent that genetic diversity can be naturally maintained without translocations or captive breeding (Factor A, D, E). 3. Non-native species (e.g., Burmese pythons, tegus, free-roaming pets, black rats) are reduced or eliminated to a degree that predation and competition is low enough for KLWR to remain viable for the foreseeable future. (Factor C, D) 4. When in addition to the above criteria, it can be demonstrated that habitat loss associated with sea level rise and development are diminished such that enough suitable habitat remains for KLWR to remain viable for the foreseeable future. (Factor E) (USFWS 2018b).

**Recovery Actions:**

- Old and abandoned roads bisecting hammock habitat should be restored to native vegetation. Research may be warranted to develop restoration techniques effective in this unique environment. (Partially complete / ongoing)
- The 1999 Recovery Plan should be revised and updated to reflect the current status and threats to the KLWR, and recovery criteria, objectives, and tasks should be developed or revised. (Not initiated)
- Genetic analyses should be conducted to provide further insight into the current KLWR population. Information on the genetic diversity of the population and the genetic makeup of individual KLWRs will provide insight into the current status of the population. (Partially complete/ongoing)
- Opportunities to convey the importance of hammock habitat to the public should be sought and pursued. Interpretive signs could be designed and distributed to public land managers on North Key Largo. In addition, an outreach/education program focused on the threats free-roaming cats and exotic pets pose to wildlife should also be developed. (Partially complete/ongoing)
- Appropriate parcels for land acquisition should be identified using current knowledge of KLWR movements and habitat use. (Not initiated)

Captive propagation and reintroduction efforts should continue to develop techniques and methods appropriate for KLWRs. (Partially complete) • Further examination of nest sites potentially limiting the KLWR may be warranted. Natural nest materials may be provided in areas occupied by KLWRs to aid in natural nest construction (Ongoing). Ex situ research may be appropriate to determine possible causes for nest site selection (Not initiated). • Information concerning the diet of KLWR would aid in habitat restoration, land acquisition, and captive propagation efforts (Partially complete). Identifying foraging patterns may allow for better assessment of KLWR's perception and response to predation risk and provide detailed movement information (Not initiated). Data from previous research could be reanalyzed to provide insights into habitat use (Not initiated). In addition, vegetation surveys measuring several habitat parameters may be important to determine factors influencing habitat use (Partially complete). • Additional information is required concerning potential disease agents and health problems that may afflict KLWRs. Rodents from Key Largo should be screened for a variety of diseases, when considered appropriate. Tentative agreements with the University of Florida - College of Veterinary Medicine, Gainesville, Florida and with the National Wildlife Health Center, Madison, Wisconsin, would allow for such investigations. (Not initiated) • Research focused on determining the relative abundance of KLWR predators, their influence on KLWR behavior, and their effect on survival and recruitment rates is warranted. Predator management strategies and/or more comprehensive predator control should be investigated if appropriate. (Partially complete/ongoing) \* Evaluate the basis of KLWR population fluctuations and consider environmental, stochastic, and habitat-associated influences as possible drivers. 23 \* Develop potential adaptation strategies to moderate or delay effects of sea level rise on KLWR. For example, increase connectivity where sea level is likely to cause fragmentation. \* Develop effective, comprehensive means to manage nonnative, invasive species and implement significant efforts aimed to eradicate these species from Key Largo and prevent recolonization. • Continue efforts to restore KLWR habitat, particularly in concert with python removal efforts. • Use monitoring nests (natural or supplemental) or other techniques to assess habitat use or preferences among hammock age classes. • Evaluate KLWR presence outside their currently known range. • Assess the genetic connectivity between previously identified subpopulations. • Work with county, state and federally-owned (and/or managed) lands in Key Largo to eliminate or reduce impacts of feral cat colonies on these lands. • Develop techniques to identify owners/care-givers of individual cats for more effective enforcement of policies (i.e., collars or microchips). • Determine the direct and indirect effects of black rats on KLWR. • Evaluate the short-term and long-term impacts from Hurricane Irma on KLWR and identify remedies. • Determine whether supplemental nests provide greater protection from predators, require less maintenance, and last longer than natural nests. • Initiate disease monitoring for raccoon roundworm and Toxoplasmosis (USFWS 2018).

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## **SPECIES ACCOUNT: *Neotoma fuscipes riparia* (Riparian woodrat (=San Joaquin Valley))**

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### ***Species Taxonomic and Listing Information***

**Listing Status:** Endangered; 02/23/2000; California/Nevada Region (Region 8) (USFWS, 2016)

### **Physical Description**

The riparian woodrat (*Neotoma fuscipes riparia*), also known as the San Joaquin woodrat, is a medium sized rodent in the Cricetidae family. It is one of eleven subspecies of dusky-footed woodrats (*N. fuscipes*). Dusky-footed woodrats are predominantly gray and cinnamon above and whitish beneath. Their tails are well furred, not scaly like those of common nonnative black rats (*Rattus rattus*), which are in a completely different family, Muridae. Adult riparian woodrats weigh from about 7 to 14 ounces. The riparian woodrat can be distinguished from other subspecies by having white rather than dusky hind feet. It is also larger, lighter and more grayish. Its tail is more distinctly bicolored. (USFWS, 2016)

### **Taxonomy**

Currently available research suggests that the riparian woodrat should be classified as *Neotoma macrotis riparia* rather than *Neotoma fuscipes riparia*. Matocq's (2002a) genetic study of *Neotoma fuscipes* as well as her analysis of *N. fuscipes* and *N. macrotis* (2002b) suggest that *Neotoma macrotis* be recognized as a distinct species from *N. fuscipes* (Matocq, pers. comm. 2011). Mammologists have largely accepted this new taxonomy (Kelly, pers. comm. 2011b, Matocq, pers. comm. 2011). Matocq (2002a) raised the possibility of hybridization as a possible explanation for the genetic results obtained from the CMSP population of riparian woodrats, which, based on mitochondrial DNA, were shown to be more closely related to *N. macrotis* of the Sierra Nevada foothills to the east of CMSP, contrasting with earlier assumptions that the riparian woodrat had originated from *N. fuscipes* found west of CMSP in the coast ranges (Matocq, pers. comm. 2011). Unpublished research by Matocq using both mitochondrial and nuclear genetic markers may clarify the relationship to other woodrat populations in the coast ranges and the Sierra Nevada foothills, and should be examined closely during the next 5-year review (Kelly et al. 2011, Kelly pers. comm. 2011b, Kelly pers. comm. 2011c, Matocq, pers. comm. 2011). (USFWS, 2012)

### **Historical Range**

Known historical distribution included areas along the San Joaquin, Stanislaus, and Tuolumne rivers, and Corral Hollow, in San Joaquin, Stanislaus, and Merced counties. (NatureServe, 2015)

### **Current Range**

Lower San Joaquin Valley, California (Williams and Kilburn 1984); presently known to be extant only at Caswell Memorial State Park (Williams 1993). (NatureServe, 2015)

### **Distinct Population Segments Defined**

No

### **Critical Habitat Designated**

No;

***Life History*****Feeding Narrative**

Adult: Woodrats are, for the most part, generalist herbivores. They consume a wide variety of nuts and fruits, fungi, foliage, and some forbs (Linsdale and Tevis 1951). (USFWS, 1998)

**Reproduction Narrative**

Adult: Not available.

**Environmental Specificity**

Adult: Low (USFWS, 2012)

**Habitat Narrative**

Adult: *N. fuscipes* (and *N. macrotis*) prefer habitat with a large amount of overall structure, with both understory vegetation and overstory cover (Gerber et al 2003). Although no studies have been performed to determine the specific habitat needs of the species, at Caswell Memorial State Park, riparian woodrats are most often observed in areas with a valley oak overstory and a wild grape (*Vitus californica*), willow (*Salix* sp.), blackberry (*Rubus discolor* or *Rubus ursinus*), wild rose (*Rosa californica*), or coyote bush (*Baccharis pilularis*) understory (Kelly et al. 2011). In addition, the best quality habitat appears to contain a significant midstory component of vines or small trees, which the riparian woodrat is thought to utilize in order to access the canopy, where they do a substantial amount of their foraging (Kelly et al. 2011). Other important components of riparian woodrat habitat include wooded or shrub-covered upland refugia to facilitate escape from flood events while preventing predation, and downed trees and dead snags that are used in place of stick lodges (Kelly et al. 2011). At Caswell Memorial State Park, riparian woodrats also make houses of sticks and other litter (Williams 1993). Houses typically are placed on the ground against or straddling a log or exposed roots of a standing tree and are often located in dense brush. Nests also are placed in the crotches and cavities of trees and in hollow logs. (USFWS, 1998; USFWS, 2012)

***Dispersal/Migration*****Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory (NatureServe, 2015)

**Dispersal/Migration Narrative**

Adult: Not available.

***Population Information and Trends*****Population Trends:**

Not available.

**Resiliency:**

Low (inferred from NatureServe, 2015)

**Redundancy:**

Low (inferred from NatureServe, 2015)

**Number of Populations:**

2 (USFWS, 2012)

**Population Size:**

~500 individuals (NatureServe, 2015)

**Population Narrative:**

Williams (1993) estimated the population of the single known occurrence at 437 individuals. There are two known populations in the same general area of California: one within Caswell Memorial State Park and the other approximately five miles away within the San Joaquin River National Wildlife Refuge (Kelly et al. 2009, Kelly et al. 2011). (NatureServe, 2015)

***Threats and Stressors***

**Stressor:** Habitat loss (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** At the time of listing, the threats under factor A were a large scale destruction of riparian habitat due to urban, commercial, and agricultural development, combined with flood control and reclamation activities such as river channelization, levee construction, dam construction, water diversion, and groundwater pumping (Service 2000a). Areas surrounding levees have been entirely cleared of riparian vegetation and the topography has been leveled and planted with row crops, vineyards, and orchards, leaving no avenues for the riparian woodrat to disperse from its current occupied habitat. Levee construction and stream channelization has degraded the quality of the remaining habitat by increasing the size and duration of flood events within the levees (Service 2000a). These threats are largely the same as they were at the time of listing; however, there may be increased fire risk at CMSP because CMSP staff do not currently manage fire fuel loads within the park (Karlton pers. comm. 2011b). (USFWS, 2012)

**Stressor:** Disease (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Bubonic plague was listed as a possible disease threat in the 2000 listing rule. The small population size and extremely limited geographical distribution of the riparian woodrat causes it to be at heightened risk from an epidemic event (Service 2000a). There have been no reports of disease related mortality since the species was listed. (USFWS, 2012)

**Stressor:** Predation (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Predation from coyotes (*Canis latrans*), gray foxes (*Urocyon cinereoargenteus*), long tailed weasels (*Mustela frenata*), raccoons (*Procyon lotor*), feral domestic cats (*Felis domesticus*) and dogs (*Canis lupus familiaris*), owls (*Strigidae*), and other raptors was known to occur in the

2000 listing rule (Kelly et al. 2009, Service 2000a). Since listing, preliminary research by Kelly et al. (2009) indicates that exotic black rats (*Rattus rattus*) may compete with riparian woodrats for food resources or habitat. This research also suggests that black rat presence negatively impacts riparian woodrat reproductive success, although the mechanism for this interaction has not been explored. Nonetheless, Kelly et al. (2009) hypothesize that black rats may prey on juvenile riparian woodrats, with further research being necessary to support or reject this hypothesis. (USFWS, 2012)

**Stressor:** Competition (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Reproductive success could also be indirectly affected by black rat presence through reduced nourishment caused by competition for food resources, increased energy expenditure in defending stick lodges or other shelter, and reduced access to high quality habitat from competition with black rats (Kelly et al. 2009). (USFWS, 2012)

**Stressor:** Random events (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Both populations of riparian woodrat stand at heightened risk of extinction due to random events. Both populations reside in locations prone to flooding. Riparian woodrats, due to their arboreal nature, are somewhat cushioned from experiencing direct mortality from flood events. Instead, flood events can destroy the stick lodges that are constructed by this species, and can impact the understory that is an important component of riparian woodrat habitat (Service 2000a). Minor flooding occurs approximately every two to three years in the various sloughs and channels in the SJRNWR, while major flooding occurs approximately every five years, during which most or all of the refuge is inundated (Rentner, in litt. 2011). Flooding at CMSP is not well documented, but the park has not flooded in its entirety since before the year 2000 (Karlton, in litt. 2011d). In the winter of 2007 flooding in CMSP was significant enough to prevent travel within the park which limited access throughout the park by CMSP personnel, and CMSP personnel were unable to determine the extent of the flooding (Karlton, in litt. 2011d). The inability of CMSP staff to traverse the park in order to survey the extent of the flooding suggests that this flooding was extensive and a potential threat to the CMSP population of riparian woodrats. (USFWS, 2012)

**Stressor:** Wildfires (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Wildfire, while less common than flooding, has occurred at the SJRNWR. There have been two wildfire events on the SJRNWR since the species was listed. In 2004, the Pelican Fire burned over 1400 acres between July 19 and July 28. Sparked by a car fire on Highway 132, it burned much of the Christman Island area, some of which is known to be occupied by riparian woodrats (Rentner, in litt. 2011). More recently, in 2008, an arson-sparked fire broke out and burned the Lara and Arambel Tracts in the southern portion of the refuge. It is unknown to what extent this fire impacted the riparian woodrat because it is unknown if the woodrat occupies

these areas (Rentner, in litt. 2011). Official records for wildfire events at CMSP are not available, although wildfire events have occurred and may have burned riparian woodrat habitat (Karlton, pers. comm. 2011a, Karlton, in litt. 2011c). Fuel reduction occurred at CMSP in 2001, and was performed by the California Conservation Corps. During this effort a California Conservation Corps worker disturbed a riparian woodrat nest hidden in a downed log and subsequently stepped on a juvenile riparian woodrat (Lee 2001). CMSP staff do not currently manage fuel loads within the park (Karlton, pers. comm. 2011b). Due to the lack of fuel management or recent wildfire activity, it is possible that fuel has accumulated within the park that would present an increased wildfire risk, or be subject to future fuel reduction efforts. Management of fuel loads can negatively impact riparian woodrats if large amounts of dead woody debris are removed as riparian woodrats use downed logs as shelter. (USFWS, 2012)

**Stressor:** Climate change (USFWS, 2012)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Research has shown that the annual mean temperature in North America has increased from 1955 to 2005. However, the magnitude varies spatially across the continent and is most pronounced during spring and winter months, and has affected daily minimum temperatures more than daily maximum temperatures (Field et al. 2007). Other effects of climate change include changes in types of precipitation (i.e., rain vs. snow), earlier spring run-off flow regimes, increased stream temperatures, and more generally, changes in the components of the stream hydrograph. Climate models also predict an increase in precipitation over most of North America except for the southwestern United States (Christensen et al. 2007). In western North America, predicted increases of precipitation have a strong north-south orientation with higher precipitation expected in northern latitudes and lower precipitation in southern latitudes (Christensen et al. 2007). Due to predicted increases in warming, future precipitation events may be more likely to constitute rain than snow, especially during the spring. This may result in a reduced snowpack, earlier snowmelt, decrease spring runoff, and extension of the base flow period in the summer and fall (Hayhoe et al. 2004; Stewart et al. 2005; Knowles et al. 2006, Bates et al. 2008). An increase in rainfall may present a threat to the riparian woodrat in the form of increased flooding frequency, especially late in the rainy season. (USFWS, 2012)

**Stressor:** Small population size (USFWS, 1998)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The only known extant population of riparian woodrat is small, with its size limited by the available habitat. It is thus at an increased risk of extinction because of genetic, demographic, and random catastrophic events (e.g., drought, flooding, fire) that threatens small, isolated populations. Because of its breeding behavior, the effective size of woodrat populations is generally much smaller than the actual population size. This increases the risk of inbreeding depression. (USFWS, 1998)

**Stressor:** Flooding (USFWS, 1998)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The woodrat population at Caswell Memorial State Park is vulnerable to flooding of the Stanislaus River. Because of its well-developed arboreality (ability to climb in trees), the woodrat itself is not as sensitive to flooding as some other brush-dwelling species (e.g., the riparian brush rabbit). However, woodrat houses are essential for survival and these can be severely impacted by flooding, thus affecting population viability. (USFWS, 1998)

### **Recovery**

#### **Reclassification Criteria:**

Reclassification criteria are not available.

#### **Delisting Criteria:**

Delisting criteria are not available.

#### **Recovery Actions:**

- A survey and mapping of all riparian areas along the San Joaquin River. (USFWS, 2012)
- Develop, in collaboration with owners of riparian land and local levee-maintenance districts, an incentive program for preserving riparian vegetation. (USFWS, 2012)
- Develop a plan for the restoration of riparian habitat, the establishment of riparian corridors, and the reintroduction, if necessary, of riparian woodrats to suitable habitat. (USFWS, 2012)
- Initiate a genetic study of the CMSP woodrats, and any other riparian woodrat populations that can be sampled, to determine inbreeding levels; and devise a procedure for ensuring that translocations neither reduce genetic diversity in the parent population nor unduly restrict it in the translocated population. (USFWS, 2012)
- Establish conservation agreements with willing landowners that do not already have conservation easements, as appropriate and necessary, to accomplish habitat restoration, linkage, and reintroduction goals. (USFWS, 2012)
- Begin efforts to restore and link riparian habitat, and reintroduce woodrats as appropriate. (USFWS, 2012)

#### **Conservation Measures and Best Management Practices:**

- A formal recovery plan for this species should be completed. (USFWS, 2012)
- Aggressively pursue the expansion of the SJRNWR along the San Joaquin River, as is currently being examined, with emphasis on plans to connect the refuge with the San Luis NWR Complex. Restored riparian habitat within such an expansion could potentially reintroduce the riparian woodrat to much of its historical range. (USFWS, 2012)
- Undertake or fund a genetic study to clarify the phylogenetic relationships of the woodrat populations at CMSP, the SJRNWR, and localities flanking the known occupied region. (USFWS, 2012)
- Assess the fire fuel load at CMSP to determine the risk posed to riparian woodrats from catastrophic wildfire in the park. Fuel reduction efforts, if necessary, must take into account the relevant habitat requirements of the riparian woodrat, especially the use of downed logs and dead wood as shelter. (USFWS, 2012)
- Perform population surveys of the SJRNWR, SJRNWR easement lands, and if possible, surrounding private property. Concurrently, the refuge should be examined to determine the habitat preferences of the species as the stick lodges typical of the species have not been observed. The

Riparian Mammals Technical Group, and specifically C.S.U. Stanislaus' Endangered Species Recovery Program, should be approached about expanding the emphasis of their efforts to include the riparian woodrat. (USFWS, 2012)

- Manage or eradicate populations of black rats (*Rattus rattus*) at CMSP and SJRNWR to reduce or eliminate competition from this exotic species. Management or eradication efforts should primarily be done using live trapping techniques, and must avoid the use of rodenticides. If possible, management or eradication efforts should be coupled with controlled experimental techniques to clarify the competitive relationships between black rats and riparian woodrats. (USFWS, 2012)
- Survey the San Joaquin River and its tributaries within the known historic range of the woodrat, as well as the south Delta region, for suitable habitat for the species. If suitable habitat is located, it should be surveyed for riparian woodrats to locate additional isolated populations of the species. Unoccupied suitable habitat should be assessed for suitability for reintroduction of the species. (USFWS, 2012)

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## **SPECIES ACCOUNT: *Odocoileus virginianus clavium* (Key deer)**

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### ***Species Taxonomic and Listing Information***

**Listing Status:** Endangered; Southeast Region (R4) (USFWS, 2015)

### **Physical Description**

The Key deer is the smallest subspecies of the North American white-tailed deer. Adult males average 80 pounds (lbs), adult females 63 lbs, and fawns weigh about 32 lbs at birth. Height at the shoulder averages 27 inches for adult bucks and 25 inches for adult does (Hardin et al. 1984). The body appears stockier than that of other deer (Klimstra et al. 1978a); the legs are shorter, and the skull is shorter and relatively wider (Klimstra et al. 1991). Pelage varies from deep reddish-brown to grizzled gray, and a distinct black cross or mask is often present between the eyes and across the brow (Klimstra 1992). Antler size and number of points for male Key deer are less than for other whitetails (Folk and Klimstra 1991a). Bucks typically grow spikes until their second year, when they produce forked antlers. They usually attain 8 points by the fourth year. Besides their size, Key deer possess a number of characteristics unique from other white-tailed deer, including high salt-water tolerance (Jacobson 1974), low birth rate, low productivity (Folk and Klimstra 1991b), more solitary nature, and weak family bonds (Hardin 1974). According to Ellsworth et al. (1994), the Key deer population is the most genetically divergent deer population in the southeastern United States

### **Taxonomy**

The Key deer is a member of the Cervidae family of the order Artiodactyla class Mammalia. It was first recognized as a subspecies distinct from the races of *O. v. osceola* and *O. v. virginianus* when Barbour and Allen (1922) described it. The population has been geographically and reproductively isolated in the Lower Keys since the last glacier melted at least 4,000 years ago.

### **Historical Range**

Formerly most of Florida Keys. (NatureServe, 2015)

### **Current Range**

The Key deer's range probably historically extended from Key Vaca to Key West (Klimstra et al. 1978b). Florida Key deer occupy 20 to 25 islands in the Lower Florida Keys within the boundaries of the NKDR, with about 75 percent of the overall population found on Big Pine Key (Lopez et al., 2004a). The NKDR and the Great White Heron NWR encompass much of this range. The principal factor influencing the distribution and movement of Key deer in the Keys is the location and availability of freshwater. Key deer swim easily between keys and use all islands during the wet season, but suitable water is available on only 13 of the 26 islands during the dry season (Folk 1991).

### **Distinct Population Segments Defined**

No. (USFWS, 2015)

### **Critical Habitat Designated**

No;

### **Life History**

**Feeding Narrative**

Adult: The Key deer is capable of exploiting a variety of foods over a range of habitat conditions. Diet varies seasonally with resource availability and changes in nutritional requirements of deer (Klimstra and Dooley 1990; Carlson et al. 1993). Key deer forage on over 160 plant species including red mangrove (*Rhizophora mangle*), blackbead (*Pithecellobium keyense*), acacia (*Acacia pinetorum*), Indian mulberry (*Morinda royoc*), and pencil flower (*Stylosanthes hamata*). Red and black mangroves (*Avicennia germinans*) constitute 24 percent by volume of the diet of the Key deer (Klimstra and Dooley 1990). Key deer require a freshwater source for survival (Folk et al. 1991).

**Reproduction Narrative**

Adult: The social structure of the Key deer varies throughout the year with the reproductive cycle. Bucks associate with females only during the breeding season and will tolerate other males when feeding and bedding only during the non-breeding season. Does may form loose matriarchal groups consisting of an adult female with several generations of her female offspring, but these associations are not stable (Hardin et al. 1976). Key deer produce fewer young per female than any other white-tailed deer population in North America. Fecundity (number of fetuses per female) and productivity (percent of females reproducing) are low, mean age of first breeding is high, and twinning is infrequent, resulting in relatively low reproductive potential. The sex ratio of Key deer favors males at birth, with a 1.75 to 1 fetal ratio and 2 to 1 fawn ratio. However, significantly higher male mortality at maturity serves to balance adult sex ratios more evenly. Annual deer mortality is a function of deer density and population size (Lopez et al. 2003).

**Habitat Narrative**

Adult: Key deer use all habitat types including pine rocklands, hardwood hammocks, buttonwood salt marshes, mangrove wetlands, freshwater wetlands, and disturbed/developed areas (Lopez 2001). The deer use uplands more than wetlands (Lopez et al. 2004b). Key deer use these habitats for foraging, cover, shelter, fawning, and bedding. Pine rocklands hold freshwater year round and are especially important to Key deer survival. About 34 percent of the range is pine rocklands and hardwood hammocks (Lopez et al. 2004c). Over 85 percent of fawning occurs in pine rocklands and hardwood hammocks (Hardin 1974). Five of 26 islands occupied by Key deer have significant pine rocklands. Key deer also use residential and commercial areas extensively where they feed on ornamental plants and grasses and can seek refuge from biting insects. Behavior: Key deer have well-defined patterns of activity and habitat use, and established trails from years of daily use are visible in many areas within Key deer habitat (Klimstra et al. 1974). Roadkill hotspots are evident from the Service's long-term mortality database, further illustrating the habitual movement patterns of Key deer. The social structure of the Key deer varies throughout the year with the reproductive cycle. Bucks associate with females only during the breeding season and will tolerate other males when feeding and bedding only during the non-breeding season. Does may form loose matriarchal groups consisting of an adult female with several generations of her female offspring, but these associations are not stable (Hardin et al. 1976). Home ranges of Key deer are variable (Lopez 2001). On Big Pine Key and No Name Key, average annual home range size (95 percent probability area; ages combined) for males and females was estimated to be 546 acres and 104 acres, respectively, during the period 1998 to 2000. Home range sizes were significantly larger from 1968 to 1972 (males, 959 acre, females 250 acres) (Silvy 1975; Lopez 2001). Males tend to

disperse from their natal (birth) range as fawns or yearlings. Adult males range over larger areas during the breeding season and may shift to an entirely new area (Silvy 1975; Drummond 1989; Lopez 2001). Territorial behavior is limited to a buck's defense of a receptive doe from other bucks, rather than the defense of a specific territory (Klimstra et al. 1974). Aggressive male behaviors (combat) between rutting males are common in Key deer, especially during the fall breeding season or rut. Key deer home ranges have become smaller and tolerance for other deer has increased because of development and feeding (Lopez et al. 2005). Urbanization: Key deer have urbanized over the last 45 years, a trend reported in Folk and Klimstra (1991c). Key deer are regularly fed at several private locations on Big Pine Key, which has resulted in increased tameness (Folk and Klimstra 1991c; Lopez et al. 2005). Peterson et al. (2004) assessed the effects of residential feeding and watering on Key deer behavior on Big Pine Key. Peterson documented deer aggregations around homes that provided food and water, and the deer exhibited increased levels of tameness. Past research has shown that the Key deer on Big Pine Key habituate to human noises, lights, and vehicular traffic (Folk and Klimstra 1991c). Folk and Klimstra (1991c) observed that Key deer "often bedded in open sites within 7 feet of a road and were not disturbed by cars, pedestrians, or cyclists. Loud noises from within 131 feet, such as circular saws, lawn leafblowers, and wood chippers brought little response." Several studies have documented that deer in general quickly habituate to noise and lights. Bashore and Bellis (1982) found that deer quickly became accustomed to noise and lights on Pennsylvania airfields. It has been suggested that less than 10 percent of Key deer on Big Pine Key exhibit "wild," or natural, characteristics (Frank 2005, personal communication). A study conducted by Harveson et al. (2007) concluded that Key deer have adapted to an urban environment.

### ***Dispersal/Migration***

#### **Motility/Mobility**

Adult: High

#### **Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory

### **Dispersal/Migration Narrative**

Adult: Home ranges of Key deer are variable (Lopez 2001). On Big Pine Key and No Name Key, average annual home range size (95 percent probability area; ages combined) for males and females was estimated to be 546 acres and 104 acres, respectively, during the period 1998 to 2000. Home range sizes were significantly larger from 1968 to 1972 (males, 959 acre, females 250 acres) (Silvy 1975; Lopez 2001). Males tend to disperse from their natal (birth) range as fawns or yearlings. Adult males range over larger areas during the breeding season and may shift to an entirely new area (Silvy 1975; Drummond 1989; Lopez 2001). Territorial behavior is limited to a buck's defense of a receptive doe from other bucks, rather than the defense of a specific territory (Klimstra et al. 1974). Aggressive male behaviors (combat) between rutting males are common in Key deer, especially during the fall breeding season or rut. Key deer home ranges have become smaller and tolerance for other deer has increased because of development and feeding (Lopez et al. 2005).

### ***Population Information and Trends***

#### **Population Trends:**

Stable

**Population Size:**

~650 in 2006

**Population Narrative:**

Population size: The Key deer population on Big Pine Key and No Name Key has increased by about 240 percent since 1972 (Harveson et al. 2005). Collectively, 453 to 517 deer occupy Big Pine and No Name Keys; the highest recorded estimate for these two islands (Lopez et al., 2004a). The Key deer population was estimated at 360 to 375 individuals in 1972, the last official survey (Silvy 1975). More recent data note an increase in population, but estimates of density and structure are lacking (Lopez 2001). Based on habitat condition and the presence of density-dependent disease in the population, the Key deer may be at or near ecological carrying capacity on Big Pine and No Name Key (Lopez 2001; Nettles et al. 2002; Lopez et al. 2004a). Harveson et al. (2005) provided estimates of deer abundance in 2000 and 2001 (646 deer) and determined that subpopulations outside of Big Pine and No Name Key remain well below the carrying capacity of the habitat available to them. The total Key deer population was estimated at about 650 in 2006, perhaps at or near historic highs (Lopez, personal communication, 2006). No significant changes were noted in the population levels in succeeding years and the status of the species is considered stable (Service 2010c). Population variability: Key deer produce fewer young per female than any other white-tailed deer population in North America. Fecundity (number of fetuses per female) and productivity (percent of females reproducing) are low, mean age of first breeding is high, and twinning is infrequent, resulting in relatively low reproductive potential. The sex ratio of Key deer favors males at birth, with a 1.75 to 1 fetal ratio and 2 to 1 fawn ratio. However, significantly higher male mortality at maturity serves to balance adult sex ratios more evenly. Annual deer mortality is a function of deer density and population size (Lopez et al. 2003). The principal factor influencing the distribution and movement of Key deer in the Keys is the location and availability of freshwater. Key deer swim easily between keys and use all islands (from Big Pine Key to Sugarloaf Key) during the wet season, but fresh water is available on only 13 of the 26 islands during the dry season (Folk 1991). Key deer are wide ranging and use virtually all available habitats, including developed areas (Lopez 2001). The Key deer population is growing because of protection from hunting, habitat protection, and the positive response of the population to decreased levels of urban development in Big Pine and No Name Keys (Lopez et al. 2004a). The protection afforded the Key deer through prohibitions on hunting, habitat management, and habitat protection through acquisition has resulted in an increase in (240 percent) in the Big Pine Key deer population. Despite the apparent increase in population levels of Key deer, there has been a contraction of the range of Key deer from 1970 to 1999 (Lopez 2001). Key deer have become increasingly abundant on Big Pine Key and adjacent islands, but have decreased to near extirpation on more distant islands such as Cudjoe and Sugarloaf Keys (Lopez 2001). Although Key deer were never abundant on Cudjoe and Sugarloaf Keys, they previously existed at such low numbers that local extirpation was thought to be likely in the near future (Lopez 2001). This contraction in the range has decreased the overall viability of the Key deer population by increasing the probability that a stochastic event, such as a hurricane or disease epidemic, may have had catastrophic impacts to the core population on and around Big Pine Key (Lopez et al. 2004). Recent relocation efforts, however, and the overall population increase have helped address this concern. The population now is at or near historical highs (Service 2007). As part of its recovery strategy, the Service relocated 39 Key deer to two islands within their existing range from 2003 to 2005. The Service moved 24

individuals (14 does, 10 bucks) from Big Pine Key to Sugarloaf Key and 15 individuals (9 does and 6 bucks) from Big Pine Key to Cudjoe Key. Sugarloaf Key and Cudjoe Key have supported a small number of Key deer in the past. Both islands were home to about five resident deer each. A survey of resident deer on Cudjoe Key prior to relocation produced two deer observed and the relocations appear to be a success due to high survival, low dispersal, and evidence of reproduction (lactation, fawn present, etc.) in translocated females (Parker et al. 2008).

### ***Threats and Stressors***

**Stressor:** Habitat loss

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Loss of habitat resulting from development is the most significant and obvious threat to Key deer (Klimstra et al., 1974). The human population on Big Pine Key more than doubled from 1980 to 2000. An estimated 116 acres per year of Key deer habitat was cleared on Big Pine Key in the early 1970s. A building moratorium, new County ROGO requirements and Habitat Conservation Plan (HCP) for Big Pine and No Name Keys reduced development in recent years, but habitat loss from development is still a threat.

**Stressor:** Fencing

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Fencing associated with development may cause direct Key deer habitat loss by preventing access to areas used for breeding, feeding, and sheltering. Native habitat that is fenced is no longer available for use by the Key deer and the fencing may block access to other areas. This loss of habitat has reduced the availability of food, water, and shelter as well as fawning areas needed by deer to survive and reproduce. Large networks of fencing have fragmented Key deer habitat and restricted movement, which reduces the availability and value of these areas to Key deer. Although the Monroe County Comprehensive Land Use Plan regulates fencing, many areas important to Key deer continue to be impacted by fences. An additional concern is the injury or death that occurs when deer become entangled when attempting to jump fences.

**Stressor:** Fire suppression

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Fire suppression promotes ecological succession in pine rockland communities, resulting in increased hardwood cover, dense brush, decreased herbaceous cover, reduced light penetration, and a general deterioration of habitat quality for Key deer (Klimstra, 1986; Carlson et al., 1993).

**Stressor:** Exotics

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Exotic vegetation is believed to restrict Key deer and concentrate their movements along established trails. This results in more Key deer crossing roads at fewer access routes or walking along roads, increasing their vulnerability to traffic. Exotic plant species such as Australian pine (*Casuarina equisetifolia*), Brazilian pepper (*Schinus terebinthifolius*), and latherleaf (*Colubrina asiatica*) are invading Key deer habitat, out competing native vegetation, and reducing habitat quality.

**Stressor:** Disease

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** As the population density nears carrying capacity, density dependent disease becomes an increasing problem (Lopez 2001). Service biologists necropsy mortalities and test for infectious diseases. Several diseases are documented, but only haemonchosis (anemia attributable to blood loss from blood-sucking parasites) is believed to have affected population dynamics in recent years (Nettles et al. 2002). Scientists first documented the presence of paratuberculosis or Johne's disease in Key deer in 1996 (Nettles et al. 2002, Quist et al. 2002). Corn et al. (2006) monitored the disease and found that it had remained localized within the Big Pine Key and Newfound Harbor subpopulations. The level of this threat to Key deer is unknown, but could potentially be significant, depending on how infectious the disease is among Key deer and sympatric animals (Quist et al. 2002). However, in the 13 years since its discovery, paratuberculosis in the population has not been significant. Nonetheless, density dependent disease is an issue that warrants continued scrutiny.

**Stressor:** Vehicular mortality

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Residential and commercial development over the past 20 years has increased the number of vehicles and vehicular traffic in the Keys. The main thoroughfare for the Keys U.S. 1 runs through much of the Key deer habitat. This additional traffic has increased the likelihood of Key deer/vehicle collisions. Vehicular mortality is the greatest known source of Key deer deaths. Telemetry data suggests that the majority of deer mortality attributed to road kills occurs on U.S. 1 (Lopez 2001). Although lower speed limits are an attempt to reduce traffic mortality, speeding motorists (Lopez 2001; Frank, personal communication, 2005) may continue to cause deaths in some areas. The Service has kept records on Key deer mortality since the 1960s and more than 73 percent of the cases are due to vehicular mortality (Silvy 1975; Lopez et al. 2003; Service unpublished data, 2009a). From 1996 to 2009, over half the vehicular mortalities have occurred along a 3.5-mile segment of U.S. 1, which bisects the southern end of Big Pine Key. Due to the high occurrences of Key deer-vehicle collisions along this road segment, the Service and biologists from the Florida Department of Transportation (FDOT) have attempted to address this mortality issue on U.S. 1 by installing and monitoring underpasses for deer. Braden et al. (2005) summary report to FDOT noted that the Key deer collisions were reduced by 83 to 92 percent inside the fenced segment and that the US 1 highway improvements have not restricted Key deer movements.

**Stressor:** Climate change

**Exposure:**

**Response:****Consequence:**

**Narrative:** The projected sea level rise may affect Key deer through changes in the underlying gradient between saline groundwater and the overlying freshwater lenses present in the lower keys. Sea level rise may also affect Key deer through changes in frequency and duration of hurricane storm surges, fire, and the availability of freshwater. On Big Pine Key, slash pine forest (rocklands) hold freshwater year round and are especially important to Key deer survival. Hurricane Georges made landfall at Big Pine Key in October 1998, and caused severe damage to the Keys vegetation and salinization of the freshwater wetlands. Ross et al. (2009) noted significant stress to the salt-intolerant slash pine forest (pine rocklands). Under the worstcase scenario, models predict inundation of a majority of the uplands important to Key deer by 2100 (Bergh 2009) and eventual conversion of existing coastal hammock and forest habitat to transitional habitat then to tidal areas dominated by mangroves.

**Recovery****Recovery Actions:**

- The protection afforded the Key deer through prohibitions on hunting, habitat management, and habitat protection through acquisition has resulted in an increase in (240 percent) in the Big Pine Key deer population. Despite the apparent increase in population levels of Key deer, there has been a contraction of the range of Key deer from 1970 to 1999 (Lopez 2001). Key deer have become increasingly abundant on Big Pine Key and adjacent islands, but have decreased to near extirpation on more distant islands such as Cudjoe and Sugarloaf Keys (Lopez 2001). Although Key deer were never abundant on Cudjoe and Sugarloaf Keys, they previously existed at such low numbers that local extirpation was thought to be likely in the near future (Lopez 2001). This contraction in the range has decreased the overall viability of the Key deer population by increasing the probability that a stochastic event, such as a hurricane or disease epidemic, may have had catastrophic impacts to the core population on and around Big Pine Key (Lopez et al. 2004). Recent relocation efforts, however, and the overall population increase have helped address this concern. The population now is at or near historical highs (Service 2007). As part of its recovery strategy, the Service relocated 39 Key deer to two islands within their existing range from 2003 to 2005. The Service moved 24 individuals (14 does, 10 bucks) from Big Pine Key to Sugarloaf Key and 15 individuals (9 does and 6 bucks) from Big Pine Key to Cudjoe Key. Sugarloaf Key and Cudjoe Key have supported a small number of Key deer in the past. Both islands were home to about five resident deer each. A survey of resident deer on Cudjoe Key prior to relocation produced two deer observed and the relocations appear to be a success due to high survival, low dispersal, and evidence of reproduction (lactation, fawn present, etc.) in translocated females (Parker et al. 2008).

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## **SPECIES ACCOUNT: *Odocoileus virginianus leucurus* (Columbian white-tailed deer)**

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### ***Species Taxonomic and Listing Information***

**Commonly-used Acronym:** CWTD

**Listing Status:** Threatened; 03/11/1967; Pacific Region (R1) (USFWS, 2016)

### **Physical Description**

White-tailed deer are generally distinguished from mule or black-tailed deer by their longer tail that is brown rather than black on the dorsal surface, a smaller metatarsal gland, and, in adult males, antlers with prongs arising from a single main beam. The Columbian white-tail is one of the large subspecies with "antlers narrowly spreading and curving steeply upward; upperparts dull in general tone, with grizzled pattern approaching Sayal Brown; top of head grizzled, the individual hairs near Mars Brown with buffy tips; tail varying from Cinnamon Buff to Tawny dorsally, terminating in a small, partially concealed, subterminal patch, and broadly fringed with white above and pure white to tip below; outer and more exposed surfaces of legs to base of hoofs near Sayal Brown." (USFWS, 2016).

### **Taxonomy**

This species is one of 38 recognized subspecies of *virginianus*, a species with a contiguous geographic distribution that extends from southern Canada to South America (USFWS, 1983). The subspecies *leucurus* (Columbian white-tail) may not be subspecifically distinct from subspecies *ochrourus* (Gavin and May 1988). Subspecies *leucurus* has hybridized with *O. hemionus* in southwestern Washington (Gavin and May 1988) (NatureServe, 2015).

### **Historical Range**

Historically, CWTD occupied a range of approximately 23,170 square miles (mi<sup>2</sup>) (60,000 square kilometers (km<sup>2</sup> )) west of the Cascades Mountains; from Grants Pass, Oregon, in the south to The Dalles, Oregon, in the east and along the Cowlitz River to the north (Smith 1985; Figure 3) (USFWS, 2013).

### **Current Range**

The range has been reduced to approximately 93 mi<sup>2</sup> (240 km<sup>2</sup>) for the Columbia River DPS (Smith 1985, p. 247; Figure 4) in limited areas of Clatsop and Columbia counties in Oregon, and Cowlitz, Wahkiakum, and now Clark counties in Washington (USFWS, 2013).

### **Distinct Population Segments Defined**

Yes; Columbia River population

### **Critical Habitat Designated**

No;

### ***Life History***

### **Feeding Narrative**

Adult: White-tailed deer are considered generalist browsers that also graze on grasses and forbs (USFWS, 2013).

### **Reproduction Narrative**

Adult: Rutting activity begins the first week of November; by the end of the month, reproductive behavior by males decreases noticeably, although some deer are capable of breeding as late as March. It is assumed the gestation period approaches that of eastern white-tailed deer (210 days). Available data indicate that nearly all adult females become pregnant and give birth to one or two fawns (USFWS, 1983).

### **Geographic or Habitat Restraints or Barriers**

Adult: Invasive reed canarygrass and areas of flooding that result from landscape modification

### **Environmental Specificity**

Adult: Broad (inferred from USFWS, 2013)

### **Habitat Narrative**

Adult: Prefers wet prairie and lightly wooded bottomlands or "tidelands" along streams and rivers; woodlands are particularly attractive when interspersed with grasslands and pastures; along the Columbia River, Sitka spruce, dogwood, cottonwood, red alder, and willow dominate the vegetation; in inland habitats, along the Umpqua River, the tree community consists of Oregon white oak, madrone, California black oak, and Douglas-fir, with a shrubby ground cover of poison oak and wild rose (Matthews and Moseley 1990) (NatureServe, 2015). Habitat selection by fawns in the Columbia River DPS remains largely undocumented, although observations by Refuge biologists suggest that fawns on the JBH Mainland Unit are most often associated with pastures of tall, dense reed canary grass (*Phalaris arundinacea* L.) and tall fescue (*Festuca arundinacea*), as well as mixed deciduous and Sitka spruce (*Picea sitchensis*) forest (USFWS 1983, Brookshier 2004). Habitat on the Julia Butler Hansen NWR includes Sitka spruce intertidal swamp and scrub-shrub tidal wetland (Hunting and Price islands), cottonwood/willow swamp and scrub-shrub tidal wetlands (Wallace Island and portions of the Westport, Oregon mainland), and a mix of tidal marsh, reed canary grass pasture, old growth nonnative blackberry (*Rubus laciniatus*), cottonwood (*Populus trichocarpa*), and tidal wetland (Crims Island) (USFWS, 2013).

### ***Dispersal/Migration***

#### **Dispersal**

Adult: Based on yearly survey efforts, however, we do know that no new subpopulations have formed without translocations, suggesting dispersal may be limited. (81 FR 71386)

#### **Dispersal/Migration Narrative**

Adult: Not available

### ***Population Information and Trends***

#### **Population Trends:**

Fluctuates with the JBH Mainland Unit subpopulation (USFWS, 2013); almost extirpated in 1900 (USFWS, 1983). the overall population trend for the Columbia River DPS does appear to decline

over time until 2004; however, closer examination revealed that the overall trend was strongly influenced by the decline at the JBHR Mainland Unit in the late 1980s. Although population estimates fluctuated, the population has been steadily increasing over time since 2004. We know that population numbers have been influenced by severe flooding in the late 1990s and early 2000s, and by the new subpopulation at Ridgefield NWR, which has been observed breeding and producing twins following translocations. Thus, we have biological evidence to support the positive population trend occurring since 2004. (81 FR 71386)

**Species Trends:**

Improving (USFWS, 2013)

**Resiliency:**

Very low (inferred from USFWS, 2013; see current range/distribution)

**Representation:**

Low (inferred from USFWS, 2013)

**Redundancy:**

Very low (inferred from USFWS, 2013)

**Number of Populations:**

1 (inferred from USFWS, 2013). 81 FR 71386 reports on the population size of 6 separate units.

**Population Size:**

603 (USFWS, 2013). Population sizes for 2015 were: Puget Island, 228, Tenasillahe Island, 155, Westport/ Wallace Island, 190, JBHR Mainland Unit, 100, Upper Estuary Islands, 36 and Ridgefield NWR, 100 for a total of 966. (81 FR 71386)

**Minimum Viable Population Size:**

50 deer per subpopulation (81 FR 71386)

**Population Narrative:**

A total of 603 deer was estimated in 2011. The Columbia River DPS has experienced population fluctuations and its overall trend has been strongly influenced by large shifts in the abundance of the JBH Mainland Unit subpopulation (Clark et al. 2010; Figure 1A, Table 2). The ultimate genetic isolation between the Douglas County and Columbia River DPS populations has led to a decrease in observed genetic diversity in each population compared to the northeastern Oregon population. The Service recommends that CWTD be reclassified as threatened, because the status has improved, the downlisting criteria have been met, and threats have decreased since listing to a point where no threat puts the DPS at risk of extinction (USFWS, 2013). CWTD were extirpated throughout most of their original range by 1900 (Jewett 1914; Bailey 1936 (USFWS, 1983).

**Threats and Stressors**

**Stressor:** Land conversion (NatureServe, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Primary cause for decline has been conversion of prairie habitat to crops and pastures. Logging has degraded forest habitat in some areas; some habitat periodically lost due to flooding of Columbia River; residential development is the primary threat in Douglas County, Oregon, especially along the North Umpqua River (Matthews and Moseley 1990) (NatureServe, 2015). Urban, suburban, and agricultural areas now limit population expansion, and existing occupied areas support densities of CWTD indicative of moderate to low-quality habitats, particularly lower lying and wetter habitat than the species would typically be associated with (USFWS, 2013).

**Stressor:** Hunting (NatureServe, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Uncontrolled sport and commercial hunting also had an impact (USFWS 1999; NatureServe, 2015). If subpopulations should decline, poaching could have an impact on CWTD numbers and would need to be monitored. Regulations and enforcement are in place to protect the CWTD; however, poaching still occurs and the level of poaching is not a threat that can be completely alleviated (USFWS, 2013).

**Stressor:** Flooding (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Flooding is a threat to CWTD habitat when grazing and fawning grounds become inundated for prolonged periods, and the risk of large flooding events could increase with impacts of climate change. In the past, significant flooding events have caused large-scale CWTD mortality and emigration from the JBH Mainland Unit (USFWS 2007). The JBH Mainland Unit has experienced three storm-related floods since 1996. These flooding events have been associated with a sudden drop in population numbers and a recovery over the following few years. During some historical flooding events, CWTD have left low-lying areas and did not return (particularly in areas which continued to sustain frequent flooding, for example Karlson Island) (USFWS, 2013).

**Stressor:** Invasive species (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The persistence of invasive species, especially reed canary grass, has reduced forage quality over much of the CWTD range but it remains unclear as to how much this change in forage quality is affecting the overall status of CWTD. While CWTD will eat the grass, it is only palatable for about 2 months in spring, and it is not a preferred forage species (USFWS, 2013).

**Stressor:** Disease (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The Revised Recovery Plan lists necrobacillosis (hoof disease) as a primary causal factor in CWTD mortality on the JBH NWR (USFWS 1983). *Fusobacterium necrophorum* is

identified as the etiological agent in most cases of hoof disease, although concomitant bacteria such as *Arcanobacterium pyogenes* may also be at play (Langworth 1977; Chirino-Trejo et al. 2003). Damp soil or inundated pastures increase the risk of hoof disease among CWTD with foot injuries (Langworth 1977). Among 155 carcasses recovered from 1974 to 1977, hoof disease was evident in 31 percent (n=49) of the cases, although hoof disease only attributed directly to 3 percent (n=4) of CWTD mortalities (Gavin et al. 1984). Deer Hair Loss Syndrome (DHLS) was documented in Columbian black-tailed deer (CBTD) in northwest Oregon from 2000 to 2004 (Biederbeck 2004). DHLS results when a deer with an immune system weakened by internal parasites is plagued with ectoparasites, such as deer lice *Damalinia (Cervicola)* spp. The weakened deer suffer increased inflammation and irritation, which result in deer biting, scratching, and licking affected areas and, ultimately, removing hair in those regions. Cases were identified in CWTD only in 2002 and 2003. CWTD captured during translocations in recent years have occasionally exhibited evidence of hair loss. On the JBH NWR, DHLS is most often observed among fawns and yearlings during winter months (USFWS 2010c). Parasite loads were tested in 16 CWTD on the JBH Mainland Unit and Tenasillahe Island in February of 1998 (Creekmore and Glaser 1999). All CWTD tested showed evidence of the stomach worm, *Haemonchus contortus*, in fecal samples. Lung worm (*Parelaphostrongylus* spp.) and trematode eggs, possibly from liver flukes (*Fascioloides* spp.) were also detected. These results are generally not a concern among healthy populations, but for a population under nutritional stress, such as the Columbia River DPS of CWTD with less than optimal forage and habitat quality available, a high parasite load can increase the likelihood of mortality, especially among fawns (Creekmore and Glaser 1999) (USFWS, 2013).

**Stressor:** Predation (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Coyote predation was determined to be the primary cause of fawn mortality, accounting for 69 percent (n=61) of all documented mortalities (USFWS, 2013).

**Stressor:** Hybridization (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Hybridization with CBTD was not considered a significant threat to the Columbia River DPS of CWTD at the time of the development of the Revised Recovery Plan (USFWS 1983). However, later studies raised concerns over the presence of BTB genes in the isolated Columbia River DPS population. Hybridization can affect the genetic viability of the Columbia River DPS and additional research regarding hybridization could give broader insight to the implications and occurrence of this phenomenon, and how it may influence subspecies designation (USFWS, 2013).

**Stressor:** Vehicle collisions (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Collision with vehicles remains a concern, especially with respect to newly translocated CWTD. In 2010, 15 CWTD were translocated to Cottonwood Island, Washington,

from Westport, Oregon. Seven of those translocated CWTD were killed by collisions with vehicles on US Highway 30 in Oregon and on Interstate 5 in Washington (Cowlitz Indian Tribe 2010). JBH NWR personnel recorded four CWTD killed by vehicle collisions in 2010 along Highway 4 and on the JBH Mainland Unit. The threat of deer collisions may increase over time as CWTD are translocated closer to urban areas and agricultural areas see increased housing development, but it is unlikely to ever rise to the level of putting the DPS at risk of extinction (USFWS, 2013).

**Stressor:** Climate change (USFWS, 2013)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Although in the foreseeable future, climate change and rising sea levels will not put the Columbia River DPS at risk of extinction, they could potentially represent a long-term future threat to CWTD occupying low lying habitat that is not adequately protected by well-maintained dikes. Climatic models have predicted significant sea level rise over the next century (Glick et al. 2007). Rising sea levels could degrade or inundate current habitat, forcing CWTD to move out of currently used habitat along the Columbia River into marginal or more developed habitat. A rise in groundwater levels could lower forage quality and allow invasive plants to expand their range into new areas (USFWS, 2013).

### ***Recovery***

#### **Reclassification Criteria:**

1. Maintain a minimum of at least 400 CWTD across the Columbia River DPS (USFWS, 2013).
2. Maintain three viable subpopulations, two of which are located on secure habitat (USFWS, 2013).

#### **Delisting Criteria:**

1. Maintain a minimum of at least 400 CWTD across the Columbia River DPS (USFWS, 2013).
2. Maintain three viable subpopulations, all located on secure habitat (USFWS, 2013).

#### **Recovery Actions:**

- Establish necessary new populations of CWTD on existing habitat (USFWS, 1983).
- Encourage public support for CWTD restoration program (USFWS, 1983).
- Annually assess viability of each extant subpopulation of CWTD (USFWS, 1983).
- Ensure viability of extant populations (USFWS, 1983).

### ***Conservation Measures and Best Management Practices:***

- Conduct a population viability analysis (PVA) of the Columbia River DPS of CWTD to address adequacy of recovery priorities and activities (this recommendation should be conducted as soon as possible as the results will affect other recovery action items for CWTD). Given that such a large proportion of CWTD reside on unprotected habitats, consideration should be given to whether the overall population, minimum secure subpopulations, and distribution of the deer within the subpopulations are still adequate to achieve recovery (USFWS, 2013).

- Identify high quality upland habitat in areas that might support populations of CWTD regardless of land ownership: a) Develop a broad-based GIS map to identify potential suitable habitat over a large part of the Lower Columbia River basin, regardless of land ownership. b) Work closely with ODFW, WDFW, CLT, and the Cowlitz Tribe to identify additional high quality upland habitat within the historic range of CWTD. c) Conduct outreach to landowners/managers to determine the potential for translocation and restoration activities (USFWS, 2013).
- Explore the feasibility of recovery tools that facilitate the relocation of species into higher quality habitat such as: a) Section 10(j) of the Act to establish an experimental population of CWTD onto other Federal, State, Tribal, or private lands within CWTD historical range (consider habitat and land use practices that are similar to Douglas County DPS, as well as habitat that is not subject to rising sea levels and the associated stressors of disease and poor-quality forage) b) Habitat Conservation Planning under section 10(a)(1)(B) of the Act to work with nonfederal partners in establishing conservation objectives and planning that would help protect CWTD c) Discuss a partnership with ODFW and WDFW to facilitate the translocation of CWTD into areas of higher quality upland habitat. d) Due to past high rates of capture-related mortality, review translocation methods with regard to target habitat types, locations, timing, etc., to evaluate effectiveness. Discuss the pros and cons of various methods currently used and, if warranted, revise/develop methodology to enhance translocation methods, including evaluation of variables such as site specificity, timing, changes in technology and methods (e.g., soft release techniques), etc. e) Work with State, Federal, Tribal, and non-governmental entities to overcome barriers to establishing populations in new areas, being sure to address adequate habitat needs as well as potential damage concerns. f) Develop habitat restoration and management guidelines that will benefit CWTD for private, State, Federal, Tribal, and non-governmental landowners (USFWS, 2013).
- Continue habitat restoration and enhancement efforts on currently occupied CWTD habitat as well as on potential future CWTD translocation areas. a) Continue habitat restoration and enhancement efforts on the JBH Mainland Unit, including pasture restoration, tree planting for browse and cover, and invasive species control. b) Increase restoration efforts on the Upper Estuary Islands to promote a sustainable subpopulation of animals there (USFWS, 2013).
- Continue predator control on the JBH and Ridgefield NWRs (USFWS, 2013).
- Monitor translocated CWTD (USFWS, 2013).
- Work with ODFW and WDFW to address potential animal damage issues as CWTD expand their range (USFWS, 2013).
- Explore options to conduct additional translocations of CWTD (especially females) to Ridgefield NWR (USFWS, 2013).
- Conduct a second controlled trial for FLIR using humans on the ground in pre-arranged locations over the three habitat types normally found during surveys. This will help confirm the previous trial and its finding that FLIR undercounts CWTD by an average of 25 percent (USFWS, 2013).
- Explore opportunities for the Service or State, Federal, Tribal, and non-governmental partners to acquire lands or conservation easements in areas where CWTD already exist or in areas adjacent to current CWTD subpopulations (USFWS, 2013).
- Evaluate CWTD body condition on JBH lands: a) Capture, collar, and recapture CWTD repeatedly to assess body fat and pregnancy condition in different habitat types over time and evaluate differences, especially after habitat improvements have been made (e.g., JBH Mainland Unit, Tenasillahe Island, Crims Island, etc.). b) Compare body condition results to Douglas County DPS CWTD conditions. c) Continue documenting diet composition especially as habitat enhancements are implemented. d) Understanding diet composition of CWTD can be useful in understanding forage use and body condition. Given this understanding, habitat manipulations could be

implemented and diet information could be re-collected in time increments to understand changes in body condition. This information could provide input to management decisions regarding habitat and forage type, quality, and quantity (USFWS, 2013).

- Conduct studies at Ridgefield NWR. a) Continue population estimation methods (e.g., FLIR surveys, ground counts) to monitor population trends for the Columbia River DPS. b) Review current population estimation methods, to determine if they are robust enough to adequately assess both true population size and to identify trends in the subpopulations. This includes area that may not have been surveyed before, but which may contain CWTD. The BTDC:DWTD ratio may vary from site-to-site, complicating population estimates (USFWS, 2013).
- Assess the long-term recovery value of working toward either securing the habitat that maintains the Westport/Wallace Island subpopulation, or obtaining a landowner agreement that provides a management commitment to continue predator control. a) How important is it to ensure the current management at Westport continues? b) Should the Service or State, Federal, Tribal, and non-governmental partners invest time and money to do so? (USFWS, 2013).
- Review implications of the lack of genetic distinctness between northeastern Oregon white-tailed deer and Columbia River DPS deer. a) Researchers suggest augmenting the Columbia River DPS gene pool with individuals from the Douglas County DPS and the northeastern Oregon population of *Odocoileus virginianus ochororous*, the latter of which has proven to be genetically similar to, but more diverse than the CWTD. b) Researchers suggest that subspecific designation may not be warranted for CWTD due to the observed genetic similarity between CWTD and *O. v. ochororous*. This potential should be further investigated. c) Gather genetic information of CWTD at different sites. d) Cooperate with ODFW and WDFW to gather additional white-tailed deer genetic samples from southeast Washington and northeast Oregon. e) Consider the efficacy and feasibility of augmenting the Columbia River DPS with deer from the Douglas County population or the northeastern Oregon population (USFWS, 2013).
- Address fawn predation and doe survival. a) Determine whether predator control needs to continue indefinitely at JBH NWR, Ridgefield NWR, Westport, and other sites. b) Determine if predator control needs to occur prior to translocation efforts, or in conjunction with those efforts (USFWS, 2013).
- Determine why sex ratios in some areas are skewed: natural mortality rate of CWTD on JBH – does 20 percent, bucks 40 percent (USFWS, 2013).
- Review the current range of the Columbia River DPS as described in the Revised Recovery Plan and re-evaluate whether additional areas/counties should be included (USFWS, 2013).
- Discuss the status of the Upper Estuary Islands subpopulation and its potential to become a 3rd secure subpopulation. a) Is it possible to include Wallace Island in the Upper Estuary Islands numbers with the requirement that manual genetic interchange would occur over the long-term if necessary? b) Evaluate CWTD movement off of Cottonwood Island following the 2010 and 2013 translocations. Attempt to identify why most CWTD leave the island after translocation. Determine whether or not it is worth continuing to try and establish a stable population on Cottonwood Island (USFWS, 2013).
- Recommendations on future management, research, or recovery actions should be developed to address the potential threats that need evaluation given the discussion in this status review: a) Habitat loss/degradation b) Fawn survival c) Predation pressures d) Climate change/flooding e) Hybridization f) Genetic diversity g) Doe survival (USFWS, 2013).
- Issuing a 4(d) rule in this case will support conservation of the species by providing opportunities for CWTD translocations to new areas previously unavailable to create new subpopulations, encouraging habitat restoration of areas on private lands that may act as dispersal corridors for

CWTD, and promoting coexistence between people and CWTD as the deer population increases. These activities will facilitate conservation partnerships with the agricultural community and private landowners to voluntarily create or restore habitat for new and existing subpopulations of CWTD, and encourage natural expansion of CWTD. Thus, we have determined that this 4(d) rule is necessary and advisable for the conservation and recovery of CWTD.

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81 FR 71386. (2016) Final Rule: Endangered and Threatened Wildlife and Plants

**SPECIES ACCOUNT: *Oryzomys palustris natator* (Rice rat)**

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***Species Taxonomic and Listing Information***

**Commonly-used Acronym:** SRR

**Listing Status:** Endangered; Southeast Region (R4) (USFWS, 2015)

**Physical Description**

SRR are semi-aquatic, omnivorous, generalized rats endemic to the Lower Keys. They appear to differ from *O. p. palustris*, the nearest congener, in having a more silvery pelage, a narrower skull, and more slender nasal bones (Spitzer and Lazell 1978). The distribution and ecology of the SRR are summarized in Service (1999).

**Taxonomy**

The rice rat is geographically separated from the nearest congeners, *O. p. palustris*, by at least 83 kilometers (km) (Humphrey and Barbour 1979, Goodyear 1987). The nearest such population is in Everglades National Park, in the southern end of the mainland. Goodyear (1987) included four Upper and Middle Keys in her survey efforts, and captured no rice rats.

**Historical Range**

See Current

**Current Range**

Restricted to the Lower Keys, from Little Pine Key to Saddlebunch Key, Monroe County, Florida, USA; nine separate keys have or had verified populations as of 1987. In the early 1990s, known to occur on eight keys, believed to be extirpated on one key, and likely extirpated on two other keys (USFWS 1993). (NatureServe, 2015). Critical habitat for the rice rat was designated in 1993 (58 FR 46030). Critical habitat encompasses all lands and waters above mean low tide at Little Pine Key, Water Keys, Big Torch Key, Middle Torch Key, Summerland Key north of U.S. Highway 1, Cudjoe Key north of U.S. Highway 1, Johnston Key, Raccoon Key, and Lower Saddlebunch Keys, south of U.S. Highway 1 but not including lands in T. 67 S., R. 27 E., Section 8 and north 1/5 of Section 17. The principal activity within the critical habitat is the operation of the National Key Deer Refuge, the boundaries of which encompass seven of the nine keys with critical habitat (Service 1993).

**Distinct Population Segments Defined**

No. (USFWS, 2015)

**Critical Habitat Designated**

Yes; 8/31/1993.

**Legal Description**

On August 31, 1993, the Service designated critical habitat for the silver sice rat (*Oryzomys palustris natator* (= *O. argentatus*)) under the Endangered Species Act, as amended ( 58 FR 46030 - 46034).

**Critical Habitat Designation**

The critical habitat designation for *Oryzomys palustris natator* includes areas in Monroe County, Florida.

Monroe County, Florida: Little Pine Key, Water Keys, Big Torch Key, Middle Torch Key, Summerland Key north of U.S. Highway 1, Cudjoe Key north of U.S. Highway 1, Johnston Key, Raccoon Key, and Lower Saddlebunch Keys, south of U.S. Highway 1 but not including lands in T. 67 S., R. 27 E., Section 8 and north 1/5 of Section 17. Included are all lands and waters above mean low tide.

#### **Primary Constituent Elements/Physical or Biological Features**

The Service has determined that physical and biological habitat features (referred to as the primary constituent elements) that support nesting, foraging, cover and dispersal are essential to the conservation of the silver rice rat. Goodyear (1984, 1987) described essential habitat for the silver rice rat as areas containing contiguous mangrove swamps, saltmarsh flats, and buttonwood transition vegetation. These vegetational types, as well as fresh water cattail marshes, contain the primary constituent elements in critical habitat for the silver rice rat. These vegetational types can be generally identified by the presence of the following species:

Mangrove swamp containing red (*Rhizophora mangle*), black (*Avicennia germinans*), and white (*Laguncularia racemosa*) mangroves and buttonwood (*Conocarpus erectus*);

Salt marshes, swales, and adjacent transitional wetlands containing saltwort (*Batis maritima*), perennial glasswort (*Salicornia virginica*), saltgrass (*Distichlis spicata*), sea oxeye (*Borrchia frutescens*), keygrass (*Monanthochloë littoralis*), and coastal dropseed (*Sporobolus virginicus*); and,

Fresh water marshes containing cattails (*Typha domingensis*), sawgrass (*Cladium jamaicense*), and cordgrass (*Spartina* spp.).

#### **Special Management Considerations or Protections**

Within these areas the major constituent elements that are known to require special management considerations or protection are mangrove swamps containing red (*Rhizophora mangle*), black (*Avicennia germinans*), and white (*Laguncularia racemosa*) mangroves, and buttonwood (*Conocarpus erectus*); salt marshes, swales, and adjacent transitional wetlands containing saltwort (*Batis maritima*), perennial glasswort (*Salicornia virginica*), saltgrass (*Distichlis spicata*), sea ox-eye (*Borrchia frutescens*), keygrass (*Monanthochloë littoralis*), and coastal dropseed (*Sporobolus virginicus*); and fresh water marshes containing cattails (*Typha domingensis*), saw-grass (*Cladium jamaicense*), and cordgrass (*Spartina* spp.).

#### ***Life History***

##### **Feeding Narrative**

Adult: Rice rats feed on a variety of plants and animals. Their diet and foraging habitat overlaps extensively with that of syntopic black rats (Goodyear 1992). They utilize several sources of fresh water including surface water associated with freshwater lenses, water droplets on vegetation, and pools of water collected in tree holes (Spitzer 1983, Goodyear 1987). Mitchell (1996) proposed that the certain relatively pristine keys with apparently suitable habitat, may fail to support rice rats due to a lack of freshwater.

**Reproduction Narrative**

Adult: May breed through the year. Litter size probably is 4-6. Gestation lasts probably 3-4 weeks (NatureServe, 2015). Reproduction occurs throughout the year, influenced by a variety of ecological factors (Wolfe 1982). Rice rats construct simple spherical nests, about 15 cm in diameter that are usually built on the ground or slightly elevated in grasses (Spitzer 1983).

**Habitat Narrative**

Adult: Suitable habitat for the SRR includes freshwater marsh, saltwater marsh, and mangrove habitats. SRRs typically use three zones that are delineated by their salinity and topography: (1) low intertidal areas, (2) salt marsh flooded by spring or storm tides, and (3) buttonwood transitional areas that are slightly more elevated and only flooded by storm tides (Goodyear 1987). In general, they use mangrove habitats primarily for foraging, and higher-elevation salt marsh habitat for nesting and foraging (Forys et al. 1996). They also tend to use various vegetation zones during different seasons. During the dry season (March to April, and December to January) they use low marsh more frequently, while during the wet season they use mid- and higher-elevation salt marsh habitats more frequently (Forys et al. 1996). Mitchell (1996) proposed that the certain relatively pristine keys with apparently suitable habitat may fail to support rice rats due to a lack of a combination of the three vegetative communities described above (Goodyear 1987). SRRs may also move from one patch of habitat to another in response to seasonal fluctuations in water levels or food availability (Smith and Vrieze 1979). Saltgrass (*Distichlis spicata*) and seashore dropseed (*Sporobolus virginicus*) are the main materials used in nest construction, though other materials such as buttonwood, mangrove, or saltwort are also used (Spitzer 1983). The type locality for the rice rat was a freshwater marsh on Cudjoe Key (Spitzer and Lazell 1978). However, they were only trapped in the three zones typified above, and not reported from freshwater again (Goodyear 1987, Mitchell 1996, Wolf 1987, Forys et al. 1996), until several were captured in freshwater marshes on Cudjoe and Big Torch Keys during 1994-1996 (Mitchell 1996). These individuals also used adjacent salt marsh.

***Dispersal/Migration*****Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory

***Population Information and Trends*****Population Trends:**

Stable

**Number of Populations:**

13

**Population Size:**

1 - 1000 individuals (NatureServe, 2015)

**Population Narrative:**

There are probably fewer than 1000 individuals in the several known EOs. Only 29 animals were trapped in the Lower Keys in 12,000 trap-nights from 1973-84 (Goodyear 1987). As of 1987,

there were 9 documented EOs, although one may be extirpated. (NatureServe, 2015). Mitchell (1996) calculated rice rat density estimates for nine local sites. She found density estimates to range only up to 0.79 rice rats per ha among sites. Among years, estimates ranged from 0 to 0.79 rice rats per ha at one site. She stated that these values are an order of magnitude lower than for “normal” rats and mice. Forsy et al. (1996) surveyed for 9,960 trap nights on nine keys between 1995 and 1996. Minimum known alive estimates ranged from 0 to 14 among sites and seasons. The extrapolated estimates of the minimum number of rice rats per ha ranged from 0 to 7. Mitchell (1996) proposed that the certain relatively pristine keys with apparently suitable habitat, such as Mallory and Porpoise Keys, are too small to support rice rat populations. Several larger Keys that failed to reveal rice rats lacked the combination of the three vegetative communities described above (Goodyear 1987). It is likely that on certain small and medium sized islands, rice rat establishment is inhibited due to a lack of refugia during extreme conditions such as during high tides, or due to this factor in combination with other factors. Larger islands may provide more diverse microhabitats and habitat components such as travel corridors, or at least a more consistent level of availability and safety.

**Status and Distribution**

The rice rat has been documented to occur on 13 islands throughout the Lower Keys. These include Little Pine, Howe, Water, Middle Torch, Big Torch, Summerland, Raccoon, Johnston, Cudjoe, Upper Sugarloaf, Lower Sugarloaf, and Saddlebunch Keys (Vessey et al. 1976; Goodyear 1984, 1987, 1992; Wolfe 1986, 1987; Forsy et al. 1996; Mitchell 1996). Critical habitat is designated on all of these islands, and most are within or partly within the NKDR or Great White Heron National Wildlife Refuge boundaries. Goodyear (1987) conducted trapping surveys on 17 keys between 1973 and 1984. Effort included 11,952 trap nights (Table 3). In that period, 31 individual rice rats were captured on the nine keys with positive results to date. Wolfe (1987) conducted trapping surveys between 1986 and 1987. He surveyed four keys with U.S. Navy holdings. Effort included 3,663 trap nights (Table 3). In those surveys, five individual rice rats were captured, only on Saddlebunch. Mitchell (1996) conducted trapping surveys on 16 keys between 1994 and 1996. Effort included 10,376 trap nights. There were 60 individual rice rats were captured, and 137 re-captures. The total number of keys with documented rice rats was brought to 12. She also calculated estimates of rice rat density for nine local sites. These included data from the 1980s that had not been reported previously. These estimates ranged from 0.00 to 0.79 rice rats per ha among years and sites. The sample of between year comparisons is too small to indicate temporal trends. Forsy et al. (1996) conducted trapping surveys on nine keys between 1995 and 1996. Effort included 9,960 trap nights (Table 3). In that period, there were 251 total captures of 99 individual rice rats. Overall grid trapping success was 3.0 percent. Humphrey and Barbour (1979) conducted limited surveys between 1978 and 1979. Surveys were conducted at six sites on five keys, Cudjoe, Little Torch, Middle Torch, Sugarloaf, and Big Pine. Effort included 670 trap nights. These authors captured no rice rats, and prematurely reported the taxa to be extinct. These data were also reported in Barbour and Humphrey (1982), in addition to an observation that another party detected an extant population on Raccoon Key. Frank (1994) reported unprecedented trapping success on Raccoon Key during March 1994. Effort included 510 trap nights. In that period, there were 55 captures, with 10.8 percent trapping success. The high abundance of rice rats in this unique case may be related to an artificial source of food on the island. The site had spilled food associated with a monkey colony. Trapping has been conducted on Big Pine and Boca Chica Keys. Despite apparently suitable habitat, however, rice rats were not detected on these islands (Goodyear 1987, Wolfe 1987). They were not trapped on Annette, No Name, Crab, Geiger, Little Torch, Porpoise, Mallory, Lower Snipe, Middle Snipe, or Ramrod Keys, either. However, based on the availability of suitable habitat and proximity to existing populations, the rice rat may also occur

on several other islands in the Lower Keys, including but not limited to Big Pine, No Name, Little Torch, Ramrod, and Boca Chica Keys. They do not occur in the Middle or Upper Keys, presumably due of the lack of suitable habitat within the salinity tolerances of the rice rat (Goodyear 1987). A rangewide survey of abundance and distribution was conducted between 2004 and 2005 (Perry 2006). Results of that study were compared to historical data (1984 – 2000), which included previously unpublished data from Service personnel collected between 1997 and 2000. Some of the trapping grids ( $n = 4$ ) in the 2004 – 2005 study replicated surveys (traps, trapping grid layout, and specific location) that were conducted between 1997 and 2000. Perry (2006) noted that he conducted additional field work in 2006. However, those data were neither fully documented in Perry (2006) nor otherwise conveyed to the Service. On keys that had previously been determined to be occupied by rice rats, trap success (rice rat trap events/trap-night effort) was high in the 2004 – 2005 study, and no extirpations were documented (Perry 2006). Trap success in 2004 – 2005 was higher than in surveys conducted before 1997. Comparing overall annual trap success in later periods (1997 – 2005) to historical data from earlier periods (1984 – 1996), trap success and rice rat abundance appeared to be higher in the later period (Perry 2006). In 1996, the highest abundance of rice rats was estimated (based on Minimum Known Alive [MNA] indices) at 2.3/hectare (ha) (5.7/acre). From 1997 to 2005, the average MNA was 8.7/ha (SE = 4.5,  $n = 4$ ) (21.5/acre). However, methods used in some of the earlier studies were not explicitly documented, so it is also possible that lower trap success and other indications of low abundance in the earlier periods were due to different methodologies in those studies (Perry 2006). Alternatively, abundance was actually lower in the pre-1997 studies (Perry 2006). Perry (2006) concluded that the rice rat “population has remained stable throughout its known range for the past 10 years”. Perry (2006) summarized how previous authors characterized rice rat populations. Previous authors reported that rice rat densities in the Keys were lower than those of marsh rice rats (*Oryzomys palustris*) on the mainland. However, MNA estimates reported in studies of mainland populations averaged 11.1/ha (SE = 2.5,  $n = 4$  studies) (27.4/acre), which was similar to results from studies of rice rats in the Keys (8.7/ha [(SE = 4.5,  $n = 4$ )] (21.5/acre). Additionally, trap success in the Keys studies (7.8 % [SE = 2.9,  $n = 4$ ]) was higher than that reported in the literature for mainland studies (5.6 % [SE = 1.2,  $n = 3$ ]). Although the difference between these values were not tested because of the small sample sizes, Perry (2006) concluded that, contrary to earlier reports, available evidence does not indicate that rice rats occur at lower densities than the mainland subspecies. Perry (2006) also noted that the mainland study that produced the highest density estimate did not include estimates of edge effect. Accordingly, the authors of that study (Smith and Vrieze 1979) reported that their values for marsh rice rat density “may be overestimates and not directly comparable to other population studies on the species.” Earlier studies generally considered rice rats to exhibit lower productivity and higher survivorship, and to be “in a state of overall decline” (Perry 2006). Perry (2006) argued that earlier conclusions about rice rat population dynamics were “based on a paucity of data, collected during sporadic, unsystematic, and often unrepeatable efforts”. Perry (2006) did not assess productivity or survivorship. Perry (2006) concluded that rice rat populations and range appear to be stable (at least over the decade, 1997 – 2006), and that densities are likely similar to those of marsh rice rats on the mainland. His findings indicate that, compared to mainland populations, rice rat abundance as of 1984 – 1996 was not as low as generally perceived (or subsequently increased to mainland-comparable levels), and that the mainland-comparable abundance of rice rats in the Keys was maintained (or had been attained) as of 1997 – 2006. Wang et al. (2005) concluded that a very recent bottleneck had occurred and thus were “consistent with reports of recent population declines”. Wang et al. (2005) conducted an

analysis of genetic diversity in the rice rat based on six microsatellite loci in 18 individuals from Saddlebunch Key. Compared to the Everglades population, allelic richness (the number of alleles) and Nei's index of gene diversity (HE) were significantly lower in the Saddlebunch Key sample, whereas observed heterozygosity (HO) did not differ among areas. Wang et al. (2005) used the program BOTTLENECK to assess for a bottleneck (reduction in effective population size,  $N_e$ ). This program provided a Wilcoxon sign-rank test and a graphical method to assess for a bottleneck. The Wilcoxon test assessed whether a significant number of loci exhibited excess heterozygosity. A population that has experienced a bottleneck is "expected to have excess heterozygosity (HE) relative to that expected under mutation-drift equilibrium . . . because allelic richness is lost at a significantly faster rate than heterozygosity after a population reduction". The graphical approach involved plotting the distribution of alleles according to frequency class. By this method, a bottleneck would be suggested by a relatively uniform distribution of allele frequencies whereas a more skewed distribution of frequency classes (an L-shaped distribution associated with many low frequency alleles relative to high frequency alleles) is considered to be indicative of a bottleneck. Wang et al. (2005) concluded that the (one-tailed) Wilcoxon test and the graphical approach were indicative of a recent bottleneck in the Saddlebunch Key population but not in the mainland population. Compared to heterozygosities expected under mutation-drift equilibrium, observed gene diversity was high in five out of six loci in the Saddlebunch Key population and four out of six loci in the mainland population, and the difference was significant for Saddlebunch Key. However, when allele frequencies were adjusted for null alleles, observed gene diversity was high in five out of six loci in the mainland population and like the Saddlebunch Key population, marsh rice rats exhibited significant heterozygote excess, "suggesting that the Everglades may also have experienced a genetic bottleneck". Allele frequency distributions are affected by sample size. Wang et al. (2005) illustrated an L-shaped distribution in allele frequencies in the Everglades (as opposed to Saddlebunch Key). That L-shaped pattern in the Everglades sample ( $n = 55$  individuals) was obscured when data was plotted after being re-sampled to simulated a sample size of 18 (equal to the Saddlebunch Key sample size). The findings of Wang et al. (2005) suggest that a bottleneck may have occurred in the rice rat, but do not constitute definitive evidence of a recent bottleneck. Lower genetic diversity observed in the rice rat (Crouse 2005, Wang et al. 2005) may also have resulted from founder effects and or subsequent genetic drift, as noted by Crouse (2005). Wang et al. (2005) concluded that their data and various calculations "indicate that a bottleneck occurred within the past 10 to 20 years and is consistent with the population declines that occurred between the 1980s and mid 1990s (Smith and Vrieze 1979; U.S. Fish and Wildlife Service 1999). Populations appear to have decreased by at least half during this time period on four of five censused Keys (including Saddlebunch Key) (Numi Mitchell, unpub. data)". Systematic trapping surveys were not conducted prior to the mid-1980s, so survey evidence does not exist to corroborate whether declines occurred in the years before, before and during, or during the 1980s to mid-1990s. The recent, systematic assessment of abundance and distribution generally indicates that the population has remained stable for at least the last 10 years (Perry 2006). Along with other endangered terrestrial mammals endemic to the Lower Florida Keys, the rice rats faces continued encroachment and habitat fragmentation due to human development (Forys et al. 1996). The species requires large, intact marsh systems for its conservation. A large amount of occupied rice rat habitat has been protected through public acquisition and management, but significant areas also remain in private ownership. Although the wetlands inhabited by the rice rat are generally protected through wetland regulations, the threat of critical habitat loss still exists because permits to destroy wetlands can be obtained with sufficient mitigation.

***Threats and Stressors*****Stressor:****Exposure:****Response:****Consequence:**

**Narrative:** Historically, the main threat to the rice rat has been degradation, fragmentation, and loss of habitat due to urbanization. Rice rats require expanses of high-quality salt marsh and estuarine habitats. Freshwater marsh, saltwater marsh, and mangrove habitats have many beneficial qualities, and Federal, State, and Monroe County permits are required for residential or commercial construction that impact them. The conservation of the rice rat may be adversely affected by construction activities for residential and commercial development, as well as by mangrove trimming. These activities can cause direct mortality of individuals through land clearing and habitat loss. Fragmentation results in isolated patches of habitat too small to support the rice rat. Additional threats on the viability of the rice rat have been difficult to quantify because of the low population densities of this species throughout the Lower Keys.

**Stressor:** Development of residential areas and accumulation of solid waste

**Exposure:****Response:****Consequence:**

**Narrative:** The development of residential areas and accumulation of solid waste encourages the establishment and increases the densities of feral and domestic cats, black rats, raccoons, and exotic fire ants. Cats are predators of rice rats and black rats may be competitors. Feral and domestic cats are abundant throughout the Lower Keys, and forage in the higher elevation salt marsh habitats used by the rice rat. Because rodents are often the most abundant items in a domestic cat's diet (Churcher and Lawton 1989), cat predation is believed to be a major threat to the rice rat. Rice rats and black rats use habitats that overlap, and islands with high densities of black rats have been suggested to support few rice rats. Dietary overlap has been shown to be far higher than that between pairs of native rodent species, and black rats have been implicated in the extinction of rice rat congeners on islands (Goodyear 1992). These data suggest that black rats may out-compete rice rats for food and habitat resources. In areas of suitable habitat, the occurrence of black rats may preclude the continued existence of rice rats. Pesticides used to control black rats may also threaten the rice rat. Raccoons are capable of killing both adult and juvenile rice rats, and their populations are unnaturally high in some areas of the Lower Keys. Exotic fire ants may cause direct mortality of rice rats. Exotic fire ants have been documented to cause declines in populations of small mammals in Texas (Killion and Grant 1993). The ants are attracted to mucous, so newborn rice rats are vulnerable to predation (Forys et al. 1996).

**Stressor:** Small, isolated, and widely distributed populations

**Exposure:****Response:****Consequence:**

**Narrative:** The small, isolated, and widely distributed populations of rice rats are also vulnerable to extinction through random demographic fluctuations, loss of genetic variability caused by small population size, and hurricanes that may affect the entire population. Degradation and loss of habitat due to urbanization remains the main threat to the rice rat. Construction activities

typically result in the direct loss of habitat as well as secondary effects that extend into surrounding habitats. Related secondary effects include habitat fragmentation, resource limitation, vehicular access, and an increase in the densities of roads, raccoons, and exotic plants and animals. Fragmentation interferes with rice rat dispersal, foraging, and nesting.

**Stressor:** Dredge and fill activity, mosquito ditching, and solid waste dumping

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Factors such as dredge and fill activity, mosquito ditching, and solid waste dumping may disrupt natural hydrological processes including tidal flows and water circulation patterns. The alteration of the natural hydrologic regime may facilitate the advancement of exotic plants and animals in rice rat habitat, and otherwise fragment and alter habitat. Water level and habitat changes may cause nesting failure and influence where rice rats can forage, reducing the ability of the habitat to support them.

### **Recovery**

#### **Delisting Criteria:**

The SRR will be considered for delisting when all of the following criteria have been met: 1. SRR populations on at least twelve (12) islands of the Lower Keys exhibit a stable or increasing trend, as evidenced by natural recruitment for multiple generations. (Factor A) 2. The SRR populations are connected to the extent that genetic diversity of the three genetic groups can be naturally maintained without translocations or captive breeding. (Factor A, D, E) 3. Predation and competition from non-native species (e.g., Burmese pythons, black rats, and free-roaming pets) are low enough for SRR to remain viable for the foreseeable future. (Factor C, D) 4. In addition to the above criteria, it can be demonstrated that habitat loss associated with development, lack of natural disturbance, and other factors particularly affecting tidal mangrove and salt marsh habitat, are diminished or reversed such that enough suitable habitat remains for SRR to remain viable for the foreseeable future despite anticipated sea level rise. (Factor A, E) (USFWS 2019).

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## SPECIES ACCOUNT: *Ovis canadensis nelsoni* (Peninsular bighorn sheep)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered, March 18, 1998 (63 FR 13134). Revised: Endangered, Distinct Population Segments, April 14, 2009 (74 FR 17288).

### Physical Description

Peninsular bighorn sheep (*Ovis canadensis nelsoni*) are medium-sized bovids in the order Artiodactyla. They have muscular bodies and thick necks. The color of Peninsular bighorn sheep varies from dark brown in the northern mountains to pale tan in the southern desert. Rams have massive brown horns that curve up and back over their ears, then down, around, and up past their cheeks in a C-shaped curl that can spread to 83 centimeters (cm) (33 inches [in.]). Ewes have short, slender horns that never form more than a half-curl. An adult Peninsular bighorn sheep is 76 to 100 cm (30 to 39 in.) tall at the shoulders, and approximately 152 cm (60 in.) long. Rams are normally larger than ewes, weighing an average of 73 kilograms (kg) (160 pounds [lb.]), while the ewe's weigh an average of 48 kg (105 lb.) (USFWS 2011).

### Taxonomy

In the 1998 final listing rule, Peninsular bighorn sheep were listed as a Distinct Population Segment (DPS) of the species bighorn sheep (*Ovis canadensis*). At the time of listing, at least six subspecies of bighorn sheep were named, including four desert bighorn sheep subspecies (*O. c. cremnobates*, *O. c. mexicana*, *O. c. nelsoni*, and *O. c. weemsi*). *Ovis canadensis cremnobates* is the name that previously had been applied to the Peninsular bighorn sheep. However, because of ongoing questions regarding the distinctiveness of the sub-specific taxa at that time, the Peninsular Ranges population was considered a DPS of the species *O. canadensis* rather than a subspecies or a DPS of a particular subspecies. In the proposed revised critical habitat rule that was published in the Federal Register on October 10, 2007, the U.S. Fish and Wildlife Service formally recognized the listed entity as the Peninsular bighorn sheep, a DPS of desert bighorn sheep (*Ovis canadensis nelsoni*). This is the currently accepted taxonomic placement of these animals. The taxonomic revision does not affect discreteness and significance of Peninsular bighorn sheep as a DPS (USFWS 2011).

### Historical Range

Historically, Peninsular bighorn sheep were found along the Peninsular Ranges from the San Jacinto Mountains in Riverside County, California, south into the Volcan de Tres Virgenes Mountains near Santa Rosalia, Baja California, Mexico (USFWS 2011). At the time of listing, the metapopulation of Peninsular bighorn sheep was known to be distributed among at least eight subpopulations in Riverside, Imperial, and San Diego counties, from the San Jacinto Mountains south to the border of Mexico (USFWS 2011). The Santa Rosa Mountains were thought to have two subpopulations at the time of listing (USFWS 2011). Population estimates at the time of listing (1998) indicated approximately 280 Peninsular bighorn sheep existed in the United States, divided among approximately eight subpopulations or ewe groups (72 FR 57740).

### Current Range

Since listing, an additional subpopulation was identified in the Santa Rosa Mountains. Currently, there are nine recovery regions for the Peninsular bighorn sheep metapopulations. These areas include San Jacinto Mountains; Santa Rosa Mountains—North of Highway 74 (North Santa Rosa Mountains); Santa Rosa Mountains—South of Highway 74 through Martinez Canyon (Central Santa Rosa Mountains); Santa Rosa Mountains—South of Martinez Canyon (South Santa Rosa Mountains); Coyote Canyon; North San Ysidro Mountains, Henderson Canyon to County Road S-22; South San Ysidro Mountains, County Road S-22 to State Highway 78; Vallecito Mountains; and Carrizo Canyon/Tierra Blanca Mountains/Coyote Mountains Area. However, the delineation of subpopulations is not limited to the delineation of recovery regions (USFWS 2011).

**Distinct Population Segments Defined**

Yes, April 14, 2009 (74 FR 17288).

**Critical Habitat Designated**

Yes; 4/14/2009.

**Legal Description**

On April 14, 2009, the U.S. Fish and Wildlife Service (Service), designated revised critical habitat for the Peninsular bighorn sheep, a distinct population segment (DPS) of desert bighorn sheep (*Ovis canadensis nelsoni*) occupying the Peninsular Ranges of Southern California, under the Endangered Species Act of 1973, as amended (Act). In total, approximately 376,938 acres (ac) (152,542 hectares (ha)) fall within the boundaries of the critical habitat designation. This revised designation of critical habitat for Peninsular bighorn sheep reduces the 2001 designation by approximately 467,959 ac (189,377 ha).

**Critical Habitat Designation**

Approximately 376,938 ac (152,542 ha) are designated as critical habitat for Peninsular bighorn sheep in four units.

Unit 1: San Jacinto Mountains Unit 1 consists of approximately 4,597 ac (1,860 ha) in the San Jacinto Mountains, Riverside County. Unit 1 is generally located within an area bounded on the east by the city of Palm Springs, bounded on the north by Windy Point and Snow Canyon, and extends south to the northern Palm Canyon area. Land ownership within the unit includes approximately 3,135 ac (1,269 ha) of BLM land; 1,171 ac (474 ha) of USFS land; and 291 ac (118 ha) of Desert Water Authority (DWA) land (Table 2). Unit 1 begins at a low-elevation of about 450 ft (137 m) on the eastern slope and rises to about 4,600 ft (1,400 m) to the west. It is the northernmost unit of revised critical habitat for Peninsular bighorn sheep. This unit was occupied at the time of listing and is currently occupied. Unit 1 contains the physical and biological features essential to the conservation of Peninsular bighorn sheep including a range of vegetation types (PCE 2), foraging and watering areas including alluvial fans (PCE 4 and 5), and steep rocky terrain with elevations and slopes that provide for sheltering, lambing, mating, movement among and between ewe groups (PCE 1), and predator evasion (PCE 3). The physical and biological features essential to the conservation of Peninsular bighorn sheep in Unit 1 may require special management considerations or protection to ameliorate the threats of urban and industrial development (particularly in lower elevation areas) due to the proximity of this unit to the Palm Springs area, and to decrease the direct and indirect effects of human disturbance to Peninsular bighorn sheep and its habitat. Please see the “Special Management Considerations or Protection” section of this final rule for a detailed discussion of the threats to Peninsular bighorn

sheep habitat and potential management considerations. We excluded approximately 4,323 ac (1,749 ha) of tribal land that meets the definition of critical habitat for Peninsular bighorn sheep from the final revised designation. We believe the designation of critical habitat would adversely impact our working relationship with the Tribe, and that Federal regulation through critical habitat designation would be viewed as an unwarranted and unwanted intrusion into tribal natural resource programs. Furthermore, the approximately 4,323 ac (1,749 ha) of tribal land within critical habitat are currently managed in a manner that provides conservation benefits to Peninsular bighorn sheep through implementation of a Tribal Council-approved management plan currently being implemented (2001 Tribal Conservation Strategy; MBA, 2001). The Tribe is also implementing a number of smaller scale habitat- and activity-specific plans that provide some benefit to Peninsular bighorn sheep: Indian Canyons Master Plan, 2002; Tahquitz Canyon Wetland Conservation Plan, 2000; Trail Plan, 2000; and the draft Tribal Fire Management Plan. Furthermore, the 4,323 ac (1,749 ha) of tribal land are within the plan area of the 2007 draft Tribal HCP (Helix Environmental Planning, 2007) that will incorporate additional conservation measures once finalized. See the “Application of Section 4(b)(2)—Other Relevant Impacts—Conservation Partnerships” section of this final rule for a detailed discussion of the tribal management plans. We also excluded lands within the plan area for the Coachella Valley MSHCP from Unit 1. In both the 2007 proposed revised rule and NOA published in the Federal Register on August 26, 2008, we stated we would consider the possible exclusion of approximately 6,287 ac (2,544 ha) of private land and Coachella Valley MSHCP permittee-owned land from the final critical habitat designation in Unit 1. We are excluding these areas from this final revised designation based on partnerships developed during the development of the Coachella Valley MSHCP that was finalized on October 1, 2008.

**Unit 2A:** North Santa Rosa Mountains Unit 2A consists of approximately 45,100 ac (18,251 ha) in the northern Santa Rosa Mountains, Riverside County. Unit 2A is generally located on the east-facing slopes of the northern Santa Rosa Mountains, and extends from near the City of Rancho Mirage in the north to Martinez Canyon in the south, limited to the east by the communities of the northern Coachella Valley. Land ownership within the unit includes approximately 45,098 ac (18,251 ha) of BLM land and 2 ac (1 ha) of DWA land (Table 2). Unit 2A begins at a low-elevation of about 50 ft (15 m) on the eastern slope and rises to about 4,600 ft (1,400 m) to the west. This unit was occupied at the time of listing and remains occupied. Unit 2A contains the physical and biological features that are essential to the conservation of the Peninsular bighorn sheep including a range of vegetation types (PCE 2), foraging and watering areas including alluvial fans (PCE 4 and 5), and steep to very steep, rocky terrain with elevations and slopes that provide for sheltering, lambing, mating, movement among and between ewe groups (PCE 1), and predator evasion (PCE 3). The physical and biological features essential to the conservation of Peninsular bighorn sheep in Unit 2A may require special management considerations or protection to ameliorate the threats of urban, industrial, and agricultural development, and to decrease the direct and indirect effects of human disturbance to Peninsular bighorn sheep and its habitat, due to the proximity of this unit to the highly developed northern Coachella Valley. In particular, the essential features in this unit may require special management considerations or protection to alleviate threats to Peninsular bighorn sheep and its habitat associated with roadways, such as State Route 74 that cuts through the midsection of this unit and may impede movement between ewe groups. Please see the “Special Management Considerations or Protection” section of this final rule for a detailed discussion of the threats to Peninsular bighorn sheep habitat and potential management considerations. We excluded approximately 467 ac (189 ha) of Agua Caliente Band of Cahuilla Indians tribal lands meeting the definition of critical habitat for

Peninsular bighorn sheep from the final revised designation. As stated above under the description of Unit 1, the designation of critical habitat would likely adversely impact our working relationship with the Tribe, and we believe that Federal regulation through critical habitat designation would be viewed as an unwarranted and unwanted intrusion into tribal natural resource programs. Furthermore, these approximately 467 ac (189 ha) of tribal land within critical habitat are currently managed in a manner that provides conservation benefits to Peninsular bighorn sheep through implementation of a Tribal Council-approved management plan currently being implemented (2001 Tribal Conservation Strategy; MBA, 2001). The 467 ac (189 ha) of tribal land are within the plan area of the 2007 draft Tribal HCP (Helix Environmental Planning, 2007) that will incorporate additional conservation measures once finalized. See the “Application of Section 4(b)(2)—Other Relevant Impacts—Conservation Partnerships” section of this final revised rule for a detailed discussion of the tribal management plans. We also excluded lands within the plan area for the Coachella Valley MSHCP from Unit 2A. In the 2007 proposed revised rule and the NOA published in the Federal Register on August 26, 2008, we stated we would consider the possible exclusion of approximately 32,472 ac (13,141 ha) of private land and Coachella Valley MSHCP permittee-owned land from the final critical habitat designation in Unit 2A. We are excluding these areas from this final revised designation based on partnerships developed during the development of the Coachella Valley MSHCP that was finalized on October 1, 2008 (see the “Application of Section 4(b)(2)—Other Relevant Impacts—Conservation Partnerships” section for a detailed discussion). Unit 2B: South Santa Rosa Mountains South to Vallecito Mountains Unit 2B consists of approximately 248,021 ac (100,371 ha) in the southern Santa Rosa Mountains, Coyote Canyon, San Ysidro Mountains, Pinyon Mountains, and Vallecito Mountains, in Riverside, San Diego, and Imperial Counties. Unit 2B is generally located on the east-facing slopes of the above ranges, loosely bounded on the east by the Coachella Valley floor, and extends from the southern Santa Rosa Mountains in the north to the Fish Creek Mountains in the south. Land ownership within the unit includes approximately 16,266 ac (6,583 ha) of BLM land; 217,206 ac (87,901 ha) of land owned by the State of California (including portions of Anza-Borrego Desert State Park); and 14,549 ac (5,888 ha) of private land (Table 2). Unit 2B begins at a low-elevation of about 150 ft (45 m) on the eastern slope and rises to about 4,600 ft (1,400 m) to the west. This unit was occupied at the time of listing and remains occupied. This unit contains the physical and biological features that are essential to the conservation of Peninsular bighorn sheep including a range of vegetation types (PCE 2), foraging and watering areas including alluvial fans (PCE 4 and 5), and steep to very steep, rocky terrain with elevations and slopes that provide for sheltering, lambing, mating, movement among and between ewe groups (PCE 1), and predator evasion (PCE 3). The physical and biological features essential to the conservation of Peninsular bighorn sheep in Unit 2B may require special management considerations or protection to: (1) Ameliorate threats of urban, industrial, and agricultural development due to the proximity of this unit to the Coachella Valley, especially the lower elevation areas in the northeastern portions of this unit; (2) decrease the direct and indirect effects of human disturbance to Peninsular bighorn sheep and its habitat due to recreational activity, since most of this unit includes lands within AnzaBorrego Desert State Park, which is open to recreational activities; (3) alleviate threats to Peninsular bighorn sheep and its habitat associated with State Route 78, which cuts through the southern portion of this unit and may impede movement between ewe groups; and (4) alleviate threats to Peninsular bighorn sheep and its habitat associated with mining operations at Fish Canyon Quarry and various mining claims in the unit. Please see the “Special Management Considerations or Protection” section of this final rule for a detailed discussion of the threats to Peninsular bighorn sheep habitat and potential management considerations.

Unit 3: Carrizo Canyon Unit 3 consists of approximately 79,220 ac (32,059 ha) in the Carrizo Canyon area of San Diego and Imperial Counties, extending south to the U.S.- Mexico border. Unit 3 is generally located in Carrizo Canyon and the surrounding In-Ko-Pah Mountains, Jacumba Mountains, Coyote Mountains, and Tierra Blanca Mountains; it is loosely bounded on the north, east, and west by the Coachella Valley floor. Land ownership within the unit includes approximately 37,747 ac (15,276 ha) of BLM land; 35,533 ac (14,380 ha) of land owned by the State of California (including portions of Anza-Borrego Desert State Park); 5,426 ac (2,196 ha) of private land; and 514 ac (208 ha) of local park land (Table 2). Unit 3 begins at a low-elevation of about 400 ft (122 m) on the eastern slope and rises to about 4,600 ft (1,400 m) to the west. This unit was occupied at the time of listing and is currently occupied. This unit contains the physical and biological features that are essential to the conservation of Peninsular bighorn sheep including a range of vegetation types (PCE 2), foraging and watering areas including alluvial fans (PCE 4 and 5), and steep to very steep, rocky terrain with elevations and slopes that provide for sheltering, lambing, mating, movement among and between ewe groups (PCE 1), and predator evasion (PCE 3). The physical and biological features essential to the conservation of Peninsular bighorn sheep in Unit 3 may require special management considerations or protection to: (1) Decrease the direct and indirect effects of human disturbance to Peninsular bighorn sheep and its habitat due to recreational activity, since most of this unit includes lands within Anza-Borrego Desert State Park, which is open to recreational activities; (2) alleviate threats to Peninsular bighorn sheep and its habitat associated with Interstate 8, which cuts through the southern portion of this unit and may impede movement between ewe groups; and (3) alleviate threats to Peninsular bighorn sheep and its habitat associated with mining operations at Ocotillo Mineral Material Site and other mining claims that may occur in the unit. Please see the "Special Management Considerations or Protection" section of this final rule for a detailed discussion of the threats to Peninsular bighorn sheep habitat and potential management considerations.

#### **Primary Constituent Elements/Physical or Biological Features**

Critical habitat units are designated for Riverside, San Diego, and Imperial Counties, California. The primary constituent elements of critical habitat for the Peninsular bighorn sheep are:

- (i) Moderate to steep, open slopes (20 to 60 percent) and canyons, with canopy cover of 30 percent or less (below 4,600 ft (1,402 m) elevation in Peninsular Ranges) that provide space for sheltering, predator detection, rearing of young, foraging and watering, mating, and movement within and between ewe groups;
- (ii) Presence of a variety of forage plants, indicated by the presence of shrubs (e.g., *Ambrosia* spp., *Caesalpinia* spp., *Hyptis* spp., *Sphaeralcea* spp., *Simmondsia* spp.), that provide a primary food source year round, grasses (e.g., *Aristida* spp., *Bromus* spp.) and cacti (e.g., *Opuntia* spp.) that provide a source of forage in the fall, and forbs (e.g., *Plantago* spp., *Ditaxis* spp.) that provide a source of forage in the spring;
- (iii) Steep, rugged slopes (60 percent slope or greater) (below 4,600 ft (1,402 m) elevation in Peninsular Ranges) that provide secluded space for lambing and terrain for predator evasion;
- (iv) Alluvial fans, washes, and valley bottoms that provide important foraging areas where nutritious and digestible plants can be more readily found during times of drought and lactation, and that provide and maintain habitat connectivity by serving as travel routes between and

within ewe groups, adjacent mountain ranges, and important resource areas (e.g., foraging areas and escape terrain); and

(v) Intermittent and permanent water sources that are available during extended dry periods and provide relatively nutritious plants and drinking water.

### **Special Management Considerations or Protections**

Critical habitat does not include manmade structures (such as buildings, aqueducts, roads, and other paved areas) and the land on which they are located existing within the legal boundaries on the effective date of this rule.

Special management considerations or protection of the physical and biological features essential to the conservation of the DPS may be needed to alleviate the effects of human activity and disturbance to Peninsular bighorn sheep and ensure that the essential features remain available for use by Peninsular bighorn sheep. Restricting human use of trail systems and natural areas during lambing season, re-routing trails, and establishing exclusionary fencing around urban areas may reduce human effects on Peninsular bighorn sheep behavior.

Degradation and fragmentation of bighorn sheep habitat may occur during the construction phase of power lines and their associated structures. Currently, a large power line (Sunrise Powerlink) is approved for construction through Peninsular bighorn sheep critical habitat. Special management considerations and protection of the physical and biological features essential to the conservation of the DPS will be implemented to alleviate the effects of power line structures and their construction on Peninsular bighorn sheep and their habitat. Future construction of major infrastructure, such as power lines, should be avoided in critical habitat, and if unavoidable, should be constructed to minimize habitat effects and allow continued connectivity among ewe groups.

Mining operations occur within southern portions of Peninsular bighorn sheep habitat in Units 2B and 3. Mining activities and associated facilities negatively impact Peninsular bighorn sheep by causing the loss of vegetation structure required for foraging activities and destroying habitats used for escape, bedding, lambing, or connectivity between ranges (PCE 1, 2, 3, 4, and 5). Disturbance could modify the sheep's behavior or cause bighorn sheep to flee an area. Special management considerations or protection of the physical and biological features essential to the conservation of the DPS may be needed to alleviate the effects of mining operations on Peninsular bighorn sheep habitat. Further mining operations should avoid (to the maximum extent possible) areas identified as meeting the definition of critical habitat for Peninsular bighorn sheep.

### ***Life History***

#### **Feeding Narrative**

Adult: Peninsular bighorn sheep are herbivores and use a wide variety of plant species as their food sources. Their diet is varied among seasons because of variation in forage availability, and at least 34 species of plants have been identified that have been eaten by Peninsular bighorn sheep. Several plant species, including joboba (*Simmondsia chinensis*), brittlebush (*Encelia farinosa*), white ratany (*Krameria canescense*), and bee sage (*Hyptis emoryi*) have been identified as important year-round food sources. During the fall, primary food sources include

grasses such as sixweeks threeawn (*Aristida adscensionis*), red brome (*Bromus madritensis rubens*), and cacti (*Opuntia* spp.). Forbs such as *insularis* (*Plantago ovata*) and common ditaxis (*Ditaxis neomexicana*) are primary food sources in the spring (USFWS 2011). During the reproductive season, due to the varied topography of bighorn sheep habitat, foraging ewes typically are concentrated on specific sites, such as alluvial fans and washes, where more productive soils support greater herbaceous growth and a greater diversity of browse species (USFWS 2011). These are therefore more important sources of higher-quality forage than steeper, rockier soils. Peninsular bighorn sheep also need foraging material, which can sometimes be scarce, in their home ranges (USFWS 2011). Peninsular bighorn ewes have a very demanding energy and protein requirement during late gestation, lambing, and nursing (USFWS 2011). Peninsular bighorn sheep are diurnal, but may be active at any time of the day or night (USFWS 2011). Their daily activity pattern includes alternating feeding and resting/ruminating periods. Forage quality influences activity patterns; when forages are low in digestibility, bighorn sheep must spend more time ruminating and digesting forage (USFWS 2011). Peninsular bighorn sheep digest plant-based material by initially softening it in the animal's first stomach, then regurgitating the digested mass (cud) and chewing it again (a process called ruminating) (USFWS 2011). Sometimes food resources are scarce in their home ranges due to the desert environments (USFWS 2000). In the Peninsular Ranges, bighorn sheep compete for resources with other native ungulates such as mule deer (*Odocoileus hemionus*) whose habitats overlap (primarily at the upper elevations of bighorn habitat), with possible geographical and seasonal differences in the degree of overlap. Peninsular bighorn sheep also face competition from domestic livestock, feral animals (horses) and humans. Cattle grazing can be detrimental to bighorn sheep populations, either through direct competition for forage or water, or through vegetation changes in response to cattle grazing (USFWS 2000).

### **Reproduction Narrative**

Adult: Peninsular bighorn sheep rams and ewes tend to loosely segregate during much of the year, and come together primarily during the mating season (rut). The largest rams are the most successful breeders, but the small rams also breed. Ewes that fail to acquire a minimum level of energy reserves may not conceive, or will produce smaller offspring with a poorer chance of survival. Reproductive success and survival of offspring depends on the mother's body weight, access to resources, quality of home range and age. Peninsular bighorn sheep produce relatively slowly and have one lamb per year (USFWS 2011). Breeding season is August through October; lambs are typically born between February and April, although some lambing may occur as late as August. Some rams are capable of successful breeding as early as 6 months of age (though breeding opportunities are limited by the social pressure of larger rams). Ewes first breed around 2 years of age (and typically until they are 16); however, even yearling ewes have produced lambs. Ewes isolate themselves from other females while bearing their lambs. As parturition (labor) approaches, ewes seek secluded sites with shelter, unobstructed views, and steep terrain for predator evasion. Lambs are weaned by 6 months of age, but ewes reduce their care of lambs when resources are scarce, putting their own nutrition needs over their lambs'. Lamb and yearling age classes experience high mortality rates relative to adult bighorns. After reaching adulthood at 2 years of age, bighorn sheep survival is high until approximately 10 years of age.

### **Geographic or Habitat Restraints or Barriers**

Adult: Human-made barriers have reduced available habitat for Peninsular bighorn sheep, and impeded movement between groups can quickly eliminate genetic diversity through genetic drift (USFWS 2011).

**Spatial Arrangements of the Population**

Adult: Clumped according to resources.

**Environmental Specificity**

Adult: Moderate

**Tolerance Ranges/Thresholds**

Adult: Moderate

**Site Fidelity**

Adult: Moderate

**Habitat Narrative**

Adult: Peninsular bighorn sheep use a wide variety of habitats, including steep slopes, cliffs, rough and rocky topography with sparse vegetation, alluvial fans, washes, and the valley floor. Bighorn sheep have evolved predator-evasion behaviors such as the use of escape terrain, which is generally defined as steep, rugged slopes. Escape terrain is important because bighorn sheep typically do not depend on speed alone to outrun their predators, but use their exceptional climbing abilities to outmaneuver predators on steep, rocky outcrops and talus slopes. Areas of gentle terrain, such as valley floors, are important linkages between adjacent mountainous regions, providing bighorn sheep with temporary access to resources. Gentle terrain (e.g., alluvial fans and washes) also provides nutritious forage during droughts and other challenging periods, such as lactation. Variations in slope and aspect also help bighorn sheep survive in a harsh environment. During hot weather, desert bighorn sheep seek shade under boulders, hanging rocks, and cliffs, or they may move to north-facing slopes where temperatures are moderate. During inclement weather, bighorns may again seek caves, overhangs, or slopes that are protected from strong winds; and on cold winter days they may move to sunny, south-facing slopes. Water is an important resource for Peninsular bighorn sheep, and water sources are most valuable to bighorn sheep if they occur in close proximity to adequate escape terrain. In summer, most Peninsular bighorn sheep can be found within a 3- to 5-km (2- to 3-mi.) radius of water. Human-made barriers have reduced available habitat for Peninsular bighorn sheep, and impeded movement between groups can quickly eliminate genetic diversity through genetic drift (NatureServe 2015; USFWS 2000; USFWS 2011).

***Dispersal/Migration*****Motility/Mobility**

Adult: High

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Seasonal movements; Peninsular bighorn sheep migrate seasonally during the hot season, leaving mountain ranges where no standing water is known to exist (USFWS 2011).

**Dispersal**

Adult: The ewes limit their dispersal and exploratory abilities to those of rams. Rams do not show the same level of year-round philopatry (faithfulness to natal home range) and move more widely than ewes, often among different groups (USFWS 2011).

**Immigration/Emigration**

Adult: Limited

**Dependency on Other Individuals or Species for Dispersal**

Adult: Forage resources.

**Dispersal/Migration Narrative**

Adult: The ewes limit their dispersal and exploratory abilities to those of rams. Rams do not show the same level of year-round philopatry (faithfulness to natal home range) and move more widely than ewes, often among different ewe groups. As rams reach 2 to 4 years of age, they begin to follow older rams away from their natal group. A small amount of genetic exchange among subpopulations via movements by rams can counteract the inbreeding and associated decreases in genetic diversity that might otherwise develop in small isolated populations (USFWS 2011). Bighorn sheep are sensitive to habitat modification because they are relatively poor dispersers, largely learning their ranging patterns from older animals, with ewes then demonstrating extreme philopatry for the remainder of their lives. The movement patterns and habits of ewes are learned by their offspring, who learn about escape terrain, water sources, foraging areas, and lambing habitat. In the Peninsular Ranges, bighorn sheep migrate seasonally during the hot season, leaving mountain ranges where no standing water is known to exist (USFWS 2011). Bighorn sheep have large home ranges that allow animals to move in response to variations in predation pressure and changes in resource availability. Home range sizes averaged 25 square kilometers (km<sup>2</sup>) (9.65 square miles [sq. mi.]) for rams and 20 km<sup>2</sup> (7.72 sq. mi.) for ewes in the San Jacinto Mountains. In the narrow band of available habitat, Peninsular bighorn sheep make use of sparse and sometimes sporadically available resources found in their home ranges. The size of individual or group home ranges depends on the juxtaposition of required resources such as water, forage, escape terrain, or lambing habitat, and therefore varies geographically. Home range size also is affected by forage quantity and quality, season, sex, and age of the animal. Many populations have a smaller home range in summer, presumably due to their limited movement away from permanent water sources at that time of year. During the cooler or wetter months of the year, bighorn sheep often exhibit an expanded range as animals move farther from water sources (USFWS 2011).

**Additional Life History Information**

Adult: The movement patterns and habits of ewes are learned by their offspring, who learn about escape terrain, water sources, foraging areas, and lambing habitat (USFWS 2011).

***Population Information and Trends*****Population Trends:**

Stable to slight increase of less than 25 percent (NatureServe 2015; USFWS 2011).

**Species Trends:**

Stable at San Jacinto Mountains, all other recovery region populations increasing (USFWS 2011).

**Resiliency:**

Moderate

**Representation:**

Low

**Redundancy:**

Moderate

**Number of Populations:**

Eight (USFWS 2011)

**Population Size:**

In 2010, the Peninsular bighorn sheep population size was 981 (USFWS 2011).

**Resistance to Disease:**

Resistance is dependent on the disease, but new nonnative diseases have hurt Peninsular bighorn sheep populations (view the Threats & Stressors for more information).

**Adaptability:**

Low

**Additional Population-level Information:**

Prior to listing, the Peninsular bighorn sheep metapopulation experienced three documented extirpations of individual subpopulations at the following locations: (1) north of Chino Canyon (San Jacinto Mountains); (2) Dead Indian Canyon (North Santa Rosa Mountains); and (3) near the United States-Mexico Border (Carrizo Canyon/Tierra Blanca Mountains/Coyote Mountains Area). This changed the distribution in such a way that the range occupied by Peninsular bighorn sheep extended from south of Chino Canyon in the San Jacinto Mountains to north of Interstate 8 in the Jacumba Mountains. Since 1982, Bighorn Institute in Palm Desert, Riverside County, has maintained a captive breeding herd to conduct research and provide for population augmentation in the San Jacinto and North Santa Rosa mountains, and conduct additional research in the Central Santa Rosa Mountains. Since 1985, 122 captive-reared adult bighorn sheep (63 ewes and 59 rams) have been released into the San Jacinto and North Santa Rosa mountains. Released captive sheep readily assimilated into wild populations, which contributed significantly to the recent population resurgences of these two ewe groups. Breeding by captive-reared bighorn sheep has also been reported in the wild (USFWS 2011). The naturally fragment distribution of Peninsular bighorn sheep ewe groups result in distinct subpopulations that are geographically separated. These subpopulations can be grouped into a metapopulation. The potential for increased inbreeding and genetic drift (random changes in genetic frequencies) accompanies decreasing population sizes, and can lead to decreasing levels of heterozygosity (a measure of genetic diversity), which may have negative demographic effects through inbreeding depression (reduction in fitness due to mating among relatives) and loss of adaptability. There is also growing evidence that the level of heterozygosity affects the disease resistance of a population (USFWS 2011).

**Population Narrative:**

Although bighorn sheep have been observed and documented in the Peninsular Ranges since early explorers such as Juan Bautista de Anza in the 1700s, range-wide population estimates were not made until the 1970s. Published estimates were as high as 971 in 1972 and 1,171 in 1974. Range-wide estimates declined to 570 in 1988, 400 in 1992, and between 327 and 524 in 1993. Starting in 1994, a biennial helicopter census was conducted throughout the Peninsular Ranges, using radio-collared animals to estimate sighting probabilities (USFWS 2011). Prior to listing, the Peninsular bighorn sheep metapopulation experienced three documented extirpations of individual subpopulations at the following locations: (1) north of Chino Canyon (San Jacinto Mountains); (2) Dead Indian Canyon (North Santa Rosa Mountains); and (3) near the United States-Mexico Border (Carrizo Canyon/Tierra Blanca Mountains/Coyote Mountains Area). This changed the distribution in such a way that the range occupied by Peninsular bighorn sheep extended from south of Chino Canyon in the San Jacinto Mountains to north of Interstate 8 in the Jacumba Mountains (USFWS 2011). Since 1982, Bighorn Institute in Palm Desert, Riverside County, has maintained a captive breeding herd to conduct research and provide for population augmentation in the San Jacinto and North Santa Rosa mountains, and conduct additional research in the Central Santa Rosa Mountains. Since 1985, 122 captive-reared adult bighorn sheep (63 ewes and 59 rams) have been released into the San Jacinto and North Santa Rosa mountains. Released captive sheep readily assimilated into wild populations, which contributed significantly to the recent population resurgences of these two ewe groups. Breeding by captive-reared bighorn sheep has also been reported in the wild (USFWS 2011). The most recent count of Peninsular bighorn sheep population estimate was in 2010; the total estimated population size was 981. Since the time of listing, all subpopulations have significantly increased in size, with the exception of the San Jacinto Mountains subpopulation. The naturally fragmented distribution of Peninsular bighorn sheep ewe groups result in eight distinct subpopulations that are geographically separated. These subpopulations can be grouped into a metapopulation, which are networks of interacting subpopulations. Since the time of listing, Peninsular bighorn sheep have attempted to recolonize the suitable historical range. In particular, areas where subpopulations were previously extirpated are beginning to be used again (USFWS 2011). The potential for increased inbreeding and genetic drift (random changes in genetic frequencies) accompanies decreasing population sizes, and can lead to decreasing levels of heterozygosity (a measure of genetic diversity), which may have negative demographic effects through inbreeding depression (reduction in fitness due to mating among relatives) and loss of adaptability. There is also growing evidence that the level of heterozygosity affects the disease resistance of a population. Due to the small population and the diversity of eight subpopulations of this species, Peninsular bighorn sheep have a moderate resiliency and redundancy, and low representation (NatureServe 2015; USFWS 2000; USFWS 2011).

### ***Threats and Stressors***

**Stressor:** Habitat destruction

**Exposure:** Urbanization alters habitat.

**Response:** Reduction in quality habitat and food resources; alteration of movement and migration areas.

**Consequence:** Reduction in population numbers; mortality.

**Narrative:** Peninsular bighorn sheep are typically restricted to habitat at elevations below 1,400 meters (4,600 feet). These low elevation areas are also the most preferable for human development. As a result, encroaching urban development and human-related disturbance have had the dual effects of restricting the remaining animals to a smaller area due to habitat loss and

severing connections between subpopulations. Housing developments and golf courses occur in many of the alluvial fans and washes; this has important implications for bighorn sheep, because these areas are valuable for movement and forage. As human development encroaches into bighorn sheep habitat, resources have been and will continue to be eliminated or reduced in value, and the survival of subpopulations will continue to be threatened. Urbanization in and around the Coachella Valley has altered foraging resources (native plants displaced with nonnative and potentially toxic plants), water resources (altering the hydrology or access to water), and habitat continuity (affecting predator-evasion requirements). The City of Borrego Springs has increased in size and approved several urban developments directly adjacent to Peninsular bighorn sheep habitat in the North San Ysidro and South San Ysidro mountains. As a result, mortality events associated with urbanization near bighorn sheep habitat have increased significantly, and will likely continue. Urban and commercial development (proliferation of residential and commercial development, roads and highways, mining, water projects, and trails and recreational uses) have caused habitat loss, degradation, and fragmentation in four recovery regions (San Jacinto, North Santa Rosa, Central Santa Rosa, and South Santa Rosa mountains); agriculture has used water resources (habitat loss) in at least one recovery region (South Santa Rosa Mountains); mines have caused habitat loss in two recovery regions (Vallecito Mountains and south Carrizo Canyon/Tierra Blanca Mountains/Coyote Mountains Area); roads and highways have caused negative effects associated with the fragmentation of six recovery regions (North Santa Rosa Mountains, Central Santa Rosa Mountains, North San Ysidro Mountains, South San Ysidro Mountains, Vallecito Mountains, and south Carrizo Canyon/Tierra Blanca Mountains/Coyote Mountains Area); trails and recreational uses caused fragmentation and degradation range-wide; and off-highway vehicle use impacted two recovery regions (Central Santa Rosa Mountains and south Carrizo Canyon/Tierra Blanca) (USFWS 2011).

**Stressor:** Alteration of the natural fire regime

**Exposure:** Reduction in habitat quality.

**Response:** Changes in distribution, forage, and predation opportunities.

**Consequence:** Avoidance of certain habitat areas.

**Narrative:** In the Peninsular Ranges, fire is a natural event that can benefit bighorn sheep forage quality by opening up dense stands of chaparral for use during early plant successional stages, while also removing the dense vegetation that predators use preferentially at higher elevations. Human fire suppression activities attempt to prevent wildfire, and may allow vegetation to grow unchecked without its natural control by periodic fires. This may influence the distribution of bighorn sheep by causing them to avoid areas with low visibility. Although not identified as a threat at the time of listing, fire suppression has been a threat to bighorn sheep habitat since its inception as a fire management strategy, because it has steadily increased fuel load and decreased foraging area at high elevations range-wide (USFWS 2011).

**Stressor:** Nonnative plants

**Exposure:** Removal or changes in key resources (water sources), and consumption of toxic plants.

**Response:** Dehydration, predation, and biological changes.

**Consequence:** Mortality and reduction in population numbers.

**Narrative:** The presence of tamarisk or saltcedar (*Tamarix* spp.) is a major threat to Peninsular bighorn sheep because of its rapid reproductive and dispersal rates, which allow it to outcompete native plant species in canyon bottoms and washes. Tamarisk significantly reduces or eliminates the standing water on which bighorn sheep depend, and it grows to thick, often impenetrable stands that block access to water sources and provide cover for

predators. Nonnative Saharan mustard (*Brassica tournefortii*) and Mediterranean grass (*Schismus barbatus*) also alter the habitat by outcompeting native species for limited resources, such as soil moisture. Nonnative plants have become a significant component of the native habitat community at low elevations in all recovery regions. Some species of nonnative ornamental plants (used for decorative purposes in urban developments) have been identified as toxic to Peninsular bighorn sheep, and have caused Peninsular bighorn sheep mortalities (USFWS 2011).

**Stressor:** Disease

**Exposure:** Thought to be associated with domestic livestock and irrigated areas.

**Response:** Endo and ectoparasitism.

**Consequence:** Mortality and reduction in population numbers.

**Narrative:** Researchers and land managers suggest that disease plays an important role in the population dynamics of Peninsular bighorn sheep. Numerous endoparasites and ectoparasites are known to occur in bighorn sheep. A variety of bacterial, fungal, and viral infections have also been isolated or detected from Peninsular bighorn sheep individuals by serologic assay. Such pathogens include bluetongue virus, contagious ecthyma virus, parainfluenza-3 virus, bovine respiratory syncytial virus, *Anaplasma*, *Chlamydia*, *Leptospira*, *Pasteurella*, *Psoroptes*, and *Dermacentor*. Disease manifestation may occur during stressful periods for the population, such as high or low population levels, reproductive activity, low nutrient availability, and climatic extremes. Lambs and older sheep may be most susceptible to disease, and disease has been considered to be responsible for high lamb mortality rates. The consequences of novel exposure to nonnative pathogens can be very serious, because Peninsular bighorn sheep have not evolved resistance to such pathogens. Several viruses discovered in sick Peninsular bighorn sheep lambs were nonnative and thought to have been introduced by domestic livestock. At the time of listing, irrigated lawns, golf courses, and ponded waters in and around the Santa Rosa Mountains were thought to facilitate the exposure and spread of pathogens—such as the biting midge and the strongyle (gastrointestinal) parasite—to Peninsular bighorn sheep. The life cycle of the strongyle parasite cannot be completed in an arid environment. However, high moisture content made available through artificially maintained urban sources (i.e., artificial water sources and irrigated lawns) provides suitable conditions for survival of the parasite through the larval stage. In addition, since the time of listing, a pneumonia outbreak that began in the Peninsular Ranges in the mid-1990s has continued (USFWS 2011).

**Stressor:** Predation

**Exposure:** Unnatural cover, and abundance and distribution of prey species.

**Response:** Increased predator attacks.

**Consequence:** Injury, increased mortality, and reduction in population numbers.

**Narrative:** Bighorn sheep evolved in the presence of predators, and have developed effective physical and behavioral mechanisms for dealing with them. Similar to other desert bighorn populations, sheep in the Peninsular Ranges have likely experienced varying levels of mountain lion predation for thousands of years. However, when other factors such as drought; habitat loss; and fragmentation due to urbanization, diseases, fire suppression, and other factors reduce populations to low levels or alter the abundance and distribution of alternate prey species (such as mule deer), then the influence of predation on population dynamics may increase. The expansion of unnatural environments at the urban interface may have increased the risk of predation in some subpopulations. Encroaching development generally increases the abundance of domestic dogs along the urban-wilderness interface, and these dogs are capable of injuring and killing lambs, ewes, and rams. Furthermore, developed areas provide unnatural cover (such

as hedgerows) and dense patches of tall vegetation, which are suitable hiding places for predators to use when ambushing prey. Since the time of listing, predation coinciding with low population numbers has been a fairly constant threat, especially in the two northernmost recovery regions (USFWS 2011).

**Stressor:** Human interaction

**Exposure:** Increase in human interaction.

**Response:** Changes in behavior.

**Consequence:** Reduction in population numbers, and avoidance of certain habitat areas.

**Narrative:** Peninsular bighorn sheep can experience high levels of human activity within their home ranges. Mortalities and avoidance of habitat have been associated with human disturbance. Peninsular bighorn sheep—especially in the Northern Peninsular Ranges—exhibit differences in behavior, and subpopulations have different experiences with humans. However, Peninsular bighorn sheep evolved to tolerate occasional normal disruptions, such as the presence of a predator. Beyond a certain threshold of human activity, they are overwhelmed. Human disturbance tends to be a problem for only about 6 months out of the year, when temperatures are tolerable for hikers. A number of hiking trails and human access points have been closed since the time of listing and, in some cases, bighorn sheep have returned to these areas now that human access has subsided (USFWS 2011).

**Stressor:** Urban-related mortality

**Exposure:** Roads, plants, cars, and swimming pools.

**Response:** Changes in behavior and migration.

**Consequence:** Mortality, reduction of population numbers, and avoidance of habitat.

**Narrative:** Since listing, mortalities of Peninsular bighorn sheep related to urbanization have continued and increased. It has been recorded that Peninsular bighorn sheep have died from causes related to urbanization, including poisoning from toxic nonnative decorative plants, attacks from domesticated canines, and drowning in swimming pools. In addition, bighorn sheep have been struck by vehicles while attempting to cross highways and roads. Since the expansion of urban development in Peninsular bighorn sheep ranges, the mortality rate related to top vehicle strikes has increased and is now a potential threat to the population. In addition, some bighorn sheep are hesitant to cross roads, reducing their access to resources (USFWS 2011).

**Stressor:** Climate change and drought

**Exposure:** Drier conditions and less water availability.

**Response:** See narrative.

**Consequence:** Reduction in population numbers, and mortality.

**Narrative:** The southwestern United States (including the Colorado Desert, where Peninsular bighorn sheep exist) has been warming and drying during the last 12,000 years, which has altered the distribution of flora and fauna. However, the greatest rate of change has occurred in the last 150 years. Furthermore, the Intergovernmental Panel on Climate Change found that the annual mean warming in North America is likely to exceed the global mean warming in most areas, and the southwest specifically is likely to experience the largest increase in summer warming, along with a likely decrease in annual mean precipitation. Mortality among desert bighorn sheep has been shown to increase with drought. The predicted summer temperature increase and precipitation decrease in the southwestern United States may alter resource distribution and availability for Peninsular bighorn sheep (USFWS 2011).

**Recovery****Reclassification Criteria:**

Peninsular bighorn sheep may be considered for downlisting to threatened status as an interim management goal, when all of the following objective, measurable criteria are met:

As determined by a scientifically credible monitoring plan, at least 25 ewes must be present in each of the following nine regions of the Peninsular Ranges during each of the 6 consecutive years (equivalent to approximately one bighorn sheep generation), without continued population augmentation: San Jacinto Mountains, Santa Rosa Mountains—North of Highway 74, Santa Rosa Mountains—South of Highway 74 through Martinez Canyon, Santa Rosa Mountains—South of Martinez Canyon, Coyote Canyon, North San Ysidro Mountains (Henderson Canyon to County Road S-22), South San Ysidro Mountains (County Road S-22 to State Highway 78), Vallecito Mountains, and Carrizo Canyon/Tierra Blanca Mountains/Coyote Mountains Area (USFWS 2000).

Regulatory mechanisms and land management commitments have been established that provide for long-term protection of Peninsular bighorn sheep and all essential habitat, as described in Section II.D.1 of the Recovery Plan. Given the major threat of fragmentation to species in metapopulation structures, connectivity among all portions of habitat must be established and assured through land management commitments, so that bighorn sheep are able to move freely throughout all habitats. In preparation for delisting, protection by means other than the Endangered Species Act must be assured. Such protection should include alternative mechanisms for regulation by federal, state, and local governments, and land management commitments that would provide the protection needed for continued population stability (USFWS 2000).

**Delisting Criteria:**

Peninsular bighorn sheep may be considered recovered to a status no longer requiring protection under the Endangered Species Act and thereafter removed from the List of Endangered and Threatened Wildlife (50 CFR Part 17) when all of the following criteria are met:

As determined by a scientifically credible monitoring plan, at least 25 adult ewes are present in each of the nine regions of the Peninsular Ranges listed under reclassification criteria, during each of 12 consecutive years (approximately two bighorn sheep generations), including the 6 years under reclassification criteria, without continued population augmentation (USFWS 2000).

The range-wide population must average 750 individuals (adults and yearlings) with a stable or increasing population trend over 12 consecutive years (approximately two generations), as in delisting criterion 1 (USFWS 2000).

Regulatory mechanisms and land management commitments have been established that provide for long-term protection of Peninsular bighorn sheep and all essential habitat, as described in Section II.D.1 of the recovery plan. Furthermore, connection among all portions of habitat must be established and assured through land management commitments, so that bighorn sheep are able to move freely throughout the Peninsular Ranges. Delisting would result in loss of protection under the Endangered Species Act; therefore, continued protection by

other means must be assured. This protection should include alternative mechanisms, land management commitments, or conservation programs that would provide the long-term protection needed for continued population viability (USFWS 2000).

**Recovery Actions:**

- Promote population increase and protect habitat (USFWS 2000).
- Initiate or continue research programs necessary to monitor and guide recovery efforts (USFWS 2000).
- Develop and implement education and public awareness programs (USFWS 2000).
- We recommend changing the recovery priority number for Peninsular bighorn sheep from 3 degrees Celsius (°C) to 9°C. Threats identified at listing continue to impact Peninsular bighorn sheep and its habitat (USFWS 2011).
- Identify migratory routes and establish permanently protected corridors or linkages between all subpopulations, especially between the following locations: a. South San Jacinto and North San Jacinto mountains, b. San Jacinto and North Santa Rosa mountains, c. South San Ysidro and North San Ysidro mountains, and d. Jacumba Mountains and Mexico (USFWS 2011).
- Work with our partners to identify specific “no development” zones, cluster proposed development, and/or trade development rights to minimize general habitat impacts and maximize the functionality of corridor/linkage areas (USFWS 2011).
- Study, monitor, and manage the effects of disease and domesticated livestock on Peninsular bighorn sheep in the United States and Mexico (USFWS 2011).
- Construct wildlife crossing overpasses or underpasses over every major barrier (highways, roads, etc.) to assist movement between subpopulations and reduce vehicle collision mortality (USFWS 2011).
- Research and address the effects of both future renewable energy projects and border activities on the recovery of Peninsular bighorn sheep, and create planning guidance to minimize impacts (USFWS 2011).
- Implement management actions to minimize impacts from recreational activities associated with hiking trails in the northern Peninsular Ranges and illegal off-highway vehicle use where it occurs (USFWS 2011).
- Research and quantify the urban and agriculture water withdrawals from the Peninsular Ranges. Address and minimize the effects of water withdrawals on the habitat and individual Peninsular bighorn sheep (USFWS 2011).
- Work with partners and programs (such as the Partners for Fish and Wildlife Program) to identify recovery-related opportunities, such as the construction of additional fences, near major urban centers—including the cities of La Quinta, Borrego Springs, and possibly Ocotillo (USFWS 2011).

***Conservation Measures and Best Management Practices:***

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***Additional Threshold Information:***

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## SPECIES ACCOUNT: *Ovis canadensis sierrae* (Sierra Nevada bighorn sheep)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; April 20, 1999 (64 FR 19300).

### **Physical Description**

The bighorn sheep (*Ovis canadensis*) is a large mammal (family Bovidae), originally described by Shaw in 1804 (64 FR 19300). The species' coat shows a great deal of color variation, ranging from almost white to fairly dark brown. The belly, rump patch, back of legs, muzzle, and eye patch are all white. Males and females both have permanent horns, but they are much larger in males. In females, the horns are slender and saber-like, never forming more than half a curl. In males, the horns are massive and curl up, back over the ears, then curve down, forward, and up past the cheeks. As the animals age, their horns become rough and scarred, and will vary in color from yellowish-brown to dark brown. In comparison to many other desert bighorn sheep, the horns of the Sierra Nevada bighorn sheep are generally more divergent as they coil out from the base. Adult male sheep stand up to 1 meter (m) (3 feet [ft.]) tall at the shoulder; males weigh up to 99 kilograms (kg) (220 pounds [lbs.]) and females 63 kg (140 lbs.) (USFWS 2015).

### **Taxonomy**

The USFWS listed the Sierra Nevada bighorn sheep as a distinct population segment (DPS), *Ovis canadensis californiana* (64 FR 19300), which was the recognized taxonomic classification at the time of listing (Cowan). However, this classification has come under recent scrutiny. Based on new genetic and morphological data, and a reanalysis of Cowan's original data, USFWS recognized the Sierra Nevada bighorn sheep as a unique subspecies of *O. canadensis*, and modified the nomenclature for this taxon from *Ovis canadensis californiana* to *Ovis canadensis sierrae* (USFWS 2008).

### **Historical Range**

This species occurred on the eastern slope of the Sierra Nevada, and, for at least one subpopulation, a portion of the western slope, from Sonora Pass in Mono County south to Walker Pass in Kern County, a total distance of about 346 kilometers (215 miles) (64 FR 19300).

### **Current Range**

Sierra Nevada bighorn sheep inhabit portions of the Sierra Nevada along the eastern boundary of California in Tuolumne, Mono, Fresno, Inyo, and Tulare counties. Habitat occurs from the eastern base of the range as low as 1,460 m (4,790 ft.) to peaks higher than 4,300 m (14,100 ft.) (73 FR 45534).

### **Distinct Population Segments Defined**

This species was originally listed as a DPS; however, due to changes in taxonomy, the species is now recognized as a subspecies rather than a DPS (73 FR 45534).

### **Critical Habitat Designated**

Yes; 8/5/2008.

**Legal Description**

On August 5, 2008, the U.S. Fish and Wildlife Service (Service), designated critical habitat for the Sierra Nevada bighorn sheep (*Ovis canadensis sierrae*) under the Endangered Species Act of 1973, as amended (Act). In total, approximately 417,577 acres (ac) (168,992 hectares (ha)) fall within the boundaries of the critical habitat designation. The critical habitat is located in Tuolumne, Mono, Fresno, Inyo, and Tulare Counties, California.

**Critical Habitat Designation**

The 12 areas designated as critical habitat are: Mount Warren, Mount Gibbs, Convict Creek, Wheeler Ridge, Taboose Creek, Sawmill Canyon, Mount Baxter, Mount Williamson, Big Arroyo, Mount Langley, Laurel Creek, and Olancho Peak.

Unit 1: Mount Warren Unit 1 consists of approximately 36,012 ac (14,574 ha) in Tuolumne and Mono Counties. Unit 1 is generally located within an area bounded on the east by U.S. Highway 395 (located about 1 mi (1.6 km) away), on the south by SR 120, on the north by Green Creek, and on the west by the ridge connecting Ragged Peak in the south to Camiaca Peak in the north. It is located northwest of the town of Lee Vining. Land ownership within the unit includes approximately 35,279 ac (14,277 ha) of Federal land, 165 ac (67 ha) of Los Angeles Department of Water and Power lands, and 568 ac (230 ha) of other private land. The Federal land is administered by the Humboldt-Toiyabe and Inyo National Forests, Yosemite National Park, and Bureau of Land Management. Unit 1 begins at a low elevation of about 7,500 ft (2,286 m) on the eastern slope and rises to about 12,000 ft (3,658 m) on the west. It encompasses some areas from 12,000 to over 14,000 ft (3,658–4,267 m). It is the northernmost unit designated as critical habitat for the Sierra Nevada bighorn sheep. This unit was occupied at the time of listing (65 FR 20, January 3, 2000; Wehausen 1996, p. 477; Sierra Nevada Bighorn Sheep Interagency Advisory Group 1997, pp. 6–7; Wehausen 1999, pp. 6, 8; 2000, pp. 5–7) and is currently occupied with a minimum population estimate of 26 individuals (Wehausen and Stephenson 2006, p. 7). Unit 1 contains all of the features essential to the conservation of the Sierra Nevada bighorn sheep. It contains steep, rocky terrain which provides for foraging (summer and winter), mating, lambing, predator avoidance, and bedding and also allows for seasonal elevational movements; contains a range of vegetation types (PCE 1 and PCE 2) (Johnson et al. 2005, pp. 4–14, 31–32, 34, 37–38; Service 2007, pp. 3–5); and contains mineral licks (PCE 3) (Chow 1992, p. 52). This unit has good high- and low-elevation winter habitat in the area north of Lee Vining Canyon. Mount Warren has a minimum winter range elevation of about 7,546 ft (2,300 m), while Tioga Crest has this type of habitat at 9,515 ft (2,900 m). In the Lundy Canyon area there is good low-elevation south-facing winter range near 8,038 ft (2,450 m). Dunderberg Peak can provide large areas free of snow in the winter. It does not connect to low-elevation winter range but does connect to summer range in Lundy Canyon; visual winter range condition is mixed to open (Service 2007, pp. 127, 129). The essential features found within Unit 1 may require special management considerations or protection to ameliorate the threats of overgrazing. Additionally, the PCEs within this unit may require special management considerations or protection to reverse the impacts of fire suppression which would provide more open habitat and potentially reduce predation, and to protect against the impacts of recreation (e.g., Virginia Lakes, Lundy Lake, Lee Vining Canyon) and development activities (Sections of State Highway 120 are located in this unit). Furthermore, PCEs within Unit 1 may require special management considerations or protection in the form of avalanche control to protect against catastrophic events.

Unit 2: Mount Gibbs Unit 2 consists of approximately 29,702 ac (12,020 ha) in Tuolumne and Mono Counties. Unit 2 is generally bounded on the north by SR 120 with U.S. Highway 395 located approximately 4 mi (6.4 km) to the east. State Route 158 lies along a portion of the southeastern boundary of this unit. The unit is bounded on the west, in part, by Lyell Canyon. It is immediately south of Unit 1 (Mount Warren) and is located southwest of Lee Vining. Land ownership within the unit includes approximately 29,702 ac (12,020 ha) of Federal land administered by the Inyo National Forest and Yosemite National Park. Unit 2 begins at a low elevation of about 7,500 ft (2,286 m) on the eastern slope and rises to 9,000–12,000 ft (2,743–3,658 m) on the west. It encompasses areas from 12,000 to over 14,000 ft (3,658–4,267 m). Unit 2 was occupied at the time of listing (Wehausen 1996, p. 477; Sierra Nevada Bighorn Sheep Interagency Advisory Group 1997, pp. 6–7; Wehausen 1999, pp. 7–8; 2000, pp. 6–7; 65 FR 20, January 3, 2000) and is currently occupied, with a minimum population estimate of 8 individuals (Wehausen and Stephenson 2006, p. 7). Unit 2 contains all of the features essential to the conservation of the Sierra Nevada bighorn sheep. It contains steep, rocky terrain which provides for foraging (summer and winter), mating, lambing, predator avoidance, and bedding and also allows for seasonal elevational movements; contains a range of vegetation types (PCE 1 and PCE 2) (Johnson et al. 2005, pp. 4–14, 31–32, 34, 37–38; Service 2007, pp. 3–5); and contains mineral licks (PCE 3) (Chow 1992, p. 52). An area between Mount Dana and Mount Wood provides considerable high-elevation habitat that is blown free of snow in the winter and connects to south-facing slopes that decline to lower elevations. Winter habitat occurs at a minimum elevation of 9,105 ft (2,775 m) around Mount Gibbs; 8,859 ft (2,700 m) around Mount Lewis; and 7,546 ft (2,300 m) around Mount Wood. Visual winter range condition is open (Service 2007, p. 127). The south-facing side of Mount Lewis is steep and supports little snow in winter. The slopes above Silver Lake offer low-elevation east-facing winter range to 7,599 ft (2,316 m). This area may provide birthing habitat in spring during some years (Service 2007, p. 129). The essential features found within Unit 2 may require special management considerations or protection to ameliorate the threats of overgrazing. Additionally, PCEs within this unit may require special management considerations or protection to reverse the impacts of fire suppression which would provide more open habitat and potentially reduce predation, and to protect against the impacts of recreation (e.g., Lee Vining Canyon) and development activities (sections of SR 120 are located along the northern boundary of this unit; SR 158 lies along a portion of the southeastern boundary of this unit). Furthermore, PCEs within Unit 2 may require special management considerations or protection in the form of avalanche control to protect against catastrophic events.

Unit 3: Convict Creek Unit 3 consists of approximately 36,514 ac (14,777 ha) in Mono and Fresno Counties. Unit 3 is generally located within an area bounded on the northeast by U.S. Highway 395 (located about 2 mi (3.2 km) away), by Fish Creek and the boundary between Inyo and Sierra National Forests on the west, and by Mono Creek on the south. This unit is located about 3 mi (4.8 km) south of Mammoth Lakes. Land ownership within the unit includes approximately 36,497 ac (14,770 ha) of Federal land and 17 ac (7 ha) of private land. Federal land is administered by the Inyo and Sierra National Forests. Unit 3 begins at a low elevation of about 7,500 ft (2,286 m) and rises to about 10,500–12,000 ft (3,200–3,658 m). The unit encompasses areas from 12,000 to over 14,000 ft (3,658–4,267 m). This unit was not occupied at the time of listing and is not currently occupied, but is essential to the conservation of the Sierra Nevada bighorn sheep. The unit contains steep, rocky terrain which provides for foraging (summer and winter), mating, lambing, predator avoidance, and bedding and also allows for seasonal elevational movements, and a range of vegetation types (PCE 1 and PCE 2) (Johnson et al. 2005,

pp. 4–14, 31–32, 34, 37–38; Service 2007, pp. 3– 5). Mineral licks (PCE 3) may or may not occur in this unit. This unit contains south-facing winter habitat above Convict Lake that descends down to 7,874 ft (2,400 m). This habitat is connected to high-elevation windswept patches on Laurel and Bloody Mountains. McGee Mountain has southfacing winter habitat down to about 8,005 ft (2,440 m) but only a small amount of high-elevation habitat. Nevahbe Ridge has windblown habitat, but it is east-facing and habitat occurs down to 8,530 ft (2,600 m) (Service 2007, pp. 127, 130). Visual winter range condition is open (Service 2007, p. 127). While this unit was not occupied at the time of listing, Sierra Nevada bighorn sheep occupied the area historically (Ober 1931, p. 32; Jones 1950, p. 40; Buechner 1960, p. 69; Barrett 1965, p. 43; Dunaway 1971, p. 19; Wehausen et al. 1987, p. 66; Wehausen 1988a, p. 100). This unit is essential to the conservation of the Sierra Nevada bighorn sheep for increasing the number of herds to reduce the significance of losing any particular herd, increasing population viability, decreasing the degree of fragmentation of the current geographic distribution between this unit and Units 4 (Wheeler Ridge) and 2 (Mount Gibbs), increasing opportunities for genetic exchange between these units, and increasing overall herd numbers to reduce extinction risk from stochastic events. Conservation of this unit is necessary to achieve the long-term viability of this subspecies within its range.

**Unit 4: Wheeler Ridge** Unit 4 consists of approximately 80,966 ac (32,766 ha) in Fresno, Inyo, and Mono Counties. Unit 4 is generally located within an area bounded by U.S. Highway 395 (located about 5–17 mi (8– 27.4 km) to the east; Evolution Creek on the south; Pavilion Dome, Pilot Nob, and Mills Creek on the west; and Mono Creek on the north. This unit is located about 12 mi (19.3 km) west of Bishop. Land ownership within the unit includes approximately 80,568 ac (32,605 ha) of Federal land and 398 ac (161 ha) of private land. Federal land is administered by the Inyo and Sierra National Forests, Kings Canyon National Park, and the Bureau of Land Management. Unit 4 begins at a low elevation of about 5,500 ft (1,676 m) on the eastern slope and rises to about 12,000 ft (3,658 m) on the west. It encompasses numerous areas from 12,000 to over 14,000 ft (3,658–4,267 m). This unit was occupied at the time of listing (Wehausen 1996, p. 477; Sierra Nevada Bighorn Sheep Interagency Advisory Group 1997, pp. 6–7; Wehausen 1999, pp. 5–6, 8; 2000, pp. 3–5, 7; 65 FR 20, January 3, 2000) and is currently occupied with a minimum population estimate of 113 individuals (Wehausen and Stephenson 2006, p. 7). Unit 4 contains features that are essential to the conservation of the Sierra Nevada bighorn sheep. It contains steep, rocky terrain which provides for foraging (summer and winter), mating, lambing, predator avoidance, and bedding and also allows for seasonal elevational movements; contains a range of vegetation types (PCE 1 and PCE 2) (Johnson et al. 2005, pp. 4–14, 31–32, 34, 37–38; Service 2007, pp. 3–5); and contains/provides mineral licks (PCE 3) (Stephenson 2007, p. 1). The area around Wheeler Ridge provides minimum elevation winter habitat at 5,578 ft (1,700 m) and is visually open (Service 2007, p. 127). Mount Tom is located south of Wheeler Ridge and provides an open winter visual condition and winter habitat at a minimum elevation of 6,398 ft (1,950 m) in Elderberry Canyon (Service 2007, p. 127, 129–130). High-elevation winter habitat is extensive on the west side of Mount Tom's north ridge. Narrow ridges on the south side can be snow free. Between Basin Mountain and Mount Humphreys, the plateau remains snow free and is accessible to sheep traveling ridge lines from Mount Tom by Four Gables and along the crest. The essential features found within Unit 4 may require special management considerations or protection to ameliorate the threats of overgrazing. Additionally, PCEs within this unit may require special management considerations or protection to reverse the impacts of fire suppression which would provide more open habitat and potentially reduce predation. Finally, PCEs within Unit 4 may require special management considerations or protection for the threats

due to mining, development, and recreation (e.g., Pine Creek area), and avalanche control may be needed to protect against catastrophic events.

**Unit 5: Taboose Creek** Unit 5 consists of approximately 28,805 ac (11,657 ha) in Inyo and Fresno Counties. Unit 5 is generally located within an area bounded on the north by Big Pine Creek and on the south by Taboose Creek. U.S. Highway 395 is about 8.5 mi (13.7 km) to the east, and Marion and Observation Peaks are located to the west. This unit is located about 5 mi (8 km) southwest of Big Pine. Land ownership within the unit includes approximately 28,805 ac (11,657 ha) of Federal land administered by the Inyo National Forest and Kings Canyon National Park. Unit 5 begins at a low elevation of about 6,000 ft (1,829 m) on the eastern slope and rises to 12,000 to over 14,000 ft (3,658–4,267 m) on the west. This unit was not occupied at the time of listing and is not currently occupied, but the unit is essential to the conservation of the Sierra Nevada bighorn sheep. The unit contains steep, rocky terrain which provides for foraging (summer and winter), mating, lambing, predator avoidance, and bedding and also allows for seasonal elevational movements, and a range of vegetation types (PCE 1 and PCE 2) (Johnson et al. 2005, pp. 4–14, 31–32, 34, 37–38; Service 2007, pp. 3–5). Mineral licks (PCE 3) may or may not occur in this unit. High windblown areas (9,187 ft (2,800 m)) occur on Birch and Kid Mountains that may support bighorn sheep. There appears to be limited low-elevation south- or eastfacing habitat unless animals move south to Red Mountain or Taboose Creeks. Taboose Creek offers patches of high-elevation winter habitat and southfacing, low-elevation habitat where it occurs as low as 6,398 ft (1,950 m). The northeast side of Kid Mountain provides some low habitat near 7,218 ft (2,200 m) (Service 2007, pp. 128, 132). The winter range visual condition is open in these areas (Service 2007, p. 128). While this unit was not occupied at the time of listing, Sierra Nevada bighorn sheep occupied the area historically (Ober 1914, p. 125; Jones 1950, p. 38; Buechner 1960, 69; Dunaway 1971 p. 19; Wehausen et al. 1987 p. 66; Wehausen 1988a, p. 101; Berger 1990, p. 94). This unit is essential to the conservation of the Sierra Nevada bighorn sheep for increasing the number of herds to reduce the significance of losing any particular herd, increasing population viability, decreasing the degree of fragmentation of the current geographic distribution between this unit and Units 6 (Sawmill Canyon) and 4 (Wheeler Ridge), increasing opportunities for genetic exchange between these units, and increasing overall herd numbers to reduce extinction risk from stochastic events. Conservation of this unit is necessary to achieve the long-term viability of this subspecies within its range.

**Unit 6: Sawmill Canyon** Unit 6 consists of about 30,508 ac (12,346 ha) in Fresno and Inyo Counties. Unit 6 is generally located within an area bounded on the east by U.S. Highway 395 (located about 3 mi (4.8 km) away), on the south by Unit 7 (Mount Baxter) and Sawmill Pass and Creek, on the west by Woods Creek and the South Fork of Woods Creek, and on the north by Taboose Creek. Land ownership within the unit includes approximately 30,508 ac (12,346 ha) of Federal land administered by the Inyo National Forest and Kings Canyon National Park. Unit 6 begins at a low elevation of about 4,500 ft (1,372 m) on the eastern slope and rises to about 10,500 to over 14,000 ft (3,200–4,267 m). Unit 6 was occupied at the time of listing (Wehausen 1996, p. 477; Sierra Nevada Bighorn Sheep Interagency Advisory Group 1997, pp. 6–7; Wehausen 1999, pp. 4–5, 8; 2000, pp. 3, 7; 65 FR 20, January 3, 2000) and is currently occupied with a minimum population estimate of 36 individuals (Wehausen and Stephenson 2006, p. 7). Unit 6 has features that are essential to the conservation of the Sierra Nevada bighorn sheep. It contains steep, rocky terrain which provides for foraging (summer and winter), mating, lambing, predator avoidance, and bedding and also allows for seasonal elevational movements, and a range of vegetation types (PCE 1 and PCE 2) (Johnson et al. 2005, pp. 4–14, 31–32, 34, 37–38;

Service 2007, pp. 3–5). It is not known if mineral licks (PCE 3) occur on this unit. Unit 6 provides foraging habitat at the northern boundary near Mount Pinchot (Service 2007, p. 132). In addition, minimum elevations of winter habitat occur in the Goodale Creek area at 6,890 ft (2,100 m) and in the Sawmill Creek area at 4,922 ft (1,500 m); winter visual condition is open (Service 2007, p. 128). The essential features found within Unit 6 may require special management considerations or protection to reverse the impacts of fire suppression which would provide more open habitat and potentially reduce predation. The PCEs in Unit 6 may also require special management considerations or protection for threats due to recreation, and avalanche control may be needed to protect against catastrophic events.

**Unit 7: Mount Baxter** Unit 7 consists of approximately 32,220 ac (13,039 ha) in Fresno and Inyo Counties. Unit 7 is generally located within an area bounded on the east by U.S. Highway 395 (located about 3 mi (4.8 km) away); on the south by Bubbs Creek and Forest Route 13S17 to Independence; on the west by Mount Bago, Gardiner Lakes, and Mount Clarence King; and on the north by Unit 6 (Sawmill Canyon) and Sawmill Pass and Creek. This unit is located about 6 mi (9.7 km) west of Independence. Land ownership within the unit includes approximately 32,198 ac (13,030 ha) of Federal land and 22 ac (9 ha) of private land. Federal land is administered by the Inyo National Forest and Kings Canyon National Park. Unit 7 begins at a low elevation of about 4,500 ft (1,372 m) on the eastern slope and rises to about 10,500 to 12,000 ft (3,200–3,658 m) on the west. It encompasses areas from 12,000 to over 14,000 ft (3,658–4,267 m). Unit 7 was occupied at the time of listing (Wehausen 1996, p. 477; Sierra Nevada Bighorn Sheep Interagency Advisory Group 1997, pp. 6–7; Wehausen 1999, pp. 3–4, 8; 2000, pp. 2–3, 7; 65 FR 20, January 3, 2000) and is currently occupied with a minimum population estimate of 69 individuals (Wehausen and Stephenson 2006, p. 7). Unit 7 contains features that are essential to the conservation of the Sierra Nevada bighorn sheep. It contains steep, rocky terrain which provides for foraging (summer and winter), mating, lambing, predator avoidance, and bedding and also allows for seasonal elevational movements; contains a range of vegetation types (PCE 1 and PCE 2) (Johnson et al. 2005, pp. 4–14, 31–32, 34, 37–38; Service 2007, pp. 3–5); and contains mineral licks (PCE 3) (Jones 1950, p. 63; Hicks and Elder 1979, p. 911). This unit provides foraging habitat along the ridges and in drainages of Mount Baxter. Minimum elevations of winter habitat in the Thibaut-Sand Mountain area occur at 5,003 ft (1,525 m), and in the Onion Valley area at 7,546 ft (2,300 m); winter visual condition is open (Service 2007, p. 128). In addition to containing the features essential to the conservation of the Sierra Nevada bighorn sheep, Unit 7 has additional conservation value as it served as a source population, due to its size and productivity, for reintroductions to the Wheeler Crest area (1979, 1980, 1982, 1986, 1988), Mount Langley (1980 and 1982), and Lee Vining Canyon area (1986, 1988) (Sierra Nevada Bighorn Sheep Interagency Advisory Group 1997, p. 6). Individuals from this population may be used for future translocations within the range. The essential features found within Unit 7 may require special management considerations or protection to reverse the impacts of fire suppression which would provide more open habitat and potentially reduce predation. PCEs within Unit 7 also may require special management considerations or protection for threats due to recreation (e.g., Baxter Pass and Onion Valley), and avalanche control may be needed to protect against catastrophic events.

**Unit 8: Mount Williamson** Unit 8 consists of about 32,560 ac (13,177 ha) in Inyo and Tulare Counties. Unit 8 is generally located within an area bounded on the east by U.S. 395 (located about 9 mi (14.5 km) away); on the south by Tulainyo Lake; on the west by the Kern River (located about 3.5 miles (5.6 km) away); and on the north by Forest Route 13S17 to

Independence (located about 1.5 mi (2.4 km) away). This unit is located southwest of Independence and northwest of Lone Pine. Land ownership within the unit includes approximately 32,560 ac (13,177 ha) of Federal land administered by the Inyo National Forest and Sequoia and Kings Canyon National Parks. Unit 8 begins at a low elevation of about 6,000 ft (1,829 m) on the eastern slope and rises to 12,000 to over 14,000 ft (3,658–4,267 m) on the west. Unit 8 was occupied at the time of listing (Wehausen 1996, p. 477; Sierra Nevada Bighorn Sheep Interagency Advisory Group 1997, pp. 6–7; Wehausen 1999, pp. 2–3, 8; 2000, pp. 1–2, 7; 65 FR 20, January 3, 2000) and is currently occupied with a minimum population estimate of 20 individuals (Wehausen and Stephenson 2006, p. 7). Unit 8 contains features that are essential to the conservation of the Sierra Nevada bighorn sheep. The unit contains steep, rocky terrain which provides for foraging (summer and winter), mating, lambing, predator avoidance, and bedding and also allows for seasonal elevational movements, and a range of vegetation types (PCE 1 and PCE 2) (Johnson et al. 2005, pp. 4–14, 31–32, 34, 37–38; Service 2007, pp. 3–5). It is not known if mineral licks (PCE 3) occur in this unit. The Shepherd Creek-Pinyon Creek area in this unit offers winter habitat at a minimum elevation of 6,808 ft (2,075 m); the George Creek-North Bairs Creek provides this habitat at 6,234 ft (1,900 m) (Service 2007, p. 128). The winter visual condition is mixed (Service 2007, p. 128). The essential features found within Unit 8 may require special management considerations or protection to reverse the impacts of fire suppression which would provide more open habitat and potentially reduce predation. This unit could provide an estimated additional 2.2 sq mi (5.8 sq km) of winter range with a relative probability of equal to or greater than 10 percent use if forests were reduced by burning (Johnson et al. 2005, p. 34). PCEs within Unit 8 may require special management considerations or protection to ameliorate the possible threat of overgrazing due to the proximity of this unit to Federal grazing allotments. Furthermore, PCEs within Unit 8 also may require special management considerations or protection for threats due to recreation (e.g., Whitney Portal and Trailhead), and avalanche control may be needed to protect against catastrophic events.

Unit 9: Big Arroyo Unit 9 consists of approximately 24,987 ac (10,112 ha) in Tulare County. Unit 9 is generally located within an area bounded on the east by the Kern River; on the north by Kern-Kaweah River, Junction Meadow, and Wallace Creek area; and on the west and south by the Big Arroyo Creek. Land ownership within the unit includes approximately 24,987 ac (10,112 ha) of Federal land is administered by Sequoia National Park. Unit 9 begins at a low elevation of about 6,500 ft (1,981 m) on the eastern slope and rises to about 12,000 ft (3,658 m) on the west. The northern boundary encompasses areas from 12,000 to over 14,000 ft (3,658–4,267 m). This unit was not occupied at the time of listing and is not currently occupied, but is essential to the conservation of Sierra Nevada bighorn sheep. The unit contains steep, rocky terrain which provides for foraging (summer and winter), mating, lambing, predator avoidance, and bedding and also allows for seasonal elevational movements, and a range of vegetation types (PCE 1 and PCE 2) (Johnson et al. 2005, pp. 4–14, 31–32, 34, 37–38; Service 2007, pp. 3–5). It is not known if mineral licks (PCE 3) are located within this unit. This unit contains no high-elevation wind-swept areas (Service 2007, p. 134). Winter habitat is provided at a minimum elevation of 6,890 ft (2,100 m) with a mixed visual condition due to scattered trees (Service 2007, pp. 128, 134). From the upper end of the Big Arroyo drainage, sheep could find access to alpine habitat on Kaweah Peaks. While this unit was not occupied at the time of listing, Sierra Nevada bighorn sheep occupied the area historically (Jones 1950, p. 35; Buecher 1960, p. 69; Barrett 1965, p. 43; Riegelhuth 1965, p. 35; Wehausen 1988b, p. 100). This unit is essential to the conservation of the Sierra Nevada bighorn sheep for increasing the number of herds to reduce the significance of losing any particular herd, increasing population viability, decreasing the degree of fragmentation of the

current geographic distribution between this unit and Units 8 (Mount Williamson), and 10 (Mount Langley), increasing opportunities for genetic exchange between these units, and increasing overall herd numbers to reduce extinction risk from stochastic events. Conservation of this unit is necessary to achieve the long-term viability of this subspecies within its range.

**Unit 10: Mount Langley** Unit 10 consists of approximately 32,845 ac (13,292 ha) in Inyo and Tulare Counties. Unit 10 is generally located within an area bounded on the east by Forest Route 16S02 located from immediately adjacent to the unit to 7 mi (11.3 km) away, on the south by Muah Mountain, on the west by Cirque Peak and the Perrin Creek area, and on the north by Lone Pine Creek. This unit is located about 7 mi (11.3 km) southwest of Lone Pine. Land ownership within the unit includes approximately 32,845 ac (13,292 ha) of Federal land administered by the Inyo National Forest, Sequoia National Park, and Bureau of Land Management. Unit 10 begins at a low elevation of about 4,500 ft (1,372 m) on the eastern slope and rises to 9,000 to 12,000 ft (2,743–3,658 m) on the west side. It encompasses areas between 12,000 and 14,000 ft (3,658–4,267 m). Unit 10 was occupied at the time of listing (Wehausen 1996, p. 477; Sierra Nevada Bighorn Sheep Interagency Advisory Group 1997, pp. 6–7; Wehausen 1999, pp. 1–2, 8; 2000, pp. 1, 7; 65 FR 20, January 3, 2000) and is currently occupied with a minimum population estimate of 90 individuals (Wehausen and Stephenson 2006, p. 7). Unit 10 contains features that are essential to the conservation of the Sierra Nevada bighorn sheep. The unit contains steep, rocky terrain which provides for foraging (summer and winter), mating, lambing, predator avoidance, and bedding and also allows for seasonal elevational movements, and a range of vegetation types (PCE 1 and PCE 2) (Johnson et al. 2005, pp. 4–14, 31–32, 34, 37–38; Service 2007, pp. 3–5). It is not known if mineral licks (PCE 3) occur in this unit. The unit provides low elevation (5,742 ft, 1,750 m) mixed winter range in the Carroll Creek-Turtle Creek area. It also provides low elevation (4,757 ft, 1,450 m), open winter range in the Slide Canyon-Cottonwood Creek area (Service 2007, pp. 128, 133). From this area, it is possible that bighorn sheep could cross a short distance of the open south-facing forest by Wonoga Peak to access the large open plateau country. It is also possible that bighorn sheep using the Cottonwood Creek area use summer range to the southeast of the Kern Plateau where elevations are about 10,000 ft (3,048 m) (Service 2007, p. 130). The essential features found within Unit 10 may require special management considerations or protection to reverse the impacts of fire suppression which would provide more open habitat and potentially reduce predation. This unit could provide an estimated additional 1.8 sq mi (4.7 sq km) of winter range with a relative probability of equal to or greater than 10 percent use if forests were reduced by burning (Johnson et al. 2005, p. 34). PCEs within Unit 10 may require special management considerations or protection to ameliorate the possible threat of overgrazing due to the proximity of this unit to Federal grazing allotments. PCEs within Unit 10 may also require special management considerations or protection for threats due to recreation (e.g., Whitney Portal and Trailhead) and development (Forest Route 16S02 crosses a portion of this unit). Furthermore, PCEs within Unit 10 may require special management considerations or protection in the form of avalanche control to protect against catastrophic events.

**Unit 11: Laurel Creek** Unit 11 consists of approximately 22,037 ac (8,918 ha) in Tulare County. Unit 11 is generally located within an area bounded on the east by the Kern River; on the south by Pistol, Laurel, and Golden Trout Creeks; on the west by a portion of Little Kern River; and on the north by Soda Creek. Land ownership within the unit includes approximately 22,037 ac (8,918 ha) of Federal land administered by the Sequoia National Forest and Sequoia National Park. Unit 11 begins at a low elevation of about 6,500 ft (1,981 m) on the eastern slope and rises to 10,500

to 12,000 ft (3,200–3,658 m) on the west. It includes a few small areas from 12,000 to over 14,000 ft (3,658–4,267 m). This unit was not occupied at the time of listing and is not currently occupied, but the unit is essential to the conservation of Sierra Nevada bighorn sheep. The unit contains steep, rocky terrain which provides for foraging (summer and winter), mating, lambing, predator avoidance, and bedding and also allows for seasonal elevational movements, and a range of vegetation types (PCE 1 and PCE 2) (Johnson et al. 2005, pp. 4–14, 31–32, 34, 37–38; Service 2007, pp. 3–5). It is unknown whether mineral licks (PCE 3) occur in this unit. This unit contains no high-elevation wind-swept areas (Service 2007, p. 134). Winter habitat is provided at a minimum elevation of 6,808 ft (2,075 m) with a mixed visual condition due to scattered trees (Service 2007, pp. 128, 134). Laurel Creek provides access to summer range. While this unit was not occupied at the time of listing, Sierra Nevada bighorn sheep occupied the area historically (Buechner 1960 p. 69; Barrett 1965, p. 43; Wehausen 1988b, p. 100). This unit is essential to the conservation of the Sierra Nevada bighorn sheep for increasing the number of herds to reduce the significance of losing any particular herd, increasing population viability, decreasing the degree of fragmentation of the current geographic distribution between this unit and Unit 10 (Mount Langley), increasing opportunities for genetic exchange between these units, and increasing overall herd numbers to reduce extinction risk from stochastic events. Conservation of this unit is necessary to achieve the long-term viability of this subspecies within its range.

Unit 12: Olancho Peak Unit 12 consists of approximately 30,421 ac (12,311 ha) in Tulare and Inyo Counties. Unit 12 is generally located within an area bounded on the east by U.S. Highway 395, on the south by Falls and Walker Creeks, on the west by a portion of the Pacific Crest National Scenic Trail, and on the north by Muah Mountain. This unit is located west of the towns of Cartago and Olancho. Land ownership within the unit includes approximately 30,421 ac (12,311 ha) of Federal land administered by the Inyo National Forest and Bureau of Land Management. Unit 12 begins at a low elevation of about 4,000 ft (1,219 m) on the eastern slope and rises to about 9,000 to 10,500 ft (2,743–3,200 m) on the west. It is the southernmost unit designated as critical habitat for the Sierra Nevada bighorn sheep. This unit was not occupied at the time of listing and is not currently occupied, but is essential to the conservation of the Sierra Nevada bighorn sheep. The unit contains steep, rocky terrain which provides for foraging (summer and winter), mating, lambing, predator avoidance, and bedding and also allows for seasonal elevational movements, and a range of vegetation types (PCE 1 and PCE 2) (Johnson et al. 2005, pp. 4–14, 31–32, 34, 37–38; Service 2007, pp. 3–5). It is not known if mineral licks (PCE 3) occur within this unit. This unit provides bighorn sheep habitat in the areas of Ash, Braley, Cartago, Olancho, and Falls Creeks. Cartago, Olancho and Falls Creeks connect by Olancho Canyon to Olancho Peak (12,123 ft, 3,695 m) which provides some alpine summer habitat (southernmost in the Sierra Nevada) (Service 2007, p. 133). Winter range occurs as open, low-elevation (4,757 ft, 1,450 m), south-facing slopes (Service 2007, pp. 128, 133). While this unit was not occupied at the time of listing, Sierra Nevada bighorn sheep occupied the area historically (Jones 1950, p. 39; Wehausen et al. 1987, p. 66; Wehausen 1988a, p. 101). This unit is essential to the conservation of the Sierra Nevada bighorn sheep for increasing the number of herds to reduce the significance of losing any particular herd, increasing population viability, decreasing the degree of fragmentation of the current geographic distribution between this unit and Unit 10 (Mount Langley), increasing opportunities for genetic exchange between these units, and increasing overall herd numbers to reduce extinction risk from stochastic events. Conservation of this unit is necessary to achieve the long-term viability of this subspecies within its range.

#### **Primary Constituent Elements/Physical or Biological Features**

Critical habitat units are designated for Mono, Fresno, Inyo, Tulare, and Tuolumne Counties, California. The primary constituent elements of critical habitat for the Sierra Nevada bighorn sheep are the habitat components that provide:

- (i) Non-forested habitats or forest openings within the Sierra Nevada from 4,000 ft (1,219 m) to 14,500 ft (4,420 m) in elevation with steep (greater than or equal to 60 percent slope), rocky slopes that provide for foraging, mating, lambing, predator avoidance, and bedding and that allow for seasonal elevational movements between these areas.
- (ii) Presence of a variety of forage plants as indicated by the presence of grasses (e.g., *Achnanthera* spp.; *Elymus* spp.) and browse (e.g., *Ribes* spp.; *Artemisia* spp., *Purshia* spp.) in winter, and grasses, browse, sedges (e.g., *Carex* spp.) and forbs (e.g., *Eriogonum* spp.) in summer.
- (iii) Presence of granite outcroppings containing minerals such as sodium, calcium, iron, and phosphorus that could be used as mineral licks in order to meet nutritional needs.

### **Special Management Considerations or Protections**

Critical habitat does not include manmade structures, such as buildings, aqueducts, airports, roads, and other paved areas, and the land on which they are located, existing on the effective date of this rule and not containing one or more of the primary constituent elements.

Fire suppression can modify the structure of Sierra Nevada bighorn sheep habitat by allowing taller vegetation, such as trees, to become established, resulting in cover for predators. Mountain lions, a primary predator of Sierra Nevada bighorn sheep, use vegetative cover and terrain to conceal themselves prior to attacks. Fires may have burned more frequently in the past in bighorn sheep habitat. Old ground and aerial photographs show habitats in the eastern Sierra Nevada had little vegetation tall enough to obstruct the vision of bighorn sheep; pinyon pine woodlands have mostly developed since 1860 (Miller and Tausch 2001, pp. 15–16). Continued suppression of fires in Sierra Nevada bighorn sheep range is a threat, as habitat succession alters the abundance of suitable bighorn sheep habitat and increases bighorn sheep vulnerability to mountain lion predation (Torres et al. 1996, p. 29). Performing habitat enhancements, such as prescribed burning, or enabling “let burn” policies, helps to provide open habitats. Open habitats will help to reduce predation by decreasing the effectiveness of ambushing by predators (such as mountain lions) from cover. Providing more open habitat will allow more opportunity for connectivity among herd units and likely promote greater gene flow to conserve genetic diversity. According to Johnson et al. (2005, p. 34), all of the herd units would benefit from forest reduction in winter range; those units that would incur the highest benefit are Units 8 and 10. Thus, the PCEs in all of the units occupied at the time of listing (Units 1, 2, 4, 6, 7, 8, and 10) may require special management considerations or protection to reverse the impacts of fire suppression.

There is limited development within Sierra Nevada bighorn sheep habitat because most habitat occurs on Federal lands; however, there is some recreational development (e.g., resorts). There are several paved and unpaved roads that access Federal lands within Sierra Nevada bighorn sheep habitat. For example, State Highway 120 is located primarily between Units 1 and 2, but some sections lie within Unit 1. Bighorn sheep have been killed due to collisions with vehicles on this road (65 FR 28; January 3, 2000). State Route 158 and Road 16S02 occur in or adjacent to portions of Units 2 and 10, respectively. The PCE’s in Units 1, 2, 4, and 10 require special

management considerations or protection to address the impacts from development activities, including road construction and maintenance within Sierra Nevada bighorn sheep habitat.

Domestic grazing allotments within the vicinity of Sierra Nevada bighorn sheep habitat should be reviewed and activities should be modified as necessary to prevent competition and contact between the domestic livestock (sheep and goats) and bighorn sheep. These modifications could include such variables as the number of domestic livestock allowed on an allotment, where the domestic livestock may graze on an allotment, and the length and timing of the grazing period. These variables can assist in reducing resource competition as well as reducing contact between domestic sheep (and goats) and bighorn sheep. The PCEs within Units 1, 2, and 4 may require special management considerations or protection to address the potential impacts of domestic sheep and goat grazing within Sierra Nevada bighorn sheep habitat. The PCEs within Units 1, 2, 3, 4, 5, 8, 10, and 12 may require special management considerations or protection to address the potential impacts of cattle grazing within Sierra Nevada bighorn sheep habitat.

Patented mining claims occur within habitat used by the Sierra Nevada bighorn sheep, but the area of the claims is small. Mining activities and associated facilities threaten bighorn sheep by causing the loss of vegetation structure required for foraging activities; the destruction of habitats used for escape, bedding, lambing, or connectivity between ranges; and the disturbance due to ongoing mining activities. Disturbance could modify bighorn sheep behavior or cause them to flee an area. Mining occurs within the habitat of Sierra Nevada bighorn sheep in Unit 4. These mines are underground, thus reducing some impacts of habitat loss. PCEs within this unit may require special management considerations or protection to address mining and associated facility development impacts within Sierra Nevada bighorn sheep habitat.

### ***Life History***

#### **Feeding Narrative**

Adult: Sierra Nevada bighorn sheep (*Ovis canadensis sierrae*) are primarily grazers; however, they may browse woody vegetation. They are opportunistic feeders, selecting the most nutritious diet from available vegetation. Plants consumed include varying mixtures of graminoids (grasses), browse (shoots, twigs, and leaves of trees and shrubs), and herbaceous plants, depending on season and location (64 FR 19300, USFWS 2007). The species competes with domestic livestock, and management practices for domestic livestock can result in overgrazing or forage competition (73 FR 45534).

#### **Reproduction Narrative**

Adult: Sierra Nevada bighorn sheep are k-selected, giving birth to one live young. The age to first reproduction ranges from 2 to 4 years, based on availability of food resources. The gestation period is 174 days. The lifespan of this species is 8 to 12 years, and throughout the course of its life, there are 6 to 8 reproductive events (USFWS 2008). Lambs are typically weaned between 1 and 7 months of age (65 FR 20). A typical sex ratio is 70 males to 100 females (73 FR 45534). Reproductive fitness is moderate, and reproductive capacity is low. The breeding season is from late fall to early winter, mostly in November and December (USFWS 2008). Rams engage in battles to determine dominance. Dominance is expressed via visual displays. Dominance interactions include displacement from a bedding site, kicking, butting, neck wrestling, and fights; horn clashes are the most widely known of these interactions. A horn clash can sound like a rifle shot, and can be heard for long distances during battles. The most dominant rams are

typically those with the largest bodies and horns. Dominance allows mating access to estrous females. Sierra Nevada Bighorn Sheep (*Ovis canadensis sierrae*) follow a polygynous mating strategy whereby the dominant males do most of the breeding (CDFW 2015a).

**Geographic or Habitat Restraints or Barriers**

Adult: Sierra Nevada bighorn sheep require extreme visual openness near precipitous rocks for predator escape. Large expanses lacking precipitous escape terrain, such as the Owens Valley, are substantial barriers to movement (USFWS 2008).

**Spatial Arrangements of the Population**

Adult: The distribution of bighorn sheep is naturally fragmented on the landscape. Male and female sheep commonly live in separate groups during much of the year (73 FR 45534).

**Environmental Specificity**

Adult: Narrow/ specialist

**Tolerance Ranges/Thresholds**

Adult: Moderate

**Site Fidelity**

Adult: Moderate

**Habitat Narrative**

Adult: The Sierra Nevada bighorn sheep inhabit open, upland, montane, and alpine habitats with rocky areas along the eastern slope of the Sierra Nevada from about 1,450 m (4,000 ft.) to approximately 4,000 m (14,500 ft.) (NatureServe 2015). The ecological integrity of the community is high. Vegetation is alpine vegetation. Although they occupy a range of habitats, their environmental specificity is narrow. Primary constituent elements include (1) nonforested habitats or forest openings within the Sierra Nevada from 4,000 ft. (1,219 m) to 14,500 ft. (4,420 m) in elevation, with steep (greater than or equal to 60 percent slope), rocky slopes that provide for foraging, mating, lambing, predator avoidance, and bedding, and that allow for seasonal elevational movements between these areas; (2) the presence of a variety of forage plants, as indicated by the presence of grasses and browse in winter, and grasses, browse, sedges, and forbs in summer; and (3) the presence of granite outcroppings containing minerals such as sodium, calcium, iron, and phosphorus that could be used as mineral licks to meet nutritional needs (73 FR 45534). The tolerance range of the species is moderate; site fidelity is also moderate. Sierra Nevada bighorn sheep require extreme visual openness near precipitous rocks for predator escape. Large expanses lacking precipitous escape terrain, such as the Owens Valley, are substantial barriers to movement (USFWS 2008). Sierra Nevada bighorn sheep exist in naturally fragmented populations. Male and female sheep commonly live in separate groups during much of the year (73 FR 45534).

***Dispersal/Migration*****Motility/Mobility**

Adult: High

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Seasonal movements.

**Dispersal**

Adult: Low (USFWS 2008)

**Immigration/Emigration**

Adult: Minimal, because subpopulations are isolated by large areas of unoccupied habitat (USFWS 2008).

**Dispersal/Migration Narrative**

Adult: Sierra Nevada bighorn sheep are highly mobile, moving seasonally and occupying a range of habitats. Dispersal is relatively low and immigration/emigration is minimal, because subpopulations are generally fragmented, limiting gene flow (USFWS 2008). The maintenance of migration corridors by means of the movement of rams between herds can counteract the effects of inbreeding that can develop with small, isolated populations (73 FR 45534).

***Population Information and Trends*****Population Trends:**

Increasing (USFWS 2007).

**Species Trends:**

Increasing (USFWS 2007).

**Resiliency:**

Moderate

**Representation:**

Low

**Redundancy:**

Low

**Population Growth Rate:**

The overall population of bighorn sheep in the Sierra Nevada showed dramatic annual increases after 1999. Six years after emergency listing, the minimum number of yearling and adult females that could be accounted for had increased by 265 percent, from 55 to at least 146; an annual compounded increase rate of 17.7 percent (USFWS 2007).

**Number of Populations:**

There are eight sub-populations: Mount Langley, Mount Baxter, Sawmill Canyon, Bubbs Creek, Mount Williamson, Wheeler Ridge, Mount Warren, and Mount Gibbs (USFWS 2008).

**Population Size:**

Estimated at more than 600 individuals in 2014 (CDFW 2015b).

**Resistance to Disease:**

Low (USFWS 2007)

**Additional Population-level Information:**

Since its emergency listing, Sierra Nevada bighorn sheep numbers have increased dramatically (USFWS 2007).

**Population Narrative:**

Population level and species level trends for the Sierra Nevada bighorn sheep are increasing dramatically since its emergency listing (USFWS 2007). There are eight sub-populations: Mount Langley, Mount Baxter, Sawmill Canyon, Bubbs Creek, Mount Williamson, Wheeler Ridge, Mount Warren, and Mount Gibbs (USFWS 2008). Currently, the population size has increased to around 600 individuals (CDFW 2015b). The resiliency of the species is moderate, but representation and redundancy are still low. The overall population of bighorn sheep in the Sierra Nevada showed dramatic annual increases after 1999. Six years after emergency listing, the minimum number of yearling and adult females that could be accounted for had increased by 265 percent, from 55 to at least 146; an annual compounded increase rate of 17.7 percent (USFWS 2007). Resistance to disease is low and continues to threaten the species.

**Threats and Stressors****Stressor:** Disease

**Exposure:** Contact with domestic sheep.

**Response:** Domestic sheep that graze in the habitat of Sierra Nevada bighorn sheep may introduce disease-causing pathogens to the herds.

**Consequence:** The transfer of disease-causing pathogens could result in major die-offs of Sierra Nevada bighorn sheep.

**Narrative:** The potential for the transfer of virulent disease organisms from domestic sheep to bighorn sheep in the Sierra Nevada was a key factor in listing the Sierra Nevada bighorn sheep. As discussed earlier, pneumonia, caused by *Pasteurella* alone, or in combination with other pathogens, is the most significant disease threat for bighorn sheep. Currently, domestic sheep graze on both private and federal land adjacent to Sierra Nevada bighorn sheep subpopulations. The potential for contact between the species occurs when stray domestic sheep enter bighorn sheep habitat, or when bighorn sheep encounter domestic sheep herds (64 FR 19300; USFWS 2007).

**Stressor:** Predation

**Exposure:** Mountain lions (*Felis concolor*) are the primary predator of bighorn sheep.

**Response:** They have been responsible for 96 percent of losses attributed to predation.

**Consequence:** Sierra Nevada bighorn sheep have incurred major losses, which was a key factor that put these sheep in danger of extinction.

**Narrative:** Mountain lion predation of bighorn sheep on winter ranges has accounted for the majority of documented mortalities since the late 1970s. This predation increased from the 1970s to the 1980s, and is postulated as the cause of a coincident and marked decrease in winter range use by bighorn sheep in the Sierra Nevada. Subsequent population declines have been attributed to this change in winter habitat selection. During 1982 and 1988 to 1990, four mountain lions that preyed on bighorn sheep in two winter ranges were removed to help protect those sheep herds (USFWS 2007).

**Stressor:** Small Population Size and Fragmented Distribution

**Exposure:** Sierra Nevada bighorn sheep have small populations and occur in fragmented habitat.

**Response:** Sierra Nevada bighorn sheep are subject to risks of loss of genetic variability, and are vulnerable to demographic effects.

**Consequence:** Loss of genetic variation is a special concern among small populations, because loss of heterozygosity occurs more quickly in small populations than in large ones. Variation in birth, death, immigration, and emigration rates, as well as the age and sex structure of populations, can cause fluctuations in population size that make small populations vulnerable to extinction.

**Narrative:** Because of the Sierra Nevada bighorn sheep has a small overall population size, a fragmented distribution of subpopulations, and likely low levels of genetic exchange among subpopulations, the loss of genetic variation and the risk of demographic effects continue to be concerns despite the increases in population size since listing (USFWS 2008).

**Stressor:** Avalanches and Heavy Winters

**Exposure:** Naturally occurring, random events such as heavy storms and avalanches may impact populations at high elevations.

**Response:** Sierra Nevada bighorn sheep that remain at high elevations during the winter are exposed to extreme cold, deep snow, and avalanches.

**Consequence:** May cause stress to the species, including low nutrient intake.

**Narrative:** Avalanches and heavy winters are a threat for Sierra Nevada bighorn sheep populations that remain at high elevation during winter. The most notable impact is the lower nutrient intake (USFWS 2008).

**Stressor:** Roadkill

**Exposure:** Two populations of Sierra Nevada bighorn sheep have ranges adjacent to paved roadways.

**Response:** Adjacent roads expose individuals from those subpopulations to potential hazards.

**Consequence:** Bighorn sheep have been killed by vehicles on several occasions.

**Narrative:** Two Sierra Nevada bighorn sheep subpopulations (Mount Warren and Mount Gibbs) have ranges adjacent to paved roadways, exposing individuals from those subpopulations to potential hazards. Bighorn sheep have been killed by vehicles in Lee Vining Canyon on several occasions (USFWS 2008).

## **Recovery**

### **Reclassification Criteria:**

A minimum of 50 yearling and adult females exist in the Kern Recovery Unit (Great Western Divide), 155 in the Southern Recovery Unit (Olancho Peak to Coyote Ridge), 50 in the Central Recovery Unit (Mount Tom to Laurel Mountain), and 50 in the Northern Recovery Unit (Mount Gibbs and Mount Warren), for a minimum total of 305 females (USFWS 2007).

The measures to prevent contact between domestic sheep/goats and bighorn sheep have been implemented and are successful (USFWS 2007).

### **Delisting Criteria:**

The minimum number of females required for downlisting per recovery unit has been maintained as an average for one bighorn sheep generation (7 years) with no intervention (e.g., population management, buffering populations through translocations, or captive breeding).

Herd status for delisting must entail at least three censuses, one at the beginning of the period (qualifying for downlisting), one at the end of the period, and one intermediate count for each herd unit. Maintaining this number of females over a generation should be sufficient to indicate that predation is managed, and that the number of individuals in the population is large enough to promote regular use of winter range. Sierra Nevada bighorn sheep need herd sizes to reach a certain threshold before they will use areas that predators may inhabit. This herd size provides for better herd vigilance against predation.

Bighorn sheep of both sexes are distributed in such a way that at least two herd units are occupied in the Kern Recovery Unit, six in the Southern Recovery Unit, two in the Central Recovery Unit, and two in the Northern Recovery Unit, for a total of 12 herd units. Currently, seven of those herd units are occupied.

A population viability analysis projects that all recovery units are viable. Recovery tasks related to monitoring and research have been accomplished, allowing the severity of secondary threats (including recreational disturbance, competition, loss of genetic diversity, and habitat changes due to altered fire regimes) to be adequately assessed. These threats have either been ameliorated or have been determined not to pose a significant risk to the population (USFWS 2007).

Regulatory mechanisms and land management commitments have been established that provide for long-term protection of Sierra Nevada bighorn sheep and both their summer and winter habitat. Protection considered long-term can be provided through appropriate institutional practices and cooperative agreements between agencies, landowners, and conservation organizations.

**Recovery Actions:**

- Immediate Actions: 1. Protect existing herds through: a. maximization of population growth; and b. predator management.
- 2. Augmenting small herds through translocations; larger numbers of individuals are more likely to make adequate use of winter range essential for achieving positive population growth because they are able to be more vigilant to the presence of potential predators.
- 3. Preventing contact between Sierra Nevada bighorn sheep and domestic sheep or goats.
- Future Actions: 1. Reintroduce bighorn sheep to vacant herd units that are essential to Recovery.
- 2. Monitor genetic variation of all herd units; take action to maintain variation, if necessary.

***Conservation Measures and Best Management Practices:***

- USFWS should work with the California Department of Fish and Game, Humboldt-Toiyabe and Inyo National Forests, and the Bureau of Land Management-Bishop Field Office to implement the recommended strategy for preventing contact between domestic sheep and bighorn sheep.
- Implementation of controlled burns and other habitat improvement projects on winter ranges for Sierra Nevada bighorn sheep.
- Selected removal of mountain lions from Sierra Nevada bighorn sheep winter range.
- Translocation efforts to augment smaller subpopulations and to establish new populations in unoccupied habitat that is necessary for recovery.

- Research should be initiated on potential threats to Sierra Nevada bighorn sheep, such as human recreation and the effects of wildfire on habitat quality and use of low-elevation winter range.

***Additional Threshold Information:***

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**References**

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## SPECIES ACCOUNT: *Panthera onca* (Jaguar)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; 1972

### **Physical Description**

The jaguar is the largest felid in the New World (Seymour 1989). Rangewide, jaguars measure about 1.5 to 2.4 meters (m) (5 to 8 feet [ft]) from nose to tip of tail and weigh from 36 to 158 kilograms (kg) (80 to 348 pounds [lb]), although the 80 and 348 lb weights are exceptional (Nowak 1999, Seymour 1989). Males are typically larger than females, with reports of males being 10 to 25 percent larger than females (Emmons 1999, Wildlife Conservation Society 2007) and up to 20 to 30 percent larger (Sunquist and Sunquist 2007). In the southern part of the range, females tend toward 45 to 68 kg (100 to 150 lb) and males toward 77 to 100 kg (170 to 220 lb). In Central America and southern Mexico, both sexes trend slightly larger than they do to the north or south. Leopold (1959) listed a weight range in Mexico of 63 to 113 kg (140 to 250 lb) for males and 45 to 82 kg (100 to 180 lb) for females. Jaguars have a relatively robust head, compact but muscular body, short limbs and tail, and powerfully built chest and forelegs (Leopold 1959, Nowak 1999, Seymour 1989, Tewes and Schmidly 1987, Wildlife Conservation Society 2007). They have the strongest teeth and jaws of any American cat, and their skulls are more massive than those of mountain lions (Brown and López-González 2001). Their canines are well developed (Seymour 1989) and effectively deployed. The overall coat of a jaguar is typically pale yellow, tan, or reddish yellow above, and generally whitish on the throat, belly, insides of the limbs, and underside of the tail, with prominent dark rosettes or blotches throughout (Seymour 1989).

### **Taxonomy**

The jaguar was divided into a number of subspecies based on physical characteristics, like skull morphology (Mearns 1901, Nelson and Goldman 1933, Hall 1981, Seymour 1989, Wozencraft 2005). Pocock (1939) as cited by Larson (1997), described eight subspecies of jaguars, including five North American subspecies (Brown and López-González 2001): *Panthera onca arizonensis*, ranging from Arizona southward to southern Sonora; *P. o. hernandesii*, ranging from southern Sonoran southward to the state of Guerrero, Mexico; *P. o. centralis*, ranging from south of the Isthmus of Tehuantepec down through Central America and into Colombia; *P. o. goldmani*, ranging from the Yucatan Peninsula; and *P. o. veraecrucis*, ranged from southern Texas and eastern Tamaulipas southward to Tabasco. More recently, molecular genetic analyses have revealed that subspecies recognition may not be warranted in jaguars (Eizirik et al. 2001, Johnson et al. 2002, Ruiz-Garcia et al. 2006). Further studies are warranted to determine if real genetic differences among jaguar populations exist. Culver, of the Arizona Cooperative Fish and Wildlife Research Unit, U.S. Geological Survey, is currently working to assess the molecular taxonomy of northern jaguars (from Arizona and Sonora) compared to data from jaguars rangewide. The results from this analysis should be available in 2012. Culver and Ochoa Hein (2016) determined that the levels of mitochondrial and nuclear genetic diversity found in the Sonora/Arizona samples were reduced relative to Eizirik et al.'s (2001) Northern and Southern populations, reflecting a general pattern for peripheral populations with a small effective size. Culver and Ochoa Hein (2016) recommended international cooperation to promote connectivity among jaguar populations. The NRU populations are of conservation interest because of unique genetic diversity and because peripheral populations (such as those within the NRU) have a

greater likelihood of suffering a local extinction (i.e., extirpation) . Additionally, peripheral populations often harbour rare genetic diversity, which might be adaptive (Culver and Ochoa Hein 2016) and, therefore, potentially beneficial for the species in light of climate change. The addition of 20 or more samples from Sinaloa and Jalisco would clarify the genetic relationships within the NRU and between the NRU and other jaguar populations. (USFWS 2018)

### Historical Range

Jaguars historically occurred in California, Arizona, New Mexico, Texas, and possibly Louisiana (U.S. Fish and Wildlife Service 1997). The last jaguar sightings in California, Texas, and Louisiana were documented in the late 1800s into the early 1900s, with the last confirmed jaguar killed in Texas in 1948 (Nowak 1975). While jaguars have been documented as far north as the Grand Canyon, Arizona, occurrences in the U.S. since 1963 have been limited to south-central Arizona and extreme southwestern New Mexico. Three records of females with cubs have been documented in the U.S. (all in Arizona), the last in 1910 (Lange 1960, Nowak 1975, Brown 1989), and no females have been confirmed in the U.S. since 1963 (Brown and López González 2001, Johnson et al. 2011; note the validity of the 1963 record (a female jaguar killed in the White Mountains of Arizona) has been disputed—see Johnson et al. 2011 for further information). Due to the lack of females in the U.S. for many years, evidence suggests that jaguars in the U.S. are part of a population, or populations, that occur largely in Mexico; this is corroborated by genetic connectivity between jaguars in Arizona and Sonora (Culver and Ochoa Hein 2016) . From 1996 through July 2017, seven, possibly eight (see end of paragraph, below), individual jaguars have been documented in the U.S. (U.S. Fish and Wildlife Service 2014; [https://www.flickr.com/photos/usfws\\_southwest/sets/72157632294203147/](https://www.flickr.com/photos/usfws_southwest/sets/72157632294203147/)). (USFWS 2018)

### Current Range

U.S.: Cochise, Pima, Santa Cruz counties, Arizona; Hidalgo County, New Mexico. Mexico: Sonora, Chihuahua, Sinaloa, Jalisco. Belize. The species is now absent from much of the former range; it has been extirpated as a resident in most or all of the northern extent of the range in the southwestern United States and northern Mexico (see Federal Register, 13 July 1994, p. 35676, for discussion of recent records), El Salvador, Uruguay, developed areas of Brazilian coast, all but the northernmost parts of Argentina, and elsewhere. The largest remaining population is in Amazonian Brazil (Seymour 1989). In recent decades, jaguars occasionally have strayed into the United States in southern Arizona-New Mexico. (NatureServe 2015) Seven, possibly 8, adult males have been observed as follows: One adult male on March 7, 1996, in the Peloncillo Mountains in New Mexico near the Arizona border (Glenn 1996, Brown and López González 2001, U.S. Fish and Wildlife Service 2014). A second adult male was observed and photographed on August 31, 1996, in the Baboquivari Mountains of southern Arizona (Childs 1998, Brown and López González 2001, U.S. Fish and Wildlife Service 2014). In February 2006, a third adult male jaguar was observed and photographed in the northern part of the San Luis Mountains in Hidalgo County, New Mexico (U.S. Fish and Wildlife Service 2014). From 2001 to 2009, a fourth adult male jaguar and the jaguar observed and photographed in 1996 in the Baboquivari Mountains were photographed (one repeatedly) by camera traps in three different mountain range complexes in south-central Arizona, near the Mexico border (U.S. Fish and Wildlife Service 2014). A fifth jaguar (adult male) was observed and photographed in November 2011 in the Whetstone Mountains (U.S. Fish and Wildlife Service 2014). And in the Santa Rita Mountains ([https://www.flickr.com/photos/usfws\\_southwest/sets/72157632294203147/](https://www.flickr.com/photos/usfws_southwest/sets/72157632294203147/)). A sixth adult male jaguar was observed and photographed in the Huachuca Mountains from December 2016 through March 2017

([https://www.flickr.com/photos/usfws\\_southwest/32161863720/in/album72157632294203147/](https://www.flickr.com/photos/usfws_southwest/32161863720/in/album72157632294203147/)). A seventh male jaguar was photographed from November 2016 through July 2017 in the Dos Cabezas and Chiricahua Mountains ([https://www.flickr.com/photos/usfws\\_southwest/sets/72157632294203147/](https://www.flickr.com/photos/usfws_southwest/sets/72157632294203147/)). A possible eighth jaguar was photographed in 2004; however, it could not be determined if the animal was a unique individual (U.S. Fish and Wildlife Service 2014). (USFWS 2018)

**Distinct Population Segments Defined**

No.

**Critical Habitat Designated**

Yes; 3/5/2014.

**Legal Description**

On March 5, 2014, the U.S. Fish and Wildlife Service (Service) designated critical habitat for the jaguar (*Panthera onca*) under the Endangered Species Act, as amended. In total, approximately 309,263 hectares (764,207 acres) in Pima, Santa Cruz, and Cochise Counties, Arizona, and Hidalgo County, New Mexico, fall within the boundaries of the critical habitat designation.

**Critical Habitat Designation**

The Service has designated 6 units as critical habitat for the jaguar. The critical habitat areas described below constitute our best assessment at this time of areas that meet the definition of critical habitat. Those 6 units are: (1) Baboquivari Unit divided into subunits (1a) Baboquivari-Coyote Subunit, including the Northern Baboquivari, Saucito, Quinlan, and Coyote Mountains, and (1b) the Southern Baboquivari Subunit; (2) Atascosa Unit, including the Pajarito, Atascosa, and Tumacacori Mountains; (3) Patagonia Unit, including the Patagonia, Santa Rita, Empire, and Huachuca Mountains, and the Canelo and Grosvenor Hills; (4) Whetstone Unit, divided into subunits (4a) Whetstone Subunit, (4b) Whetstone-Santa Rita Subunit, and (4c) Whetstone-Huachuca Subunit; (5) Peloncillo Unit, including the Peloncillo Mountains both in Arizona and New Mexico; and (6) San Luis Unit, including the northern extent of the San Luis Mountains at the New Mexico-Mexico border.

Unit 1: Baboquivari Unit Subunit 1a—Baboquivari-Coyote Subunit: Subunit 1a consists of 16,925 ha (41,823 ac) in the northern Baboquivari, Saucito, Quinlan, and Coyote Mountains in Pima County, Arizona. The main, larger section of this subunit is generally bounded by the eastern boundary of the Tohono O'odham Nation to the west and north, the western side of the Altar Valley to the east, and up to and including Leyvas Canyon and Three Peaks to the south. There are four small areas of land that are disconnected from the main section of this subunit. One is a privately owned area within the boundaries of the Tohono O'odham Nation approximately 4 km (2.5 mi) west of the main, largest section and approximately 22.7 km (14.1 mi) south of State Highway 86. The second largest area is almost directly north of the main, largest section and is primarily Federally and State owned, with a small amount of private land included within the boundary. Between this area and the main, largest section is a small piece of State land included within the boundary. The last area is north and slightly west of the main section, and is a privately owned area within the boundaries of the Tohono O'odham Nation. Land ownership within the entire unit includes approximately 4,396 ha (10,862 ac) of Federal lands; 9,239 ha (22,831 ac) of Arizona State lands; and 3,290 ha (8,130 ac) of private lands. The Federal land is administered by the Service and Bureau of Land Management. We consider the Baboquivari-

Coyote Subunit occupied at the time of listing (37 FR 6476; March 30, 1972) (see “Occupied Area at the Time of Listing” section, above), and it may be currently occupied, based on jaguar photos from 1996 and from 2001–2008 (see Table 1 in the “Class I Records” section, above). It contains all elements of the physical or biological feature essential to the conservation of the jaguar, except for connectivity to Mexico. The primary land uses within Subunit 1a include ranching, grazing, borderrelated activities, Federal land management activities, and recreational activities throughout the year, including, but not limited to, hiking, birding, horseback riding, and hunting. Activities that may require special management may include, for example, habitat clearing, the construction of facilities, expansion of linear projects that may fragment jaguar habitat, some fuels-management activities, and some prescribed fire. Subunit 1b—Southern Baboquivari Subunit: Subunit 1b consists of 8,624 ha (21,312 ac) in the southern Baboquivari Mountains in Pima County, Arizona. This subunit is generally bounded by the eastern boundary of the Tohono O’odham Nation to the west, up to but not including Leyvas and Bear Canyons to the north, the western side of the Altar Valley to the east, and the U.S.- Mexico border to the south. There is one small, privately owned area within the boundaries of the Tohono O’odham Nation that is disconnected from the main section of this subunit. It is located approximately 1.2 km (0.75 mi) west of the main, largest section and approximately 10 km (6.2 mi) north of the U.S.-Mexico border. Land ownership within the unit includes approximately 624 ha (1,543 ac) of Federal lands; 6,157 ha (15,213 ac) of Arizona State lands; and 1,843 ha (4,555 ac) of private lands. The Federal land is administered by the Service and Bureau of Land Management. The Southern Baboquivari Subunit provides connectivity to Mexico and was not occupied at the time of listing, but is essential to the conservation of the jaguar because it contributes to the species’ persistence by providing connectivity to occupied areas. The primary land uses within Subunit 1b include ranching, grazing, borderrelated activities, Federal land management activities, and recreational activities throughout the year, including, but not limited to, hiking, birding, horseback riding, and hunting.

Unit 2: Atascosa Unit Unit 2 consists of 58,625 ha (144,865 ac) in the Pajarito, Atascosa, and Tumacacori Mountains in Pima and Santa Cruz Counties, Arizona. Unit 2 is generally bounded by the eastern side of San Luis Mountains (Arizona) to the west, roughly 4 km (2.5 mi) south of Arivaca Road to the north, Interstate 19 to the east, and the U.S.-Mexico border to the south. Land ownership within the unit includes approximately 53,807 ha (132,961 ac) of Federal lands; 2,296 ha (5,672 ac) of Arizona State lands; and 2,522 ha (6,231 ac) of private lands. The Federal land is administered by the Coronado National Forest and Bureau of Land Management. We consider the Atascosa Unit occupied at the time of listing (37 FR 6476; March 30, 1972) (see “Occupied Area at the Time of Listing” section, above), and it may be currently occupied based on multiple photos of two, or possibly three, jaguars from 2001–2008 (see Table 1 in the “Class I Records” section, above). It contains all elements of the physical or biological feature essential to the conservation of the jaguar. The primary land uses within Unit 2 include Federal land management activities, border-related activities, grazing, and recreational activities throughout the year, including, but not limited to, hiking, camping, birding, horseback riding, picnicking, sightseeing, and hunting. Activities that may require special management may include, for example, habitat clearing, the construction of facilities, expansion of linear projects that may fragment jaguar habitat, some fuels-management activities, and some prescribed fire.

Unit 3: Patagonia Unit Unit 3 consists of 142,248 ha (351,501 ac) in the Patagonia, Santa Rita, Empire, and Huachuca Mountains, as well as the Canelo and Grosvenor Hills, in Pima, Santa Cruz, and Cochise Counties, Arizona. Unit 3 is generally bounded by a line running roughly 3 km (1.9

mi) east of Interstate 19 to the west; a line running roughly 6 km (3.7 mi) south of Interstate 10 to the north; Cienega Creek and Highways 83, 90, and 92 to the east, including the eastern slopes of the Empire Mountains; and the U.S.-Mexico border to the south. Land ownership within the unit includes approximately 101,354 ha (250,452 ac) of Federal lands; 11,847 ha (29,274 ac) of Arizona State lands; and 29,046 ha (71,775 ac) of private lands. The Federal land is administered by the Coronado National Forest, Bureau of Land Management, and National Park Service. We consider the Patagonia Unit occupied at the time of listing (37 FR 6476; March 30, 1972) based on the 1965 record from the Patagonia Mountains (see “Occupied Area at the Time of Listing” section, above) and currently occupied based on photos taken from October 2012, through September 11, 2013, of a male jaguar in the Santa Rita Mountains (see Table 1 in the “Class I Records” section, above). The mountain ranges within this unit contain all elements of the physical or biological feature essential to the conservation of the jaguar. The primary land uses within Unit 3 include Federal land management activities, border-related activities, grazing, and recreational activities throughout the year, including, but not limited to, hiking, camping, birding, horseback riding, picnicking, sightseeing, and hunting. Activities that may require special management may include, for example, habitat clearing, the construction of facilities, expansion of linear projects that may fragment jaguar habitat, some fuels-management activities, and some prescribed fire.

Unit 4: Whetstone Unit Subunit 4a—Whetstone Subunit: Subunit 4a consists of 25,284 ha (62,479 ac) in the Whetstone Mountains, including connections to the Santa Rita and Huachuca Mountains, in Pima, Santa Cruz, and Cochise Counties, Arizona. Subunit 4a is generally bounded by a line running roughly 4 km (2.5 mi) east of Cienega Creek to the west, a line running roughly 6 km (3.7 mi) south of Interstate 10 to the north, Highway 90 to the east, and Highway 82 to the south. Land ownership within the subunit includes approximately 16,066 ha (39,699 ac) of Federal lands; 5,445 ha (13,455 ac) of Arizona State lands; and 3,774 ha (9,325 ac) of private lands. The Federal land is administered by the Coronado National Forest and Bureau of Land Management. We consider the Whetstone Subunit 4a occupied at the time of listing (37 FR 6476; March 30, 1972) (see “Occupied Area at the Time of Listing” section, above), and, based on photographs taken in 2011, it may be currently occupied (see Table 1 in the “Class I Records” section, above). The mountain range within this subunit contains all elements of the physical or biological feature essential to the conservation of the jaguar, except for connectivity to Mexico. The primary land uses within Subunit 4a include Federal land management activities, grazing, and recreational activities throughout the year, including, but not limited to, hiking, camping, birding, horseback riding, picnicking, sightseeing, and hunting. Activities that may require special management may include, for example, habitat clearing, the construction of facilities, expansion of linear projects that may fragment jaguar habitat, some fuels-management activities, and some prescribed fire. Subunit 4b—Whetstone-Santa Rita Subunit: Subunit 4b consists of 5,143 ha (12,710 ac) between the Empire Mountains and northern extent of the Whetstone Mountains in Pima County, Arizona. Subunit 4b is generally bounded by (but does not include): The eastern slopes of the Empire Mountains to the west, a line running roughly 6 km (3.7 mi) south of Interstate 10 to the north, the western slopes of the Whetstone Mountains to the east, and Stevenson Canyon to the south. Land ownership within the subunit includes approximately 532 ha (1,313 ac) of Federal lands and 4,612 ha (11,396 ac) of Arizona State lands. The Whetstone-Santa Rita Subunit provides connectivity from the Whetstone Mountains to Mexico and was not occupied at the time of listing, but is essential to the conservation of the jaguar because it contributes to the species’ persistence by providing connectivity to occupied areas. The primary land uses within Subunit 4b include grazing and recreational activities throughout

the year, including, but not limited to, hiking, camping, birding, horseback riding, picnicking, sightseeing, and hunting. Subunit 4c—Whetstone-Huachuca Subunit: Subunit 4c consists of 7,722 ha (19,081 ac) between the Huachuca Mountains and southern extent of the Whetstone Mountains in Santa Cruz and Cochise Counties, Arizona. Subunit 4c is generally bounded by Highway 83, Elgin-Canelo Road, and Upper Elgin Road to the west; Highway 82 to the north; a line running roughly 4 km (2.5 mi) west of Highway 90 to the east; and up to but not including the Huachuca Mountains to the south. Land ownership within the subunit includes approximately 1,350 ha (3,336 ac) of Federal lands; 2,981 ha (7,366 ac) of Arizona State lands; and 3,391 ha (8,379 ac) of private lands. The Federal land is administered by the Coronado National Forest and Bureau of Land Management. The Whetstone-Huachuca Subunit provides connectivity from the Whetstone Mountains to Mexico and was not occupied at the time of listing, but is essential to the conservation of the jaguar because it contributes to the species' persistence by providing connectivity to occupied areas. The primary land uses within Subunit 4c include Federal forest management activities, grazing, and recreational activities throughout the year, including, but not limited to, hiking, camping, birding, horseback riding, picnicking, sightseeing, and hunting.

Unit 5: Peloncillo Unit Unit 5 consists of 41,571 ha (102,724 ac) in the Peloncillo Mountains in Cochise County, Arizona, and Hidalgo County, New Mexico. Unit 5 is generally bounded by the eastern side of the San Bernardino Valley to the west, Skeleton Canyon Road and the northern boundary of the Coronado National Forest to the north, the western side of the Animas Valley to the east, and the U.S.-Mexico border on the south. Land ownership within the unit includes approximately 28,393 ha (70,160 ac) of Federal lands; 7,861 ha (19,426 ac) of Arizona State lands; and 5,317 ha (13,138 ac) of private lands. The Federal land is administered by the Coronado National Forest and Bureau of Land Management. We consider the Peloncillo Unit occupied at the time of listing (37 FR 6476; March 30, 1972) (see "Occupied Area at the Time of Listing" section, above), and it may be currently occupied based on a track documented in 1995 and photographs taken in 1996 (see Table 1 in the "Class I Records" section, above). It contains all elements of the physical or biological feature essential to the conservation of the jaguar. The primary land uses within Unit 5 include Federal land management activities, border-related activities, grazing, and recreational activities throughout the year, including, but not limited to, hiking, camping, birding, horseback riding, picnicking, sightseeing, and hunting. Activities that may require special management may include, for example, habitat clearing, the construction of facilities, expansion of linear projects that may fragment jaguar habitat, some fuels-management activities, and some prescribed fire.

Unit 6: San Luis Unit Unit 6 consists of 3,122 ha (7,714 ac) in the northern extent of the San Luis Mountains in Hidalgo County, New Mexico. Unit 6 is generally bounded by the eastern side of the Animas Valley to the west, a line running roughly 1.5 km (0.9 mi) south of Highway 79 to the north, an elevation line at approximately 1,600 m (5,249 ft) on the east side of the San Luis Mountains, and the U.S.-Mexico border to the south. Land within the unit is entirely privately owned. We consider the San Luis Unit occupied at the time of listing (37 FR 6476; March 30, 1972) (see "Occupied Area at the Time of Listing" section, above), and it may be currently occupied based on photographs taken in 2006 (see Table 1 in the "Class I Records" section, above). Unit 6 contains almost all elements of the physical or biological feature essential to the conservation of the jaguar except for expansive open space of at least 100 km<sup>2</sup> (38.6 mi<sup>2</sup>). This unit is included because, while by itself it does not provide at least 100 km<sup>2</sup> (38.6 mi<sup>2</sup>) of jaguar habitat in the United States, additional habitat can be found immediately adjacent south of the U.S.- Mexico border, and, therefore, this area represents a small portion of a much larger area of

habitat. The primary land uses within Unit 6 include border-related activities, grazing, and some recreational activities throughout the year, including, but not limited to, hiking, horseback riding, and hunting. Activities that may require special management may include, for example, habitat clearing, the construction of facilities, expansion of linear projects that may fragment jaguar habitat, some fuels-management activities, and some prescribed fire.

#### **Primary Constituent Elements/Physical or Biological Features**

Critical habitat units are designated for Pima, Santa Cruz, and Cochise Counties, Arizona, and Hidalgo County, New Mexico. Within these areas, the primary constituent elements of the physical or biological feature essential to the conservation of jaguar consists of expansive open spaces in the southwestern United States of at least 100 km<sup>2</sup> (32 to 38.6 mi<sup>2</sup>) in size which:

- (i) Provide connectivity to Mexico;
- (ii) Contain adequate levels of native prey species, including deer and javelina, as well as medium-sized prey such as coatis, skunks, raccoons, or jackrabbits;
- (iii) Include surface water sources available within 20 km (12.4 mi) of each other;
- (iv) Contain greater than 1 to 50 percent canopy cover within Madrean evergreen woodland, generally recognized by a mixture of oak (*Quercus* spp.), juniper (*Juniperus* spp.), and pine (*Pinus* spp.) trees on the landscape, or semidesert grassland vegetation communities, usually characterized by *Pleuraphis mutica* (tobosagrass) or *Bouteloua eriopoda* (black grama) along with other grasses;
- (v) Are characterized by intermediately, moderately, or highly rugged terrain;
- (vi) Are below 2,000 m (6,562 feet) in elevation; and
- (vii) Are characterized by minimal to no human population density, no major roads, or no stable nighttime lighting over any 1-km<sup>2</sup> (0.4-mi<sup>2</sup>) area.

#### **Special Management Considerations or Protections**

Jaguar habitat and the features essential to their conservation are threatened by the direct and indirect effects of increasing human influence into remote, rugged areas, as well as projects and activities that sever connectivity to Mexico. These may include, but are not limited to: Significant increases in border-related activities, both legal and illegal; construction of roadways, power lines, or pipelines; construction or expansion of human developments; mineral extraction and mining operations; military activities in remote locations; and human disturbance related to increased activities in or access to remote areas.

Jaguars in the United States are understood to be individuals dispersing north from Mexico (perhaps in some cases becoming resident in the United States), where the closest breeding population occurs about 210 km (130 mi) south of the U.S.-Mexico border in Sonora near the towns of Huasabas, Sahuaripa (Brown and Lo'pez Gonza'lez 2001, pp. 108–109), and Nacori Chico (Rosas-Rosas and Bender 2012, pp. 88– 89). Therefore, impeding jaguar movement from Mexico to the United States would adversely affect the Northwestern Recovery Unit's ability to cyclically expand and contract as jaguar populations in that unit recover.

Continuing threats from construction of border infrastructure (such as pedestrian fences and roads), as well as illegal activities and resultant law enforcement response (such as increased human presence, vehicles, and lighting), may limit movement of jaguars at the U.S.-Mexico border (Service 2007, pp. 23–27; 2008, pp. 73– 75). The border from the Tohono O’odham Nation, Arizona, to southwestern New Mexico has a mix of pedestrian fence (not permeable to jaguars), vehicle fence (fence designed to prevent vehicle but not pedestrian entry; it is generally permeable enough to allow for the passage of jaguars), legacy (older) pedestrian and vehicle fence, and unfenced segments (primarily in rugged, mountainous areas). Fences designed to prevent the passage of humans across the border also prevent passage of jaguars. However, there is little to no impermeable fence in areas designated as critical habitat, and we do not anticipate the construction of impermeable fence in such areas. Additionally, fences may cause an increase in illegal traffic and subsequent law enforcement activities in areas where no fence exists (such as rugged, mountainous areas). This activity may limit jaguar movement across the border and result in general disturbance to jaguars and degradation of their habitat. While current levels of law enforcement activity do not pose a significant threat, a substantial increase in activity levels could be of concern. We note that some level of law enforcement activity can be beneficial, as it decreases illegal traffic. Significant increases in illegal crossborder activities in the designated critical habitat areas could pose a threat to the jaguar, and, therefore, border security actions provide a beneficial decrease in crossborder violations and their impacts. In summary, special management considerations or protection of the physical or biological feature essential to the conservation of jaguar habitat may be needed to alleviate the effects of border-related activities, allowing for some level of permeability so that jaguars may pass through the U.S.-Mexico border.

Construction of roadways, power lines, or pipelines (all of which usually include maintenance roads), construction or expansion of human developments, mineral extraction and mining operations, and military operations on the ground can have the effect of altering habitat characteristics and increasing human presence in otherwise remote locations. Activities that can permanently alter vegetation characteristics, displace native wildlife, affect sources of water, and/or alter terrain ruggedness, such as construction and mining, may render an area unsuitable for jaguars. In addition, these activities, as well as military operations on the ground in remote areas, bring an increase in human disturbance into jaguar habitat, potentially fragmenting it further.

Special management considerations of the physical and biological feature essential to the conservation of the jaguar may be needed to alleviate the effects on jaguar habitat of new road construction or construction or expansion of power line and pipeline projects; human developments; mining operations; and ground-based military activities. Future projects should avoid (to the maximum extent possible) areas identified as meeting the definition of critical habitat for jaguars, and if unavoidable, should be constructed or carried out to minimize habitat effects.

Critical habitat does not include manmade structures (such as buildings, aqueducts, runways, roads, and other paved areas) and the land on which they are located existing within the legal boundaries on April 4, 2014.

### ***Life History***

**Feeding Narrative**

Adult: Jaguars, like other large cats, rely on a combination of cover, surprise, acceleration, and body weight to capture their prey (Schaller 1972 and Hopcraft et al. 2005, as cited by Cavalcanti 2008). Jaguars usually catch and kill their prey by stalking or ambush and biting through the nape as do most Felidae (Seymour 1989). The list of prey taken by jaguars range-wide includes more than 85 species (Seymour 1989). Known prey include, but are not limited to, peccaries, capybaras, pacas, agoutis, deer, opossum, rabbits, armadillos, caimans, turtles, livestock, and various other reptiles, birds, and fish (Seymour 1989, Núñez et al. 2000, Rosas-Rosas 2006, Rosas-Rosas et al. 2008). Jaguars are considered opportunistic feeders, especially in rainforests, and their diet varies according to prey density and ease of prey capture (Seymour 1989). Jaguars equally use medium- and large-size prey, with a trend toward use of larger prey as distance increases from the equator (López-González and Miller 2002). In coastal Jalisco, Núñez et al. (2000) found that jaguars killed eight different prey species. In order of preference (via biomass consumed), the four main prey species of jaguars were white-tailed deer (54 percent of biomass consumed), collared peccary (14.96 percent), coati (14.85 percent), and armadillo (12.49 percent). Combined, these species contributed 89 percent of occurrence and 98 percent of the biomass consumed by jaguars. Other prey items included black iguana, birds, opossum, and rabbit (Núñez et al. 2000). In northeastern Sonora, where the northern most breeding population of jaguars occurs, Rosas-Rosas (2006) found that large prey (>10 kg) accounted for >80 percent of the total biomass consumed. Specifically, cattle accounted for more than half of the total biomass consumed (57 percent), followed by white-tailed deer (23 percent), and collared peccary (5.12 percent). Medium sized prey (1–10 kg), including lagomorphs and coatis, accounted for <20 percent of biomass. Small prey (<1 kg body weight) were not found in scats. It is thought that collared peccary and deer are mainstays in the diet of jaguars in the U.S.-Mexico borderlands (62 FR3 9147), though other available prey, including livestock and coatis, are likely taken as well. In other areas, however, different prey items become important in their diet such as reptiles (e.g., caimans and turtles) or large rodents (e.g., paca and capybara) (Da Silveira et al. 2010). (USFWS 2012a)

**Reproduction Narrative**

Adult: Jaguars may breed year-round rangewide; however, at the southern and northern ends of their range there is evidence for a breeding season (Seymour 1989). On average, gestation is 101 days with cubs being born in a sheltered place (Seymour 1989). Litters range from one to four although usually consists of two cubs (Seymour 1989). Cubs remain with their mother for 1.5 to 2 years (Seymour 1989). Sexual maturity ranges from 2 to just over 3 years for females and 3 to 4 years for males (Seymour 1989). According to Seymour (1989), in Belize, Rabinowitz (1986) found few wild jaguars over 11 years of age. A wild male jaguar in Arizona was documented to be at least 15 years of age (Johnson et al. 2011). In Jalisco, two wild females were documented to be at least 12 and 13 (Núñez-Pérez, August 2, 2011, email to FWS). Therefore, the lifespan of the jaguar in the wild is estimated to be approximately 10-15 years; however this estimation is based on limited information.

**Geographic or Habitat Constraints or Barriers**

Adult: Fragmented habitat; large rivers (e.g., Amazon River) large/high mountains (e.g. Andean mountain chain) (USFWS 2012a)

**Spatial Arrangements of the Population**

Adult: Wide-ranging

**Habitat Narrative**

Adult: Jaguars are known from a variety of vegetation communities (Seymour 1989). Toward and at middle latitudes, they show a high affinity for lowland wet communities, including swampy savannas or tropical rain forests. Swank and Teer (1989) stated that jaguars prefer a warm, tropical climate, usually associated with water, and are rarely found in extensive arid areas. However, jaguars have been documented in arid areas, including thornscrub, desertscrub, lowland desert, mesquite grassland, Madrean oak woodland, and pine-oak woodland communities of northwestern Mexico and southwestern U.S. (Boydston and López-González 2005, McCain and Childs 2008, López-González and Brown 2002). The more open, dry habitat of southwestern U.S. has been characterized as marginal in terms of water, cover, and prey densities (Rabinowitz 1999). Brown and López-González (2001) report that the major habitat requirement appears to be a closed vegetative structure and that jaguars usually avoid open country like grassland or desertscrub. For this reason, jaguars rarely occur above 2,591 m (8,500 ft) (Brown and López-González 2001). Studies have also shown that jaguars selectively use large areas of relatively intact habitat away from certain forms of human influence. Zarza et al. (2007) report that towns and roads had an impact on the spatial distribution of jaguars. (USFWS 2012a)

***Dispersal/Migration*****Motility/Mobility**

Adult: High

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migrant

**Dispersal**

Adult: High

**Dispersal/Migration Narrative**

Adult: Jaguars can disperse across large distances (sometimes hundreds of kilometers). However, the jaguars' ability to effectively disperse across human-dominated landscapes that separate the current fragments has become very limited, and each fragment contains a small, isolated population that is already suffering from the effects of genetic drift (Haag et al. 2010). Jaguars require connectivity to areas of high quality habitat to endure over time.

***Population Information and Trends*****Population Trends:**

Decline

**Species Trends:**

Decline

**Resiliency:**

Small and isolated jaguar populations do not appear to be highly persistent (Haag et al. 2010, Rabinowitz and Zeller 2010). (USFWS 2018)

**Representation:**

n/a

**Redundancy:**

n/a

**Population Growth Rate:**

Low

**Number of Populations:**

Unknown

**Population Size:**

Habitat loss and fragmentation; killing for trophies/illegal trade in body parts; pro-active or retaliatory killing associated with livestock depredations; and competition for wild meat with human hunters (citations in Quigley et al. 2017) are the primary threats contributing to its current status, considered to have a decreasing population trend according to the IUCN (Quigley et al. 2017). Decline in range (20% loss from 2002 to 2015) indicate the species is trending toward Vulnerable (IUCN category) (Quigley et al. 2017). The estimated rangewide jaguar population is 173,000 (95% CI: 138,000–208,000) individuals, mostly concentrated in the Amazon Basin, with jaguar populations tending to be small and fragmented outside of this area (Jędrzejewski et al. 2018). The legal protected status in countries throughout its range does not appear to have secured jaguars in their core or corridor areas. Connectivity among jaguar populations is being lost at local and regional scales (sources as cited in Quigley et al. 2017) and small and isolated jaguar populations do not appear to be highly persistent (Haag et al. 2010, Rabinowitz and Zeller 2010). Additionally, jaguars require sufficient prey, and when prey is overharvested, jaguars may turn to livestock to meet their dietary needs, resulting in retaliatory killing. (FWS 2018)

**Minimum Viable Population Size:**

A model created from a population habitat viability analysis for jaguars in Mexico indicated that poaching mortality significantly reduces population growth and increases the risk of extinction of small populations (Carrillo et al. 2007). This effect is stronger in females, as when take is over 3% of the female population, the population becomes non-viable over a period of 100 years (Carrillo et al. 2007). According to the model, population sizes of < 100 individuals are not viable (Carrillo et al. 2007). (USFWS 2018)

**Resistance to Disease:**

n/a

**Adaptability:**

n/a

**Population Narrative:**

The number of occurrences or subpopulations is difficult to define for this species (individuals of which may range over vast areas) and not a very meaningful measure of conservation status. Population size and area of occupancy are more relevant considerations. Total adult population

size is unknown but is thought to have exceeded 100,000 in the 1960s (annual kills in Brazil alone were estimated at 15,000 in the 1960s). However, based on estimates of density and geographic range (Nowell and Jackson 1996), the jaguar's total effective population size has been estimated at fewer than 50,000 mature breeding individuals. A population of 600-1,000 exists in Belize, and there may be 500 in Guatemala and no more than 500 in all of Mexico (see Nowak 1999). Studies in the 1980s estimated numbers in the Pantanal of Brazil and its peripheral area to range from 1,000 to 3,500 individuals with an additional 1,400 individuals to the north of the Pantanal in the Guapore River Basin (see Swank and Teer 1989). The Paraguayan Gran Chaco may host a few thousand jaguars based on densities of 1 per 25 to 75 square kilometers in an area of 176,000 square kilometers. An analysis by Sanderson et al. (2002) determined that over 70% of the area where jaguars are thought to still occur has a high probability of supporting their long-term survival. Fifty-one jaguar conservation units were prioritized as the basis for a comprehensive jaguar conservation program; each of these could be regraded as an occurrence or subpopulation with good viability. (NatureServe 2015)

### ***Threats and Stressors***

**Stressor:** Rangewide habitat destruction and modification

**Exposure:** Not assessed; see narrative

**Response:** Not assessed; see narrative

**Consequence:** Not assessed; see narrative

**Narrative:** Range wide, habitat destruction, modification, and fragmentation form one of the two most significant threats to the jaguar (Nowell and Jackson 1996, Medellín et al. 2002, Núñez et al. 2002, Chávez and Ceballos 2006, Medellín 2009, Rodríguez-Soto et al. 2013, Petracca et al. 2014b, Quigley et al. 2017, Jędrzejewski et al. 2018). Rates of jaguar extirpations continue to increase, mainly due to habitat alteration (Jędrzejewski et al. 2018). Various factors, particularly habitat loss, have caused a considerable reduction in the historical range of the jaguar (Sanderson et al. 2002, Zeller 2007, Rabinowitz and Zeller 2010, Quigley et al. 2017). As of 2015, jaguars are estimated to occupy 51% of their historical range (Quigley et al. 2017). Most loss of occupied range has occurred in the southern U.S., northeastern Mexico, northern Brazil, and southern Argentina (Sanderson et al. 2002). Jaguar range mapping in 2015 indicates increasing fragmentation of jaguar populations, particularly in eastern and southeastern Brazil, northern Venezuela and the Maya Forest (Selva Maya) of Mexico and Guatemala. Deforestation rates are high in Latin America (e.g., Figure 10, p. 80) and close to 70% of deforestation in Latin American can be attributed to industrial agriculture (Quigley et al. 2017). (USFWS 2018)

**Stressor:** Human population growth

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Human population growth has both direct and indirect impacts on jaguar survival and mortality. For example, human growth and development tend to fragment habitat and isolate populations of jaguars and other wildlife. For carnivores in general, the impacts of high road density have been well documented and thoroughly reviewed (e.g., Noss et al. 1996, Carroll et al. 2001, as cited by Menke and Hayes 2003). Carnivores are particularly vulnerable to extinction in fragmented landscapes, owing to intrinsic biological traits, such as large body sizes, large area requirements, low densities, and slow population growth rates, as well as external anthropogenic threats, including hunting and other forms of direct mortality (sources as cited in Matthews et al.

2014). Roads may have direct impacts to carnivores and carnivore habitats, including mortality caused by vehicles (see Factor E), disturbance, habitat loss and fragmentation, changes in prey numbers or distribution, and provision of increased access for legal or illegal harvest (Menke and Hayes 2003, Colchero et al. 2010, Matthews et al. 2014). Roads are among the most widespread and impose some of the most lasting impacts on ecosystems of all human-made linear infrastructures (sources as cited in Matthews et al. 2014). (USFWS 2018)

**Stressor:** Illegal killing

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Illegal killing of jaguars is the other of the two most significant threats to the jaguar (Medellin 2009, Chávez and Ceballos 2006, Medellín et al. 2002, Núñez et al. 2002, Nowell and Jackson 1996) and, to recover jaguars, likely requires the most immediate response. Commercial hunting and trapping of jaguars for their pelts has declined drastically since the mid-1970s, when anti-fur campaigns and Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) controls progressively shut down international markets (Nowell and Jackson 1996). Although hunting (for pelts) has decreased, there is still demand for jaguar paws, teeth, and other products (Nowell and Jackson 1996). Additionally, illegal killing of jaguars due to conflicts with humans is a major threat to jaguars. Jaguars are known to kill cattle and are killed by ranchers as pest species (Nowell and Jackson 1996). People compete with jaguars for prey and jaguars are frequently shot on sight, despite protective legislation (Nowell and Jackson 1996). Continuing deforestation in Latin America and fragmentation of forest habitat isolates jaguar populations so that they are more vulnerable to human persecution (Nowell and Jackson 1996). Experts from throughout the jaguar range agree that one of the most severe causes of mortality is the direct hunting of jaguars, either because jaguars have caused some conflict by killing livestock or to sell the jaguar as a trophy or its skin or teeth (Medellin 2009). This illegal and indiscriminate killing eliminates hundreds or even thousands of jaguars each year in Latin America and must be controlled to reduce the risk of extinction (Medellin 2009). In western Mexico, illegal killing is considered the main threat to jaguars (Núñez-Pérez, pers. comm. 2011). In northwestern Mexico, Rosas-Rosas and Valdez (2010) reported that illegal hunting of jaguars and their potential prey species and habitat fragmentation are probably the main threats to long-term conservation of jaguars in their northernmost western range. According to the 1997 listing rule (U.S. Fish and Wildlife Service 1997), the primary threat to jaguars in the U.S. is illegal shooting (see listing rule for a detailed discussion). This, however, is no longer accurate and the most recent known shooting of a jaguar in Arizona was in 1986 (Brown and Lopez-González 2001). Illegal hunting of potential jaguar prey species is one of the main threats to long-term conservation of jaguars in northwestern Mexico (Rosas-Rosas 2006). Human population growth can put pressure on game populations that are used for human consumption. These same game populations are often prey for jaguars. (USFWS 2018)

**Stressor:** Deforestation

**Exposure:**

**Response:**

**Consequence:** Fragmentation of habitat

**Narrative:** Chávez and Ceballos (2006) reported that deforestation was one of the two most important threats to jaguars in Mexico; 60% of the jaguar's historical range in Mexico has been lost; the nationwide population was fewer than 5,000 individuals; and a variety of threats

suggested that, absent effective conservation efforts, jaguar imperilment in Mexico would only worsen. Deforestation rates are high in Latin America (e.g., Figure 10, p. 80) and close to 70% of deforestation in Latin American can be attributed to industrial agriculture (Quigley et al. 2017). Fragmentation of forest habitat isolates jaguar populations so that jaguars are more vulnerable to human persecution (Nowell and Jackson 1996, Quigley et al. 2017). Prey density is also reduced in leftover forest patches, which can lead to increased interactions between jaguars and livestock (Quigley et al. 2017). (USFWS 2018)

**Stressor:** Border Issues

**Exposure:**

**Response:**

**Consequence:** Range limitation or disruption

**Narrative:** Border Issues A number of activities along the U.S.-Mexico border may affect jaguar conservation. Continuing threats from construction and maintenance of border infrastructure (e.g., pedestrian and vehicle fences and walls, towers, roads), as well as illegal activities and resultant law enforcement response (e.g., increased human presence, vehicles, lighting) may limit movement of jaguars at the U.S.-Mexico border (U.S. Fish and Wildlife Service 2007 and 2008). In 2006, Congress passed the Secure Fence Act (Public Law 109–367), mandating that 700 miles of physical fencing be installed along the U.S.-Mexico border by the end of 2008. Fences and walls designed to prevent the passage of humans across the border also prevent passage of jaguars. Because jaguars in Arizona and New Mexico are part of a population(s) in northern Mexico, impeding jaguar movement from the Mexico to the U.S. would adversely affect the presence and persistence of jaguars in the U.S. Additionally, fences and walls may cause an increase in illegal traffic and subsequent law enforcement activities in areas where no barrier exists. This activity may limit jaguar movement across the border and result in general disturbance to jaguars and degradation of their habitat. As mentioned above, the U.S.-Mexico border is currently permeable to jaguar movement, and continued work with the U.S. Customs and Border Protection (CBP) will remain important to allow for ongoing opportunities for crossborder movement of jaguars. (USFWS 2018)

**Stressor:** Predator control programs

**Exposure:**

**Response:**

**Consequence:** Incidental or purposeful take

**Narrative:** Wildlife damage management programs may impact jaguars where these programs are implemented in the jaguar range. In the U.S., the U.S. Department of Agriculture Animal and Plant Health Inspection – Wildlife Services implements a nationwide animal damage control program that may impact jaguars in the southwestern U.S. Although jaguars are not a target of the program, according to the USFWS (1999), jaguars may be incidentally impacted by certain animal damage control methods used in the program (e.g., use of toxic chemicals, leghold traps, snares, dogs). However, incidental take of jaguars resulting from this program is authorized under section 7 of the ESA, and Wildlife Services implements reasonable and prudent measures to minimize any such take (U.S. Fish and Wildlife Service 1999). To date, no incidental take has been documented resulting from Wildlife Service’s program. (USFWS 2018)

**Stressor:** Loss of Genetic Diversity

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Little is known about the genetic health of jaguars. However, it has been documented that largescale habitat removal and fragmentation of once contiguous habitat have caused the reduction of genetic diversity in local jaguar populations, as well as drift-induced differentiation among local fragments. Citing a number of sources, Rabinowitz and Zeller (2010) explain that reduction or loss of genetic exchange leads to smaller effective population sizes, increased levels of genetic drift and inbreeding, and potential deleterious effects on sperm production, mating ability, female fecundity, and juvenile survival. Furthermore, they state that such effects eventually compromise adaptive potential, reduce fitness, and contribute to extinction risk for a population and, ultimately, for the species. (USFWS 2018)

**Stressor:** Climate Change

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Although it is too early to tell if the northern edge of jaguar range is expanding poleward, maintaining and enhancing the opportunity for range expansion of jaguars into suitable habitat may be a prudent precaution. Apart from monitoring and conserving the opportunity for range expansion, addressing the threat of climate change is generally beyond the scope of jaguar recovery planning and implementation. (USFWS 2018)

**Recovery****Reclassification Criteria:**

To be developed (USFWS 2012)

Downlisting criteria: A. PARU i. The status of the jaguar changes to Least Concern (LC) under the IUCN Red List criteria (as defined by the World Conservation Union, <http://www.iucnredlist.org/>), which would mean threats have been reduced such that the jaguar population is no longer at risk of a  $\geq 30\%$  decline because its area of occupancy, extent of occurrence, and/or habitat quality, as well as actual or potential levels of exploitation, have been stable for at least 20 years ( $\sim 3$  generations). (Factors A, C, D, E) B. NRU i. Maintain approximately 60% occupancy (proportion of cells) in each of the core areas over 20 years ( $\sim 3$  generations), as described in Appendix D. (Factors A, C, D, E) ii. Over 20 years ( $\sim 3$  generations), genetic distance (e.g.,  $F_{ST}$  or  $G_{ST}$ ) between the Sonora and Jalisco Core Areas does not significantly increase, and inbreeding coefficients (e.g.,  $F_{IS}$  or  $G_{IS}$ ) within each of the Sonora and Jalisco Core Areas do not significantly increase, as described in Appendix E. (Factors A, D, E) iii. Over a period of 20 years ( $\sim 3$  generations), the average of at least 30% of the adult population within the Sonora and Jalisco Core Areas are female (based on data gathered through surveying, monitoring, genetic analysis, etc.). (Factor E) iv. Within each core area (Sonora and Jalisco), a network of  $\geq 100\text{-km}^2$  blocks (the minimum area capable of supporting at least three breeding females) of high-quality habitat (as described in Appendix F) and habitat connections between blocks has been mapped and conditions in each block and connective area are described based on field visits. (Factor A) v. Within the Sinaloa Secondary Area, one or more potential linkages between the Jalisco and Sonora Core Areas sufficient to allow natural jaguar dispersal have been mapped based on documented use by jaguars, potential barriers and impediments have been mapped and/or identified based on field visits, and strategies for mitigating these impediments in the corridor have been developed and are being implemented. (Factor A) vi. Within the

Borderlands Secondary Area, two or more non-overlapping potential transborder linkages sufficient to allow natural jaguar dispersal have been mapped, potential barriers and impediments have been mapped based on field visits, and strategies for mitigating impediments in the corridor are being implemented. Additionally, half of the mapped linkages are clear of impediments and have obtained a sufficient level of protection within the corridor such that jaguar passage is attainable as measured by jaguar movement or other appropriate surrogate species, such as mountain lions. (Factor A, D) vii. The threat of direct human killing of jaguars is decreased or maintained at sustainable levels as measured by acceptable evidence or an index as described in Appendix G. (Factors D, E) viii. Effective Federal, State, Tribal, and/or local laws are in place or are being developed in the NRU that ensure that killing of jaguars is prohibited or regulated such that viable populations of jaguars can be maintained, and jaguars are highly unlikely to need to protection of the ESA again. (Factors D, E). (USFWS 2018)

**Delisting Criteria:**

To be developed (USFWS 2012)

A. PARU i. The status of the jaguar changes to Least Concern (LC) and maintain the LC status under the IUCN Red List criteria (as defined by the World Conservation Union, <http://www.iucnredlist.org>) for at least 20 more years after first qualifying for LC, which would mean threats have been reduced such that the jaguar population is no longer at risk of a  $\geq 30\%$  decline because its area of occupancy, extent of occurrence, and/or habitat quality, as well as actual or potential levels of exploitation, have been stable for at least 40 years (~6 generations, inclusive of the 20 years (~3 generations) required to downlist). (Factors A, C, D, E) B. NRU i. Maintain approximately 60% occupancy (proportion of cells) in each of the core areas over 40 years (~6 generations, inclusive of the 20 years (~3 generations) required to downlist), as described in Appendix D. ii. Over 40 years (~6 generations, inclusive of the 20 years (~3 generations) required to downlist), genetic distance (e.g., FST or GST) between the Sonora and Jalisco Core Areas does not significantly increase, and inbreeding coefficients (FIS or GIS) within each of the Sonora and Jalisco Core Areas do not significantly increase, as described in Appendix E. (Factor A, D, E) iii. Over a period of 40 years (~6 generations, inclusive of the 20 years (~3 generations) required to downlist), the average of at least 30% of the adult population within the Sonora and Jalisco Core Areas are female (based on data gathered through surveying, monitoring, genetic analysis, etc.). (Factor E) iv. Agency policies and regulations (including transportation), land use regulations, and land owner agreements in Mexico are sufficient to ensure that the network of  $\geq 100\text{-km}^2$  blocks (the minimum area capable of supporting at least three breeding females) of highquality habitat (as described in Appendix F) and habitat connections between blocks (as described in criterion 3.3.1.B.iv, above) within each core area (Sonora and Jalisco) will support genetically and demographically viable jaguar populations for the foreseeable future. Genetic and demographic viability will be demonstrated by meeting criteria i-iii, above. (Factors A, D, E). (USFWS 2018)

**Recovery Actions:**

- Document jaguar presence in a reliable fashion.
- Assess the habitat quality of occupied areas through some type of objective and defensible method.
- Conduct studies to better understand the impacts of highways on jaguar movement and the effectiveness of under and overpasses and other design measures to facilitate jaguar movement across these highways.

- Maintain and improve, when necessary, connectivity for movement of jaguars throughout the landscape and between populations to increase the long-term survival of subpopulations. This should include developing and maintaining highway under or overpasses and other design measures to facilitate jaguar movement where needed.
- Test and put in place conservation tools that 1) encourage the protection of jaguar habitat and corridors and 2) reduce illegal killing of jaguars. Although the tools will vary with the site, these are likely to include consideration of tax or other incentives (particularly economic incentives); enactment of new laws and enforcement of existing ones; research; and education programs to increase awareness of the value and current status of jaguars. Protection of jaguar habitat and corridors will not only help conserve jaguars, but will also improve ecosystem resiliency and the health of natural communities.
- Reduce conflicts between jaguars and livestock through improvement of livestock management practices, which have been proven through field studies. Applying these practices may not eliminate jaguar killing of livestock, but it will reduce it and consequently the motivation for retaliatory killing of jaguars.
- Reduce illegal and, where needed, legal hunting of jaguar prey through improved law enforcement programs and other programs aimed at developing alternative food sources for local communities.
- Obtain gender- and age-specific estimates of dispersal rates and travel distances.
- Obtain accurate information on the drought cycle.
- Obtain a better understanding of the extent to which poaching and depredation loss are compensatory with other types of mortality in order to conduct a more accurate PVA.
- Conduct research to understand the impact of subsistence hunting on jaguar prey species.
- Conduct research to understand interspecific competition between jaguars and pumas, specifically focused on competition for prey.
- The 2018 USFWS Recovery Action and Outline includes in general the individual items above as gathered and reported in 2012. However, the list is more specific and captured by its main sections here: 1. Ascertain the status and conservation needs of the jaguar: (1.1) Survey and monitor jaguars; (1.2) Increase collaboration with other carnivore researchers to gather information on jaguars in their study areas; (1.3) Develop and maintain jaguar observation report procedures and databases; (1.4) Conduct ecological research on jaguars; (1.5) Conduct periodic population viability analyses for jaguars as new information is acquired. 2. Assess and maintain or improve genetic fitness, demographic characteristics, and health of the jaguar: (2.1) Assess conservation genetic criteria for jaguars; (2.2) Investigate the taxonomic status of jaguars; (2.3) Assess demographic/vital characteristics of jaguars; (2.4) Develop estimates of dispersal rates and travel distances through genetic methods within the NRU and neighboring populations; (2.5) Evaluate and improve health conditions of jaguar populations. 3. Assess and maintain or improve the status of native prey populations: (3.1) Develop and conduct a study of jaguar prey abundance; (3.2) . Evaluate health conditions of jaguar prey populations, including the effects of diseases; (3.3) Design and implement a study that would quantify the relationship between jaguars and their prey as it relates to climate change; (3.4) Assess, evaluate, and implement wildlife management practices and laws that ensure sustainable prey bases for jaguars. 4. Assess, protect, and restore sufficient quantity, quality, and connectivity of habitat to support viable populations of jaguars: (4.1) . Assess jaguar habitat and corridors and their use; (4.2) Protect jaguar habitat and corridors; (4.3) Restore jaguar habitat and corridors. 5. Assess, minimize, and mitigate the effects of expanding human development on jaguar survival and mortality: (5.1)

Minimize the impacts of roads on jaguars; (5.2) Assess, avoid, minimize, and mitigate the impacts of other human development on jaguars (e.g., mines, dams, border infrastructure, housing and urban development, energy projects, railroads, large scale agriculture, etc.); (5.3) Monitor the effectiveness of actions implemented in 5.1. and 5.2. 6. Minimize direct human-caused mortality of jaguars: (6.1) Measure direct human-caused mortality of jaguars (\* Recovery Criteria 3.3.1.B.vii. and 3.3.2.B.vii.); (6.2) Determine, develop, fund, and implement education, outreach, and/or incentive programs to prevent the illegal killing of jaguars (also see action 6.6); (6.3) Analyze existing laws, strengthen and enact new laws, if needed, and enforce laws that control and reduce killing of jaguars (\* Recovery Criteria 3.3.1.B.viii. and 3.3.2.B.viii.); (6.4) Implement community programs to monitor and protect jaguars; (6.5) Monitor the effectiveness of the tools/programs/laws developed and implemented above in 6.2., 6.3., and 6.4; (6.6) Reduce conflicts between jaguars and livestock operations (the term livestock is used to include all hooved animals produced within the jaguar's range with which conflicts may occur; however, cattle are the primary concern). 7. Ensure long-term jaguar conservation through adequate funding, public education and outreach, and partnerships: (7.1) Secure funding for jaguar conservation; (7.2) Educate the public and professionals on jaguar conservation. 8. Practice adaptive management in which recovery is monitored and recovery tasks are revised by the USFWS in coordination with the JRT as new information becomes available: (8.1) Use adaptive management principles to evaluate this recovery effort on an ongoing basis, and make necessary changes, based on experience, outcomes, and changed circumstances; (8.2) Compile and discuss jaguar recovery accomplishments and updates with the JRT at least once per year; (8.3) Exchange information between agencies in Mexico and the U.S. to discuss progress in implementing state and federal jaguar recovery/conservation plans in the U.S. and Mexico; (8.4) Establish a binational agreement or letter of intent (Mexico-U.S.) to implement binational recovery actions in the Jaguar Recovery Plan and PACE. (USFWS 2018)

***Conservation Measures and Best Management Practices:***

- Support ecotourism or other conservation efforts that promote the conservation of jaguars within the NRU.
- Improve incentive programs for landowners within jaguar habitat in the NRU that support jaguar persistence, and increase the number of landowners enrolled in such programs.
- Support and increase the number of ranchers enrolled in photo incentive programs (i.e., payment for jaguar photos) in the NRU.
- Improve and increase landowner awareness of the importance of jaguars and their habitat.
- Mitigate human activity, such as road and mine development, through improved regulation within jaguar habitat in the NRU.
- Collaborate with Departments of Transportation, Regional Transportation Authorities, landowners, Department of Homeland Security, county planning offices, and others to voluntarily include jaguar conservation in their plans and activities.
- Work with the Ministry of Communication and Transportation in Mexico to encourage the use of under or overpasses and other appropriate measures (i.e., guiding fences) that decrease the risk of mortality associated with roads and facilitate jaguar movement across roads in the NRU (i.e., Highway 2 in northern Sonora when it is expanded from two to four lanes [planned for 2013] and any other highways that impede or could impede jaguar movement).

- The 2018 Recovery Plan has a detailed implementation plan (part 5, pages 126 to 168, and especially the detailed schedule that spans most of this section. Each action is reflected in the recovery goals (items 1 through 8 and subsections) and are too long to list in this summary. The value of this plan depends on the extent to which it is implemented; the USFWS has neither the authority nor the resources to implement many of the proposed recovery actions throughout the species' range outside of the U.S. The recovery of the jaguar is dependent upon the voluntary cooperation of many other organizations and individuals who are willing to implement the recovery actions. (USFWS 2018)

## References

USFWS 2012a. Recovery Outline for the Jaguar (*Panthera onca*), April 2012, U.S. Fish and Wildlife Service, Southwest Region. 59 p.

USFWS 2012b. Endangered and Threatened Wildlife and Plants, Designation of Critical Habitat for Jaguar, Proposed Rule: Federal Register (77 FR 50214)

NatureServe 2015. NatureServe Explorer: An online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available <http://explorer.natureserve.org>. (Accessed: June 15, 2015 ). USFWS (2018). Jaguar Recovery Plan – Final. Prepared by the Technical Subgroup of the Jaguar Recovery Team in conjunction with the Implementation Subgroup of the Jaguar Recovery Team and the USFWS, for Region 2, Southwest Region USFWS, Albuquerque, New Mexico. July 2018. 525 p.

Text used above not referenced in May 2018 update of SOS, but it's content is essentially the same as in the following reference (which has the same basic information but more detail): USFWS (2018). Jaguar Recovery Plan – Final. Prepared by the Technical Subgroup of the Jaguar Recovery Team in conjunction with the Implementation Subgroup of the Jaguar Recovery Team and the USFWS, for Region 2, Southwest Region USFWS, Albuquerque, New Mexico. July 2018. 525 p.

USFWS. 2012a. Recovery Outline for the Jaguar (*Panthera onca*), April 2012, U.S. Fish and Wildlife Service, Southwest Region. 59 p.

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## SPECIES ACCOUNT: *Pekania pennanti* (Fisher)

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### *Species Taxonomic and Listing Information*

**Commonly-used Acronym:** Fisher (West Coast Distinct Population Segment)

**Listing Status:** Endangered

### **Physical Description**

The fisher, as described by Powell (1981, p. 1), is a medium-sized light brown to dark blackish-brown mammal, with the face, neck, and shoulders sometimes being slightly gray. The chest and underside often has irregular white patches. The fisher has a long body with short legs and a long bushy tail. Fishers show regional variation in typical body weight. For example, fishers from western North America weigh more in the northern parts of their range than those living in the southern extent of their range (Lofroth et al. 2010, p. 10).(FWS 2014a)

### **Taxonomy**

The fisher (*Pekania pennanti*) is classified in the order Carnivora, family Mustelidae, a family that also includes weasels, mink, martens, and otters (Anderson 1994, p. 14). Characteristic of the genus *Pekania* is its large body size compared with *Martes* species, and the presence of an external median rootlet on the upper carnassial (fourth) premolar (Anderson 1994, p. 21). (FWS 2014a) The reclassification of the fisher to the genus *Pekania* has been accepted by Bradley et al. (2014, pp. 4, 6, 13) and added to the Revised Checklist of North American Mammals North of Mexico, 2014. (FWS 2016a)

### **Historical Range**

Fishers are found only in North America (Anderson 1994, pp. 22–23). Our present understanding of the historical (before European settlement) distribution of fishers is based on the accounts of natural historians of the early twentieth century and general assumptions of what constitutes fisher habitat. The presumed fisher range prior to European settlement of North America (circa 1600) was throughout the boreal forests across North America in Canada from approximately 60 degrees north latitude, extending south to the Great Lakes area and also along the Appalachian, Rocky, and Pacific Coast Mountains (Figure 5) in the United States (Hagmeier 1956, entire; Hall 1981, pp. 985–987; Powell 1981, pp. 1–2; Douglas and Strickland 1987, p. 513; Gibilisco 1994, p. 60; Lewis et al. 2012a, p. 9). The distribution of fishers has been described by numerous authors who delineate different distribution boundaries depending on the evidence used for occurrences. The presumed presence of fishers has been drawn along the lines of forest distribution, and the species has been consistently described as an associate of boreal forest in Canada, mixed deciduous-evergreen forests in eastern North America, and coniferous forest ecosystems in the west (Lofroth et al. 2010, p. 39). For this reason, range maps of historical distribution typically portray large areas of continuous occurrence, although it is likely that the suitability of habitat to support fishers within the portrayed range varied over time and spatial scales, subject to climatic variation, large-scale disturbances, and other ecological factors (Gibilisco 1994, p. 70; Graham and Graham 1994, pp. 57–58). Fishers do not occur in all forested habitats today, and evidence would indicate they did not occupy all forest types in the past (Graham and Graham 1994, p. 58). Likewise, recent genetic investigations point to the lack of a ubiquitous presence of fishers across the landscape. Tucker et al. (2012, entire) identified an

apparent break in the distribution and a range reduction along the length of the Sierra Nevada, which they estimated occurred prior to the influence of European settlement. (FWS 2014a)

**Current Range**

Since the 1950s, the overall population of fishers have recovered in some of the central (Minnesota, Wisconsin) and eastern (New England) portions of their historical range in the United States as a result of trapping closures, habitat regrowth, and reintroductions (Brander and Brooks 1973, pp. 53–54; Powell 1993, p. 80; Gibilisco 1994, p. 61; Lewis and Stinson 1998, p. 3; Proulx et al. 2004, pp. 55–57; Lewis et al. 2012a, p. 11). Fisher distribution is expanding into Virginia, from West Virginia in the Appalachian Mountains, but it is unclear whether they are establishing breeding populations (VDGIF 2012, p. 1). Two DPS's have been the subject of consideration for listing, Northern Rocky Mountain DPS (76 FR 38504, June 30, 2011) including forested areas of western Montana and north-central to northern Idaho, not currently proposed for listing and the West Coast DPS in Oregon, Washington and Northern California . (FWS 2014a)

**Critical Habitat Designated**

No;

***Life History*****Food/Nutrient Resources****Food Source**

Adult: Fishers are opportunistic predators, primarily of squirrels (*Tamiasciurus*, *Sciurus*, *Glaucomys*, and *Tamias* spp.), mice (*Microtus*, *Clethrionomys*, and *Peromyscus* spp.), snowshoe hares (*Lepus americanus*), and birds (numerous spp.) (reviewed in Powell 1993, pp. 18, 102; reviewed in Lofroth et al. 2010, pp. 74–76, 161–163). Fishers may indirectly shape forest plant communities through their influence on the population dynamics of prey species that are important seed predators in western coniferous forests (e.g., tree squirrels and other rodents that cache or hoard seeds) (e.g., Roemer et al. 2009, p. 170). Carrion and plant material (e.g., berries) also are consumed (Powell 1993, p. 18). The fisher is one of the few predators that successfully kills and eats porcupines (*Erethizon dorsatum*), (Powell 1993, p. 135). (FWS 2014a) Fishers in coastal Washington also prey upon mountain beaver (*Aplodontia rufa*) (Lewis 2014, p. 109). (FWS 2016a)

**Competition**

Adult: None apparent (FWS 2012a, inferred)

**Reproductive Strategy**

Adult: Polygamous

**Lifespan**

Adult: 10 years

**Breeding Season**

Adult: Fishers are solitary except females with kits and during the breeding season, which is generally from late February to the middle of May (Wright and Coulter 1967, p. 77; Frost et al. 1997, p. 607). The breeding period in California and Oregon begins in late February and lasts

through April based on observations of significant changes of fisher movement patterns (reviewed by Lofroth et al. 2010, p. 56). (FWS 2014s)

### **Key Resources Needed for Breeding**

Adult: Throughout their range, fishers use tree or snag cavities (Paragi et al. 1996a, entire; Truex et al. 1998, p. ii; Weir 2003, p. 12; Aubry and Raley 2006, p. 16; Higley and Matthews 2006, p. 10; Self and Callas 2006, p. 6; Weir and Corbould 2008, pp. 105–106; Davis 2009, p. 23) to give birth and raise their young (Coulter 1966, p. 81). Kits may be moved to numerous den locations (Arthur and Krohn 1991, p. 382; Paragi et al. 1996a, p. 80; Higley and Matthews 2006, p. 7) before they are weaned (Powell 1993, p. 67). (FWS 2014a)

### **Reproduction Narrative**

Adult: Fishers live to be about 10 years of age in the wild and captivity (Arthur et al. 1992, p. 404; Powell et al. 2003, p. 644) with both sexes reaching maturity their first year but often not becoming effective breeders until 2 years of age (Powell and Zielinski 1994, p. 46; Powell et al. 2003, p. 638). Fishers are solitary except females with kits and during the breeding season, which is generally from late February to the middle of May (Wright and Coulter 1967, p. 77; Frost et al. 1997, p. 607). The breeding period in California and Oregon begins in late February and lasts through April based on observations of significant changes of fisher movement patterns (reviewed by Lofroth et al. 2010, p. 56). Uterine implantation of embryos occurs 10 months after copulation; active gestation is estimated to be 36 days; and birth occurs nearly 1 year after copulation (Wright and Coulter 1967, pp. 74, 76; Frost et al. 1997, p. 609; Powell et al. 2003, p. 639). The proportion of adult female fishers that den each year in western North America is 0.64 (range = 0.39–1.00) (Lofroth et al. 2010, pp. 55–57; Matthews et al. 2013, pp. 103–104). Individual fishers may not give birth every year and reproductive rates may change as females age (Weir and Corbould 2008, p. 28). Among fishers who do give birth, the mean litter size for fishers is between one and three kits (litter size range from one to six kits) (Powell 1993, p. 53; Powell et al. 2003, pp. 639–640). The average litter size for 19 females during 4 den seasons on the Hoopa study area in Northern California was 1.9 kits (Matthews et al. 2013, p. 103). Within the analysis area females give birth between mid-March and mid-April (Truex et al. 1998, p. 36; Aubry and Raley 2006, p. 12; Higley and Matthews 2006, p. 8; Self and Callas 2006, p. 9; Weir and Corbould 2008, p. 78). Newborn kits are entirely dependent on the mother and are weaned at about 10 weeks of age (Powell 1993, p. 67). At about 4 months of age kits are mobile enough to travel with their mothers (Aubry and Raley 2006, p. 13). Throughout their range, fishers use tree or snag cavities (Paragi et al. 1996a, entire; Truex et al. 1998, p. ii; Weir 2003, p. 12; Aubry and Raley 2006, p. 16; Higley and Matthews 2006, p. 10; Self and Callas 2006, p. 6; Weir and Corbould 2008, pp. 105–106; Davis 2009, p. 23) to give birth and raise their young (Coulter 1966, p. 81). Kits may be moved to numerous den locations (Arthur and Krohn 1991, p. 382; Paragi et al. 1996a, p. 80; Higley and Matthews 2006, p. 7) before they are weaned (Powell 1993, p. 67). Once weaned, the kits stay with the female, utilizing multiple structures (for example, tree cavities, hollow logs, log piles) (Truex et al. 1998, p. 35; Aubry and Raley 2006, pp. 7, 16–17; Higley and Matthews 2006, pp. 6–7) within the female's home range until juveniles disperse in the fall or winter following their birth (Aubry and Raley 2006, p. 12; Matthews et al. 2009, p. 9). Kits become independent of their mother and develop their own home ranges by 1 year of age (Powell et al. 2003, p. 640).

### **Dispersal/Migration**

**Motility/Mobility**

Adult: Highly mobile (FWS 2014a)

**Dispersal**

Adult: Long distance dispersal has been documented for fishers with males moving greater distances than females. Juveniles dispersing from natal areas are capable of moving long distances and navigating various landscape features such as highways, rivers, and rural communities to establish their own home range (York 1996, p. 47; Weir and Corbould 2008, p. 44). Dispersal characteristics may be influenced by factors such as sex, availability of unoccupied areas, turnover rates of adults, and habitat suitability (Arthur et al. 1993, p. 872; York 1996, pp. 48–49; Aubry et al. 2004, pp. 205–207; Weir and Corbould 2008, pp. 47–48). Long distance dispersal by juveniles is made at a high cost and is usually not successful. (FWS 2014a)

**Dispersal/Migration Narrative**

Adult: Based on field observation and microsatellite genotype analyses of the fisher population in the southern Cascades, Aubry et al. (2004, p. 217) found empirical evidence of male-biased juvenile dispersal and female philopatry (the drive or tendency of an individual to return to, or stay in, its home area) in fishers, which may have a direct bearing on the rate at which fishers can colonize formerly occupied areas within their historical range. Tucker's (2013, p. 65) use of bi-parentally inherited genetic markers to investigate sex-biased dispersal of southern Sierra Nevada fishers yielded mixed results, but suggested that males disperse more often than do females. Research at the Hoopa study area also supports the theory that fishers have male-biased dispersal and female philopatry (Matthews et al. 2013 p. 105). (FWS 2014a)

**Additional Life History Information**

Adult: Dispersal by juvenile fisher begins during or after their first fall or winter when they are about seven to 10 months old (Aubry and Raley 2006, p. 14; Naney et al. 2012, p. 72). Juveniles in the southern Oregon Cascade Range began dispersing at about 10 months old in early February (Aubry and Raley 2006, p. 14). In the southern Sierra Nevada, juvenile dispersal likely begins in March (Sweitzer et al. 2015b, p. 5; Sweitzer et al. 2015d, pp. 36). (FWS 2014a)

***Population Information and Trends*****Population Trends:**

Despite the lack of precise empirical data on fisher numbers in the analysis area, the reduction in the range of the fisher on the west coast, as indicated by the lack of detections or sightings over much of its historical range, and apparent isolation from the main body of the species range (Drew et al. 2003, p. 59; Wisely et al. 2004, p. 646; Knaus et al. 2011, p. 11; Lewis et al. 2012a, p. 11; Tucker et al. 2014, pp. 132-133), reveal that the extant fisher populations are reduced in size relative to our understanding of their historical distribution. (FWS 2014a)

**Species Trends:**

In studies that have measured fisher populations over time, some have observed stable densities and others have recorded substantial changes. The NCSO population includes the original native fisher population in northern California and southern Oregon, the Southern Oregon Cascades (SOC), and the Northern Sierra Nevada (NSN) Reintroduced Populations. No published population or density estimates are available for the entire NCSO Population. (FWS 2014a) There are not enough data available from the Southern Oregon Cascades to determine

population trends. Recent detections of fisher in areas where they were not previously recorded (for example, northern and eastern portions of Crater Lake National Park and portions of the Lakeview and Medford BLM study areas) may or may not represent an expansion of this population. However, based on the current survey efforts along with multiple unsolicited sightings of fisher in the past few years on KFRA where fisher were previously known to be absent, fisher appear to be expanding into the KFRA (S. Hayner 2016, pers. comm.). (FWS 2016a)

**Resiliency:**

Female fishers with dens show stronger site fidelity, but still may use five or more den sites throughout a season (Paragi et al. 1996a, p. 80). This characteristic may make fishers more resilient to fire. However, because they are less vagile than spotted owls, fishers may be more sensitive to barriers to dispersal created by large patches of stand replacing fire. (FWS 2014a)

**Number of Populations:**

For the west coast DPS subject to listing, there are two populations: the Northern California-Southwestern Oregon Population and the Southern Sierra Nevada Population. Five separate translocations (reintroduced populations) have been attempted during the last 53 years (Aubry and Lewis 2003, p. 82; Lewis et al. 2012a, p. 8). Two of these reintroduction efforts were unsuccessful, one resulted in an established population (Southern Oregon Cascades), and the two most recent reintroductions (Olympic Peninsula and Northern Sierra Nevada) have not reported that they have met their criteria for success. (FWS 2014a)

**Population Size:**

There have been several approaches used to estimate the Northern California-Southwestern Oregon population size. Based on these various approaches, the Northern California-Southwestern Oregon population estimates range from a population size of 258 to 4,018. (FWS2014a) For the purpose of modeling population viability, Lamberson et al. (2000, p. 2) used expert opinion to estimate a population size between 100 and 500 individuals in the Southern Sierra Nevada Population. It is possible the Southern Oregon Cascades Reintroduced and Northern California Southwestern Populations may have become interconnected by dispersing fishers. There are no reliable estimates of population size. (FWS2016a) The Southern Oregon Cascades Reintroduced Population has persisted for over 30 years, despite estimates of a small population size. For both the Olympic Peninsula Reintroduced Population and the Northern Sierra Nevada Reintroduced Population, it is too early to determine if the populations will persist. Current indications are encouraging, but it will take time to determine population trend and stability of these two new reintroductions. (FWS 2016a)

**Population Narrative:**

Estimates of fisher abundance and vital rates are difficult to obtain and often based on harvest records, trapper questionnaires, and tracking information (Douglas and Strickland 1987, p. 522), and recent information is limited. Habitat modeling and behavioral or other natural history characteristics (e.g., home range sizes) also are used to estimate population sizes over a geographic area (Lofroth 2004, pp. 19–20; Lofroth et al. 2010, p. 50). Fisher densities over areas of suitable habitat have been reported, but there are no total or comprehensive population sizes for the fisher in the eastern United States or Canada. In the western range, fisher population size has been estimated using habitat models and home range size estimates. Habitat-based methods likely overestimate population sizes because some apparently suitable

habitat may not be occupied. A combination of habitat modeling, protocol surveys, and occupancy modeling can improve habitat-based population estimates. (FWS 2014a)

### ***Threats and Stressors***

**Stressor:** Habitat Loss (Loss of late-successional forest from past activities and disturbances)

**Exposure:** Natural disturbance (fire, forest insects, disease) and vegetation management (timber harvest, silviculture, fuel reduction)

**Response:** Negatively affect fisher reproduction and energy budgets (Lofroth et al. 2010, pp. 123–130, Naney et al. 2012, p. 22). Also, in many of the ecosystems in the analysis area, these structural elements are important habitat components for fisher prey (Aubry et al. 1991, pp. 292–294; Carey and Johnson 1995, pp. 347–349; Bowman et al. 2000, p. 23).

**Consequence:** Loss and fragmentation of fisher habitat; Loss of late-successional forest from past activities and disturbances

**Narrative:** Past and ongoing loss and fragmentation of fisher habitat may contribute to the decline of fisher populations (Aubry and Lewis 2003, p.82). Fragmentation can be caused by several anthropogenic factors (for example, vegetation management, conversion to agriculture, residential construction, and highways) and natural sources, such as large rivers, mountain ridgelines, and valley deserts or grasslands between forested areas (Green et al. 2008, pp. 19, 27, 29; Naney et al. 2012, p. 15). Anthropogenic factors causing fragmentation may compound habitat loss by isolating patches of suitable habitat within area of unsuitable or less suitable habitat, within which fishers may not be able to establish home ranges, forage (by affecting prey species composition, abundance, and availability), find suitable rest and den sites, or simply travel through (Buskirk and Powell 1994, p. 288; Hayes and Lewis 2006, p. 34; Weir and Corbould 2008, p. 148).

**Stressor:** Wildfire (Effects of fire, emergency fire suppression activities, post-fire management activities)

**Exposure:** Each population region experiences wildfires of differing sizes, frequencies, and severities. Within a region, different land cover types also burn with varying frequency and severity. These fire regimes are affected by naturally occurring climate and vegetation conditions as well as by human management decisions.

**Response:** Fires can cause reductions to or removal of important elements of fisher habitat, including vegetative diversity, over-story canopy cover, understory cover, and key structural elements (large hollow trees, large down logs, large live trees). Both low-severity fire and high-severity fire can cause changes to fisher habitat elements. Low-severity fire may reduce some habitat elements, such as understory cover, while increasing others, such as vegetative diversity, and both remove and create dead wood elements such as snags and down wood. High-severity fire is more likely to remove forest cover from large blocks of habitat.

**Consequence:** Resting and denning sites are likely to be lost as a result of fires, especially stand-replacing fires. When overstory canopy is markedly reduced, as in mixed- or moderate-severity fires, important microclimate characteristics are altered (for example, increased temperature or reduced shelter from wind and precipitation). Additionally, conflicts with other species or conspecifics may increase due to the open stand structure and absence of rest sites.

**Narrative:** Some fire suppression activities may affect fisher habitat. These include backburning (intentional burning to control the progression of wildfire), construction of fuel breaks (removal of all flammable material down to mineral soil), and removal of snags or other large trees. Some fire suppression activities occur on a relatively small spatial scale, while others occur over much

larger areas. In regard to emergency suppression, Backer et al. (2004, p. 937) state: “[t]he ecological impacts of fire-suppression activities can be significant and may surpass the impacts of the fire itself.” Salvage logging (harvest of dead or soon to be dead trees with commercial value) also occurs on the vast majority of private timberlands in the analysis area. Of large fires that burned U.S. Forest Service lands, salvage logging is ongoing on the Chips Fire and was completed on portions of the Biscuit, B and B, and Tripod Fires. Smaller fires are also salvage logged, but the number of these operations is difficult to estimate. This type of harvest can lead to increased erosion and sedimentation, damage to soils and nutrient-cycling processes, removal of snags and live trees, decreased regeneration of trees, shortened duration of early-successional ecosystems, increased spread of weeds from vehicles, damage to recolonizing vegetation, reduction in hiding cover and downed woody material for fisher prey, increased short-term and medium-term fire risk, and alterations of patterns of landscape heterogeneity (USDI FWS 2011, p. III-48). Moreover, these activities reduce the ecosystem benefit of disturbance from fire in diversifying and rejuvenating landscapes (Lindenmayer et al. 2004, p. 1303). The recent threat assessment for fishers also acknowledged that modification of forest structure from fire was greater when followed by post-fire salvage logging (Naney et al., 2012, page 31). Establishment of conifer plantations after salvage logging has been linked to higher severity in future fires (Perry et al. 2011, p. 709).

**Stressor:** Climate Change

**Exposure:** Changes in temperature and precipitation

**Response:** Climate change is likely to affect fisher habitat by altering the structure and tree species composition of fisher habitat, and also through the changes to habitat of prey communities. These effects may cause mortality, decrease reproductive rates, alter behavioral patterns, or lead to range shifts.

**Consequence:** Climate throughout the analysis area will become warmer over the next century, and in particular summers will be hotter and drier, with more frequent heat waves. In the northern portion of the analysis area, winters will likely become wetter, but even these areas will likely experience increased water deficits during the growing season. Ecotypes that support fisher habitat may decrease in area, especially in the Sierra Nevada, but also in Northern California-Southwestern Oregon, the Western Oregon Cascades, and possibly the Washington Eastern and Western Cascades, as a result of climate change. Where habitat area decreases the number of fishers that can be supported by the habitat will also decrease.

**Narrative:** In all or most sub-regions of the analysis area, fisher habitat will be altered, with likely shifts away from conifer forest and towards an increased hardwood component, or from maritime conifer forest to drier temperate conifer forest. It is uncertain how these habitat shifts will affect fisher populations. Modeling projections are done at a large scale and effects to species can be complex, unpredictable, and highly influenced by local level biotic and abiotic factors. In addition, disturbance regimes will change. Through much of the analysis area, fires are expected to increase in frequency and area burned. Insect and disease outbreaks will also increase. These changes will alter the structure of forested stands within fisher habitat, may increase the proportion of early-successional forest on the landscape, and may also combine synergistically to alter ecosystem types, which could result in losses of fisher habitat throughout the analysis area. Fisher populations are already fragmented and greatly reduced from their historical range. Loss of habitat could threaten the viability of native and reintroduced populations, and would reduce the likelihood of reestablishing connectivity between populations. (FWS2014a) In all or most sub-regions of the analysis area, fisher habitat will be altered, with likely shifts away from conifer forest and towards an increased hardwood

component, or from maritime conifer forest to drier temperate conifer forest. Potential changes in habitat suitability and fisher response are likely to vary regionally (for example, an increased hardwood component in conifer forests may have a neutral or even positive effect on fishers, whereas replacement of mixed conifer-hardwood forests with woodland will have a negative effect). It is uncertain how these habitat shifts will affect fisher populations, as it is not clear whether fisher response to these changes will be positive, neutral, or negative. Projections of future conditions in some cases predict losses of suitable fisher habitat, whereas others predict potential increases in suitable fisher habitat. Many predicted habitat changes are projected to occur over a relatively long period of time, further adding to the uncertainty in our ability to reliably predict future conditions for fisher. (FWS2016a)

**Stressor:** Vegetation Management

**Exposure:** Forest management activities (harvest, vegetation removal)

**Response:** Loss of cover and denning areas

**Consequence:**

**Narrative:** Percent changes in forest area disturbed between 1985 and 2008 range from a low of 9 percent on federally protected lands (all lands where harvest is not among the management goals) in Washington to a high of approximately 39 percent on private lands in both Oregon and Washington and tribal lands in Oregon. Beginning in the mid-1990s, around the time of the NWFP implementation, the magnitude of disturbances on federal lands declined substantially and has remained lower than that occurring on private lands (Kennedy et al. 2012, p. 128). As an indication of potential for future fisher habitat conditions on private timberlands, more than 75 percent of the future tree harvest is expected to come from private timberlands (Johnson et al. 2007, p. 37; Spies et al. 2007b, p. 50) and modeling of future timber harvests over the next 50 years indicates that current harvest levels on private lands in western Oregon can be maintained at that rate (Adams and Latta 2007, p. 13). Reduction in understory complexity and plant species diversity can result from silvicultural and fuels reduction treatments (for example, single species tree plantations, removal of hardwoods, pre-commercial thinning, herbicide application); and as a result may affect prey species abundance and diversity. However, the effects of understory treatment to fishers can vary greatly by the ecosystem type, the intensity and scale of treatments (Naney et al. 2012, pp. 29–37), and the response of the prey communities being affected by the treatments. (USFWS 2014a) As described earlier, while historical loss of older forests via timber harvest through much of the 1900s resulted in a substantial loss of fisher habitat in the west coast fisher analysis area, harvest volume has sharply declined throughout this area since 1990, primarily on Federal lands, but also on non-Federal lands. Although timber harvest is still ongoing throughout the DPS, there is habitat ingrowth that is occurring. Modeling in the southern Sierra Nevada region indicates that ingrowth of fisher habitat has even replaced habitat loss by all disturbances in the southern Sierra Nevada region since 1990, resulting in a net gain of habitat since that time; this holds true even including the preliminary estimates of habitat loss as a result of the 2013 and 2014 fires in the region. On Federal lands in the NWFP region, habitat ingrowth has been greater than that lost due to timber harvest in all fisher subregions except for the western Oregon Cascades (Table 7), and ingrowth is expected to eventually outpace total losses under existing management to the degree that within 50 to 100 years, older forests would be within the range of amounts occurring prior to logging and extensive fire suppression. However, there is a concern that some of those gains may be outdone with potential increased losses to fire in some of the drier regions.

**Stressor:** Human Development

**Exposure:** Besides permanently removing potential fisher habitat, human developments in rural areas are changing land use from forest to other land cover types, which can fragment previously continuous habitat or hamper fisher movements. (USFWS 2014a)

**Response:** Human developments associated with population growth will have an increasing impact on fisher habitat into the foreseeable future. The timing of development across the analysis area is ongoing. Within much of the analysis area, human development is generally considered to be of relatively low concern for fishers, and occurs at relatively small spatial scales in forested landscapes (Naney et al. 2012, p. 53). In other sub-regions, we estimated a higher scope; that is, development is likely to affect a larger proportion of fisher habitat. In western Washington (encompassing Coastal Washington and Western Washington Cascades), Bradley et al. (2007, pp. 268-269) estimated that from 1988 through 2004, 1.04% of privately-owned forest land was lost per year to agriculture, residential, or urban land uses. (USFWS 2014a)

**Consequence:**

**Narrative:** Severity varies depending on the type of development. We consider recreational development to be of low severity (approximately 5 percent) and urbanization to be of very high severity (90 percent). Other types of development, such as conversion to farmland or low-density rural housing, fall in between the two extremes. In Western Washington, approximately two thirds of the converted land shifted to agriculture and mixed-rural land uses, and approximately one third was developed for residential or urban use. Combined with our assumption that there will also be some low-severity recreational development, we therefore estimate severity to be approximately 50 percent for Coastal Washington and the Western Washington Cascades. For the Sierra Nevada, where most of the converted forested land is used for residential areas, we estimated severity to be approximately 60 percent. In the other sub-regions, we assume that development is as or more likely to consist of low-severity recreational use than higher-severity residential use, and estimate severity between 30 and 40 percent. (USFWS 2016a)

**Stressor:** Habitat loss attributed to linear features (highways and other infrastructure)

**Exposure:** As well as being sources of vehicle-collision mortality (addressed below in section on Collisions with Vehicles), most linear features represent some level of permanent removal or change of potential fisher habitat. (USFWS 2014a)

**Response:** Roads, highways, and associated developments can also substantially influence movement patterns of wildlife (Beier 1995, p. 234). Major highways and state highways may be impediments to fisher movements (e.g., home range establishment, juvenile dispersal, breeding season movements by males), thereby affecting population connectivity. A single linear feature may have a small effect on fisher movements, but multiple linear features (e.g., paved highways, railroad rights-of-way, and rivers) nearby may create more formidable filters and barriers to movement (Naney et al. 2012, p. 36). (USFWS 2014a)

**Consequence:**

**Narrative:** As we calculate the scope and severity of habitat loss from linear features, the timing of the habitat loss is mainly in the past. However, this stressor still affects fisher populations currently and will continue to do so for the foreseeable future. New road construction in fisher habitat is likely to be associated with human development (see previous section addressing Human development as stressor on fisher habitat) and is not included in the scope and severity calculations for linear features. Regardless of new construction, we expect that habitat previously lost due to linear features will remain as non-habitat for the foreseeable future. (USFWS 2014a)

**Stressor:** Trapping and Scientific Purposes

**Exposure:** Incidental trapping when traps are set for other species

**Response:** Loss of individuals from the population

**Consequence:**

**Narrative:** It is currently not legal to intentionally trap fishers in Washington, Oregon, or California. However, fishers are susceptible to incidental capture in traps set for other species (Earle 1978, p. 88; Luque 1983, p. 1; Lewis and Zielinski 1996, pp. 293–295). In all three states it is legal to harvest many mammals that are found in fisher habitat, including bobcat (*Lynx rufus*), gray fox (*Urocyon cinereoargenteus*), coyote (*Canis latrans*), mink (*Mustela vison*), and other furbearers. Red fox (*Vulpes vulpes*) and marten (*Martes americana*) may also be trapped in Washington and Oregon. In addition, it is unknown how many fishers are illegally harvested in each state each year.

**Stressor:** Disease and predation

**Exposure:** Disease in a wildlife population can contribute to the risk of extinction. First, it can kill animals at a faster rate than they can reproduce. Second, it can reduce the population size and increase the risk of extinction from stochastic events (Woodroffe 1999, p. 185). Third, diseases tend to have more severe effects on populations when the populations are small or insular, or when the disease agent acts synergistically with other population-limiting factors (Gabriel et al. 2012b, p. 139). Mortality from predation could be a significant stressor to fisher populations in the analysis area. Potential predators include mountain lions (*Felis concolor*), bobcats, coyotes, and large raptors (Powell and Zielinski 1994, p. 25; Truex et al. 1998, pp. 80–82; Higley and Matthews 2009, p. 14; Wengert 2010). Individuals weakened by parasitism or infectious diseases may be more vulnerable to predation. (USFWS 2014a)

**Response:**

**Consequence:** Weakened population; loss of individuals.

**Narrative:** These stressors are ongoing. Previously considered to be of minimal impact to fisher populations throughout their range, predation and disease now appear to be the most significant causes of mortality for California fishers. If disease affects fisher populations in patterns similar to disease outbreaks in other mustelids, there is the potential for disease to greatly reduce the size and extent of current fisher populations. (USFWS 2014a) We have become aware of information associated with disease that is related to canine distemper outbreaks in the Rogue River watershed in Oregon, which occurs north of the Oregon/California state boundary within the northern end of the NCSO population area. Specifically, ODFW reported that a recent outbreak (between January 2012 and January 2014) of canine distemper affected a wide variety of mid-size carnivores, potentially including fisher (Niemela 2015, pers. comm.). (USFWS 2016a) Lewis (2014, p. 67) reported that the cause of mortality for 14 of 35 reintroduced fishers recovered from 2008 to 2010 within the ONP population died from predation. Wengert (2013, pp. 38–39, 52, 59) reported that 62 of 101 fisher carcasses recovered from two California research projects (one in a portion of the SSN population, Kings River Fisher Project; and one in a portion of the NCSO population, Hoopa Valley Indian Reservation Fisher Project) were attributed to predation. (USFWS 2016a)

**Stressor:** Exposure to Toxicants (Rodenticides)

**Exposure:** Recent research documenting exposure to and mortalities from anticoagulant rodenticides (ARs), and other toxicants in California fisher populations, has raised concerns regarding both individual and population level impacts of toxicants within the fisher's range in the Pacific States (Gabriel et al. 2012a, entire). (USFWS 2014a)

**Response:** Carcass residues or possible death

**Consequence:**

**Narrative:** Legal uses of rodenticides may pose risks to fishers in some parts of their range. Rodenticides have a long history of use in forestry and crop agriculture. Use by homeowners of “ranchette” properties (one to five acres of land per home) may also contribute a legal source of rodenticides adjacent to or within fisher habitat (CDPR 2013a, pp. 5-6). The State of California requires that all agricultural pesticide use be reported monthly to county agricultural commissioners. The state maintains a broad definition of “agricultural use” so as to include applications to parks, golf courses, cemeteries, rangeland, pastures, and along roadside and railroad rights-of-way. The primary exceptions to the reporting requirements are that homeand-garden use, and most industrial and institutional uses are not required to be reported (California DPR website, <http://www.cdpr.ca.gov>). Therefore, we have concluded that the data pertaining to forest habitats (including habitat supporting fishers) is not captured adequately in these statistics nor does this reporting requirement represent the best source of data for assessing the potential affects on fishers from the use of ARs. A comparison of the areas where ARs are reported as being applied under labeled uses in California in relation to areas that are supportive of fisher habitats demonstrates legal applications of ARs are not likely the source for the ARs that have been observed in fishers by researchers. Although all sources of AR exposure in fishers have not been conclusively determined, large quantities of ARs have been found at illegal marijuana cultivation sites within occupied fisher habitat on public, private, and tribal lands in California (Gabriel et al. 2012a, p. 12; Thompson et al. 2014, pp. 97-98); ARs are found in significant amounts scattered around young marijuana plants to discourage herbivory and along plastic irrigation lines to poison rodents that might chew on them. The proximity of a large number of marijuana cultivation sites to fisher populations in California and Oregon (Figure 19, Figure 20) and the lack of other probable sources of ARs within occupied fisher habitat have led researchers to implicate marijuana cultivation sites as the source of AR exposure in fishers (Gabriel et al. 2012a, p. 12; Thompson et al. 2014, pp. 97-98). (USFWS 2014a) The total mortality of fishers in California due to toxicosis is 15 (Gabriel et al. 2015, p. 5; Wengert 2016, pers. comm.). Gabriel et al. 2015 (p. 7) reported for the fishers they analyzed in California the average incidence of toxicosis, from 2007-2011, was 5.6 percent. However, from 2012-2014, they detected an increase to 18.7 percent in incidence per year of toxicosis. In addition, Gabriel et al. 2015 (p. 7) found that, between 2012 and 2014, toxicant exposure of fishers in California has increased from 79 percent (46 of 58 individuals) to 85 percent (86 of 101 individuals). (USFWS 2016a)

### **Recovery**

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## SPECIES ACCOUNT: *Perognathus longimembris pacificus* (Pacific pocket mouse)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; September 29, 1994 (59 FR 49752).

### **Physical Description**

The Pacific pocket mouse (*Perognathus longimembris pacificus*) is a member of the family Heteromyidae. All members of this family are nocturnal granivores with external, fur-lined cheek pouches. The body pelage of the little pocket mouse is silky, and the dorsal pelage ranges in color from predominately brown, to a pinkish or ochraceous buff. The ventral pelage is whitish. There are typically two small patches of lighter hairs at the base of the ear. The tail can be either distinctly or indistinctly bicolored. The Pacific pocket mouse is among the smallest subspecies of little pocket mice, ranging up to 131 millimeters (mm) (5.2 inches [in.]) in length from nose to tip of tail. Little pocket mice weigh 7 to 9 grams (0.25 to 0.33 ounces) (USFWS 1998).

### **Taxonomy**

The Pacific pocket mouse is one of 16 recognized subspecies of the little pocket mouse (*Perognathus longimembris*) (USFWS 2010). This subspecies is the smallest member of the family Heteromyidae, which consists of spiny pocket mice (*Heteromys* and *Liomys*), pocket mice (*Perognathus* and *Chaetodipus*), kangaroo rats (*Dipodomys*), and kangaroo mice (*Microdipodops*) (59 FR 49752).

### **Historical Range**

Historical records indicate that the Pacific pocket mouse occurred in eight general areas, encompassing some 29 separate trapping sites. The records included locations in Los Angeles County in Marina del Rey/El Segundo, Wilmington, and Clifton. Locations in Orange County include the San Joaquin Hills and Dana Point. The species was found in Black Gully and nearby "Spyglass Hill" in the San Joaquin Hills. There are possible recent records from Crystal Cove State Park, although they are awaiting confirmation due to uncertainty over results of recent walk-over and trapping surveys. In San Diego County, the Pacific pocket mouse occurred at three general locales: the San Onofre Area, the Santa Margarita River Estuary, and the lower Tijuana River Valley (59 FR 49752).

### **Current Range**

The Pacific pocket mouse is endemic to the immediate coast of southern California—from Marina del Rey and El Segundo in Los Angeles County, south to the vicinity of the Mexican border in San Diego County. Range-wide surveys and all other relevant information indicate that the Pacific pocket mouse remains patchily distributed. Much of the habitat that may have supported the subspecies has been lost in association with large-scale development of the coastal lowlands of southern California. There are no records of Pacific pocket mouse from Los Angeles County since 1938, and the Clifton and Wilmington locales have been developed. Habitat in the Hyperion area of Marina del Rey/El Segundo has been lost to urban development. The El Segundo dunes contain the best remaining habitat in the vicinity of Marina del Rey/El Segundo, but a portion of the dunes formerly supported a residential development and this area

has reportedly been trapped extensively without success. Between 1972 and 1993, there were no records of the Pacific pocket mouse until the subspecies was rediscovered at the Dana Point Headlands in Orange County. This was the only known extant population of the subspecies at the time of its listing in 1994. Shortly thereafter, Pacific pocket mouse was discovered at three additional locations, all within the boundaries of Marine Corps Base Camp Pendleton in San Diego County. The sites include San Mateo North, on a south-facing slope in the northwestern corner of Camp Pendleton; San Mateo South, in a military training area on Camp Pendleton, approximately 3.2 kilometers (km) (2 miles [mi.]) from the coastline; and Santa Margarita, immediately north of the Santa Margarita River (USFWS 2010). Current occupied habitat for the Pacific pocket mouse is estimated to be less than 100 total acres at the three sites (Bolster 1998).

**Distinct Population Segments Defined**

No

**Critical Habitat Designated**

No;

***Life History*****Feeding Narrative**

Adult: The Pacific pocket mouse collects and stores seeds in fur-lined cheek pouches (USFWS 1998). They primarily consume seeds, selecting seeds averaging 1.4 mm (0.06 in.) in size, and occasionally feed on insects and green vegetation (USFWS 2010). They have a low bioenergetic requirement; they are nocturnal and hibernate/aestivate based on food availability and temperature (USFWS 2010). Co-occurring rodents of similar size (such as the western harvest mouse [*Reithrodontomys megalotus*]) may compete for the same seeds (USFWS 2010). An intensive field study of the dietary preferences of Pacific pocket mouse, based on fecal analysis, found that their diet included arthropods and seeds or green vegetation from California buckwheat (*Eriogonum fasciculatum*), California broom (*Lotus scoparius*), lemonadeberry (*Rhus integrifolia*), sage (*Salvia* sp.), storksbill (*Erodium* sp.), Cleveland's cryptantha (*Cryptantha clevelandii*), and grasses. They were also found to consume a higher proportion of forb seeds in the spring, and a higher proportion of grass seeds later in the year, which may relate to seasonal food availability (USFWS 2010).

**Reproduction Narrative**

Adult: Relatively little is known of the breeding biology of Pacific pocket mice. The Pacific pocket mouse breeding season begins in April and runs through July; pregnant and lactating females have been found from April through June, with immatures noted from June through September (USFWS 1998). The little pocket mouse has a gestation period of 22 to 23 days. The Pacific pocket mouse typically has one litter per year, but may have as many as two litters per year, with approximately two to eight young per litter (NatureServe 2015). The young are weaned around 30 days, and sexual maturity is reached at 41 days of age. A small number of juveniles may breed within 1 month of weaning (USFWS 1998). The sex ratio of this subspecies varies by population: at the Dana Point Headlands, the sex ratio was 1.1 to 1 (male to female); at the San Mateo Creek population, the sex ratio was 0.8 to 1 (male to female) (USFWS 1998). The lifespan is approximately 4 to 6 years in captivity. Key resources needed for reproduction include suitable habitat and an abundance of green vegetation for feeding and nesting purposes. Larger

population size and greater survivorship are positively associated with rainfall and annual plant seed availability. These factors suggest that the Pacific pocket mouse may be capable of shifting demographic strategies depending on resource availability. Under periods of high rainfall and plant production, the Pacific pocket mouse is likely to exhibit maximum reproduction and relatively low survival rates, while minimum reproduction and maximum survival rates would be expected during times of drought and poor primary production (USFWS 2010).

**Spatial Arrangements of the Population**

Adult: Clumped

**Environmental Specificity**

Adult: Narrow/specialist.

**Tolerance Ranges/Thresholds**

Adult: Low

**Site Fidelity**

Adult: Low

**Dependency on Other Individuals or Species for Habitat**

Adult: Observations on captive animals indicate that this subspecies, like other heteromyids, is aggressively solitary (USFWS 1998).

**Habitat Narrative**

Adult: The Pacific pocket mouse is commonly associated with coastal sage scrub vegetation, but has been found in a range of plant communities, including coastal strand, coastal dunes, ruderal vegetation on river alluvium, and coastal sage scrub. Within these vegetation associations, the Pacific pocket mouse is thought to prefer open, sparsely vegetated areas and small open patches in dense vegetation (USFWS 2010). The subspecies occurs on fine-grain, sandy or gravelly substrates in the immediate vicinity (within 4 km [2.5 mi.]) of the Pacific Ocean (USFWS 1998). Sandy soils may be necessary for constructing their burrows, where they cache food. Observations on captive animals indicate that this subspecies, like other heteromyids, is aggressively solitary (USFWS 1998). Key resources needed for habitat include sandy soils with open, sparsely vegetated areas and small open patches in dense vegetation. The Pacific pocket mouse has not been documented in dense nonnative grasslands, which are often associated with loam to clay soils. It is therefore suspected that the density of vegetation at ground level, along with the soil conditions, make this vegetation community unsuitable for this subspecies (USFWS 2010).

***Dispersal/Migration*****Motility/Mobility**

Adult: Low

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Moderate; Pacific pocket mice exhibit substantial individual variability in movement, with some individuals appearing to remain relatively sedentary and others making long-distance excursions of 150 meters (m) (492 feet [ft.]) or more, sometimes coinciding with a shift in use area. Males consistently are observed to have larger home ranges than females, with additional variability in movement over time and space possibly relating to breeding status, the age composition of the population, population density, and/or site conditions (USFWS 2010).

**Immigration/Emigration**

Adult: Immigrates/emigrates; may be able to quickly colonize unoccupied suitable habitat adjoining areas of occupancy, with the rate of invasion likely dependent on the density of the surrounding population (USFWS 2010).

**Dispersal/Migration Narrative**

Adult: The Pacific pocket mouse is a relatively sedentary subspecies with low mobility. It is nonmigratory, but has a moderate capacity for dispersal. Pacific pocket mice exhibit substantial individual variability in movement, with some individuals appearing to remain relatively sedentary and others making long-distance excursions of 150 m (492 ft.) or more, sometimes coinciding with a shift in use area. Males consistently are observed to have larger home ranges than females, with additional variability in movement over time and space possibly relating to breeding status, the age composition of the population, population density, and/or site conditions. They may immigrate/emigrate to quickly colonize unoccupied suitable habitat adjoining areas of occupancy, with the rate of invasion likely dependent on the density of the surrounding population. In 1993, 36 animals were captured in approximately 1.4 ha (3.5 ac.) of occupied habitat on the Dana Point Headlands. One study reported a movement (by recapture) of about 22 m (72 ft.). Most other recaptures are at the first capture location (USFWS 2010).

**Additional Life History Information**

Adult: In 1993, 36 animals were captured in approximately 1.4 hectares (ha) (3.5 acres [ac.]) of occupied habitat on the Dana Point Headlands. One study reported a movement (by recapture) of about 22 m (72 ft.). Most other recaptures are at the first capture location (USFWS 2010).

***Population Information and Trends*****Population Trends:**

Declining (NatureServe 2015)

**Species Trends:**

Declining (NatureServe 2015)

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Number of Populations:**

Four (USFWS 2010)

**Population Size:**

Fewer than 1,000 individuals (NatureServe 2015).

**Resistance to Disease:**

Moderate

**Adaptability:**

Low

**Additional Population-level Information:**

A genetic study found that, among the extant populations, San Mateo North and San Mateo South share the greatest number of genetic markers. This suggests that these populations were the most recent to be historically connected. Currently, these sites appear to be effectively isolated from one another and are separated by fallow agricultural fields and associated roads in the San Mateo floodplain, San Mateo Creek, a State Park campground, and Cristianitos Road. The Santa Margarita population, the largest of the known extant occurrences of the Pacific pocket mouse, is critical to maintenance of the subspecies because it is the only known population of appreciable size and extent where large numbers and re-colonization dynamics are likely to protect against localized extirpations (USFWS 2010).

**Population Narrative:**

Pacific pocket mouse populations are subject to wide variability. However, the overall trend has been and continues to be one of decline. Currently, there are fewer than 1,000 individuals (NatureServe 2015). A genetic study found that, among the four extant populations, San Mateo North and San Mateo South share the greatest number of genetic markers. This suggests that these populations were the most recent to be historically connected. Currently, these sites appear to be effectively isolated from one another and are separated by fallow agricultural fields and associated roads in the San Mateo floodplain, San Mateo Creek, a State Park campground, and Cristianitos Road. The Santa Margarita population, the largest of the known extant occurrences of the Pacific pocket mouse, is critical to maintenance of the subspecies because it is the only known population of appreciable size and extent where large numbers and re-colonization dynamics are likely to protect against localized extirpations (USFWS 2010).

***Threats and Stressors***

**Stressor:** Habitat destruction

**Exposure:** Urban, suburban, and agricultural uses.

**Response:** Mortality

**Consequence:** Extirpation of populations, and extinction.

**Narrative:** The conversion of native habitats resulting from urban, suburban, and agricultural development apparently is the leading cause of the large-scale destruction of Pacific pocket mouse habitat. A recent comprehensive review of the Pacific pocket mouse included considerations of the fate of confirmed (historically-occupied) Pacific pocket mouse habitat. The large majority of native habitats within the historic range of the Pacific pocket mouse in coastal

Los Angeles, Orange, and San Diego counties has been converted to urban, suburban, and agricultural uses (USFWS 1998).

**Stressor:** Habitat loss and fragmentation

**Exposure:** Development in coastal southern California.

**Response:** Removal and degradation of habitats.

**Consequence:** Smaller populations, increased risk of extinction, and increased susceptibility to stochastic events.

**Narrative:** Habitats within the historic range of the Pacific pocket mouse have been highly fragmented or degraded by highways, roads, structures, lighting, foot traffic, other human activities, and the proliferation of nonnative plant and animal species (USFWS 1998). There is a potential that undiscovered populations of the Pacific pocket mouse may persist within their historic range. Habitat loss and habitat fragmentation in coastal southern California have continued and are likely to continue in the foreseeable future. The development continues to fragment the remaining coastal habitat with the potential to support Pacific pocket mice within their historical range (USFWS 2010). Fragmented and degraded habitats support smaller populations, which are more susceptible to random extinction events (USFWS 1998).

**Stressor:** Nonnative Argentine ants (*Linepithema humile*)

**Exposure:** Argentine ants have invaded coastal sage scrub in Pacific pocket mouse habitat.

**Response:** They may displace native ants.

**Consequence:** Native ants are important for seed dispersal in habitat fragments. Argentine ants have the potential to alter ecosystem processes important for maintenance of the sage scrub vegetation community by displacing native ants (USFWS 2010).

**Narrative:** Nonnative Argentine ants have been identified as a potential threat to the Pacific pocket mouse, because of their ability to displace native ants that are important for seed dispersal in habitat fragments. This threat was identified based on the potential for Argentine ants to alter ecosystem processes important for maintenance of the sage scrub vegetation community occupied by the Pacific pocket mouse (USFWS 2010).

**Stressor:** Depredation: domestic and feral cats

**Exposure:** Occur in residential developments that are adjacent to Pacific pocket mouse populations.

**Response:** Domestic and feral cats have the ability to rapidly deplete rodent populations.

**Consequence:** May diminish remaining populations of the Pacific pocket mouse.

**Narrative:** Domestic and feral cats have been observed entering occupied Pacific pocket mouse habitat at San Mateo South. Cats pose a predatory threat to Pacific pocket mice at the San Mateo South location and other known population locations. Cats have the ability to rapidly deplete rodent populations (USFWS 2010).

**Stressor:** Small population size

**Exposure:** Pacific pocket mice can sometimes exhibit dramatic population fluctuations.

**Response:** Small populations are more likely to become extirpated due to demographic, environmental, and genetic stochastic risks.

**Consequence:** Extirpation of populations, and extinction.

**Narrative:** Pacific pocket mice can sometimes exhibit dramatic population fluctuations. Small populations are more likely to become extirpated due to demographic, environmental, and genetic stochastic risks (USFWS 2010).

**Stressor:** Military training activities

**Exposure:** Increased foot and off-road vehicle traffic, and addition of new training elements.

**Response:** Removal or reduction of vegetation, and soil compaction.

**Consequence:** Crushing of burrows, and reducing the quality of soils for constructing burrows.

**Narrative:** Military training at the Oscar One training site has caused observed impacts, including removal or reduction of vegetation, soil compaction, addition of new training elements, and increased foot and off-road vehicle traffic. Direct impacts to mice have included crushing of burrows, degradation of habitat quality by reducing vegetative cover and availability of seed resources, and reduction in the quality of soils for constructing burrows (USFWS 2010).

**Stressor:** Fire and fuel management

**Exposure:** Fire suppression at Dana Point and San Mateo North, as well as prescribed fire within the Santa Margarita Pacific at Camp Pendleton.

**Response:** Alteration of habitat and available food resources.

**Consequence:** Stress on remaining populations, contributing to population decline.

**Narrative:** Fire management practices, both for fire suppression and for prescribed fire, have impacts on the habitat of the Pacific pocket mouse. At Dana Point and San Mateo North, a lack of recent fire has resulted in a majority of these sites being dominated by mature large-stature sage scrub shrubs that overlap with one another and provide nearly continuous canopy cover over the ground. This may be reducing habitat quality for the Pacific pocket mouse by eliminating habitat openings and by suppressing growth of annual forbs and grasses that are an important source of seeds. Prescribed fire is frequently used within the range of the Santa Margarita population to prevent fires ignited by ordnance training from escaping the vicinity of the live firing ranges. It is likely that the high frequency of fires in Edson Range is suppressing the Pacific pocket mouse population (USFWS 2010).

**Stressor:** Recreational activities

**Exposure:** Public use.

**Response:** Habitat disturbance and degradation, and invasion by nonnative species.

**Consequence:** Increased extirpation risks.

**Narrative:** Recreational use of areas that contain Pacific pocket mouse populations occurs in Dana Point Headlands, San Mateo North, and San Mateo south. Use of these areas may cause habitat disturbances, including deposition of trash, creation of trails, bare areas, and compacted soils. Indirect impacts may include the invasion of these sites by nonnative annual grasses and weeds, and disturbance by nonnative animals. This may exacerbate the extirpation risks of these populations by degrading habitat quality (USFWS 2010).

**Stressor:** Existing regulatory mechanisms

**Exposure:** Populations occurring on federally owned land.

**Response:**

**Consequence:** May exacerbate the extirpation risks of these populations by degrading habitat quality.

**Narrative:** Three of the four known extant populations occur on federally owned land in Marine Corps Base Camp Pendleton. The interagency consultation requirements of the Endangered Species Act (ESA) are the primary regulatory mechanism mandating Pacific pocket mouse conservation, and are likely inadequate to provide for conservation of the Pacific pocket mouse

in the absence of the protections afforded by ESA. These impacts may exacerbate the extirpation risks of these populations by degrading habitat quality (USFWS 2010).

### ***Recovery***

#### **Reclassification Criteria:**

Recovery of the Pacific pocket mouse will likely take approximately 25 years. The U.S. Fish and Wildlife Service (USFWS) may consider reclassifying the Pacific pocket mouse to threatened status if and when:

Ten populations are independently viable and stable or increasing, and their habitats are secure (free from risk of loss) and fully protected through fee ownership by a resource agency or conservation program, conservation easement, or other means of permanent protection. Populations of Pacific pocket mice shall be considered viable if the appropriate analysis of measured population parameters indicate that each population has a 95 percent or greater chance of surviving for 100 years (USFWS 2010).

Occupied habitat consists of a minimum of 2,000 ha (4,940 ac.) that are secure and fully protected through fee ownership by a resource agency or conservation program, conservation easement, or other means of permanent protection (USFWS 2010).

All Pacific pocket mouse populations are managed through a program to maintain genetic diversity for future generations (USFWS 2010).

All Pacific pocket mouse populations and essential habitat are managed so that current and potential threats (e.g., predation and disease) are eliminated or minimized to the extent that each population is not at risk of extirpation. Essential habitat is defined to mean that habitat necessary for the full recovery of the subspecies (USFWS 2010).

#### **Delisting Criteria:**

The USFWS will consider delisting the Pacific pocket mouse if and when:

All actions necessary for reclassification to threatened have been implemented.

Any necessary protection, restoration, and enhancement activities (on all sites that have been determined to be essential to the recovery of the subspecies) are successfully completed (USFWS 1998).

Populations of the Pacific pocket mouse are representative of the full (existing) genetic variability and historical geographical range of the subspecies, and occur in habitats that collectively represent the full range of parameters observed and described in the past or during prescribed, future research and monitoring efforts (USFWS 1998).

To delist the subspecies, we must also determine that the following five factors no longer continue to adversely affect the survival and recovery of the subspecies: (1) the present or threatened modification, or curtailment, of the subspecies' habitat or range; (2) overutilization for commercial, recreational, scientific, or educational purposes; (3) disease and predation; (4) inadequacy of existing regulatory mechanisms; and (5) other human-made or natural factors

affecting the continued existence of the subspecies. A final decision relating to the delisting of the subspecies would be made only after a thorough review of all relevant information, including prescribed research (USFWS 1998).

**Recovery Actions:**

- Identify and protect all extant populations and essential habitat (USFWS 1998).
- Prepare and implement habitat management plans (USFWS 1998).
- Enhance and expand Pacific pocket mouse habitat (USFWS 1998).
- Conduct research on the life history, ecology, and population biology of the Pacific pocket mouse (USFWS 1998).
- Identify and implement measures to create additional populations (USFWS 1998).
- Enhance public awareness of and appreciation for the Pacific pocket mouse recovery program through educational and interpretive programs (USFWS 1998).
- Work with the U.S. Marine Corps to develop and implement management plans to support extant populations at San Mateo North, San Mateo South, and Santa Margarita (USFWS 2010).
- Work with the San Diego Zoological Society Institute for Conservation Research to establish a captive-bred population of Pacific pocket mouse that can be used to support translocation research and establishment of additional Pacific pocket mouse populations (USFWS 2010).
- Work with the U.S. Marine Corps to perform translocation experiments in the vicinity of the Santa Margarita Pacific pocket mouse population (USFWS 2010).
- Assess the status of the San Mateo North Pacific pocket mouse population and translocate Pacific pocket mice to that location following habitat enhancement efforts to augment or reestablish a Pacific pocket mouse population in this vicinity (USFWS 2010).
- Contact and work with landowners at identified receiver sites to obtain permission and perform environmental reviews necessary to support translocations to those sites (USFWS 2010).
- Work with the Marine Corps to reestablish functional connectivity between the San Mateo North and San Mateo South Pacific pocket mouse populations (USFWS 2010).

***Conservation Measures and Best Management Practices:***

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***Additional Threshold Information:***

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## **SPECIES ACCOUNT: *Peromyscus gossypinus allapaticola* (Key Largo cotton mouse)**

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### ***Species Taxonomic and Listing Information***

**Commonly-used Acronym:** KLCM

**Listing Status:** Endangered; Southeast Region (R4) (USFWS, 2015)

### **Physical Description**

The KLCM is an island subspecies of the cotton mouse (*P. gossypinus*), a widespread species in the southeastern United States. Schwartz (1952) described the Key Largo cotton mouse as a medium-sized mouse with large ears and protuberant eyes. The KLCM has a reddish to dusky brown back and a white underside. The body length ranges from 6.6 inches to 7.4 inches, the tail length ranges from 2.8 inches to 3.4 inches, and the hind-foot length ranges from 0.82 inch to 0.90 inch (Service 1999).

### **Current Range**

The KLCM historically inhabited all of the tropical hardwood hammock forests from the northern end of Key Largo southward to Tavernier in Plantation Key. The distribution of the KLCM is now restricted to Key Largo north of the intersection of U.S. Highway 1 and CR 905, known locally as North Key Largo (Frank et al. 1997). The Service introduced the KLCM to Lignumvitae Key in 1970. However, the last recorded sighting was in 1977 (Service 2009). The KLCM was not observed during a trapping study on Lignumvitae Key in 2007 (Greene 2007) and it appears that the population no longer exists.

### **Distinct Population Segments Defined**

No. (USFWS, 2015)

### **Critical Habitat Designated**

Yes;

### ***Life History***

#### **Feeding Narrative**

Adult: The KLCM is an herbivore, and its diet consists of leaves, buds, seeds, and fruits.

#### **Reproduction Narrative**

Adult: Cotton mice breed throughout the year, and produce two to three litters annually with a mean litter size of four. Life expectancy – The KLCM's life expectancy ranges from about 5 months to 3 years (Service 2009).

#### **Habitat Narrative**

Adult: The KLCM occurs within a variety of habitats including early successional, and mature tropical hardwood hammocks, and *Salicornia* coastal strands (Humphrey 1992). The species is also known to use recently burned areas where bracken fern (*Pteridium aquilinum*) dominates the ground cover (Goodyear 1985). The KLCM builds leaf-lined shelters in logs, tree hollows,

rock crevices, or within or near woodrat nests. The shelter entrances measures 1.2 inches to 3.5 inches in diameter, and is often partially covered with leaves or bark.

### ***Dispersal/Migration***

#### **Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory

### ***Population Information and Trends***

#### **Population Trends:**

Unknown

#### **Population Narrative:**

Because efforts to monitor the KLCM population over the last 30 years have been meager, trends in the population are difficult to ascertain. Barbour and Humphrey (1982) reported a density of 11.5 KLCM per hectare, Humphrey (1988) reported a density of 21.2 KLCM per hectare, and Frank et al. (1997) reported a density of 6.2 KLCM per hectare. Castleberry et al. (2008) conducted the most current monitoring efforts of the KLCM population in North Key Largo in 2007 and estimated a KLCM population of about 17,000 individuals with an increasing trend in the population based on live trapping conducted from November to December. The KLCM was formerly distributed throughout Key Largo, but is now restricted to tropical hardwood hammocks on North Key Largo. However, the majority of high quality hammock habitat available on North Key Largo has been protected through acquisition and is being managed for conservation by the Service and State of Florida. Because of these efforts and current land use regulations in place by Monroe County, the threat of occupied habitat loss from development on North Key Largo is low. A total of 2,498 acres of suitable KLCM habitat currently occurs in North Key Largo. About 88 percent of this acreage (2,188 acres) is protected under public ownership.

### ***Threats and Stressors***

**Stressor:** Non-native predators

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Non-native predators – A potential serious threat to the KLCM is the presence of feral and free roaming domestic cats. Cats are known to occur within the CLNWR and the Key Largo Hammocks State Botanical Site. Densities of domestic cats appear to be greater near the residential areas of North Key Largo such as Ocean Reef, Garden Cove, and the Ocean Shores developments. Cats are known to prey upon a variety of wildlife species, and studies indicate that mice often compose a large proportion of the diet (Churcher and Lawton 1989). Moreover, cats may hunt even when fed daily by humans (Liberg 1985). In addition to direct mortality, predators such as cats may also have indirect effects on prey species. The risk of predation may alter the behavior of prey species resulting in reduced growth rates and reproductive output (Arthur et al. 2004). Consequently, it is likely that cat predation is affecting the KLCM population. However, in the absence of specific studies, the effects of cat predation on the KLCM population are difficult to quantify. The Service is attempting to address the problem of free roaming cats

on North Key Largo and contracted the U.S. Department of Agriculture's Wildlife Services in 2005 to remove the cats from the CLNWR. However, because humans continue to release cats in this area, future efforts to remove cats from the area will be necessary. Other non-native predators, such as fire ants and exotic snakes, also pose a threat to the KLCM (Service 2009). The role of fire ants in the ecology of the North Key Largo hammocks is unknown. However, fire ants have substantially affected wildlife populations in other areas (Killion and Grant 1993). Because the KLCM is a ground nester, it may be vulnerable to predation by fire ants. The exotic Burmese python may also be a significant predator of the KLCM. The Service has funded a project currently being conducted by the U.S. Geological Survey (USGS) to detect and control Burmese pythons on Key Largo using visual surveys and experimental traps (Service 2008). Seven Burmese pythons have been captured in Key Largo since April 2007 (Snow 2008). Finally, black rats have also been established on Key Largo, and competition from this species may adversely affect the KLCM.

**Stressor:** Property development

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Property development – Past commercial and residential development in the Keys has reduced the extent of habitat available to the KLCM, and degraded the condition of remaining habitat. Brown (1978) and Hersh (1981) attributed the apparent extirpation of this species from Key Largo south of the U.S. Highway 1/CR 905 intersection to land clearing followed by residential and commercial development. Habitat fragmentation, combined with a decreased range, makes the KLCM more vulnerable to natural catastrophes such as hurricanes and fire (Service 1993). However, since the 1990's, the loss and degradation of cotton mouse habitat has been significantly diminished with the establishment of the Monroe County's Rate of Growth Ordinance, and land purchases for conservation by State and Federal entities.

**Stressor:** Climate change

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Climate change is also considered an important threat to the KLCM. Sternberg et al. (2007) and Su Yean Teh et Al. (2008) in their assessment of the middle and upper keys susceptibility to sea level rise concluded that tropical hardwood hammocks characteristic of the upper Florida Keys will ultimately be replaced by mangrove communities. Worst-case models by Bergh (2009) forecast an 88 percent loss in hammock vegetation within Key Largo by 2100. Consequently, in order to survive, the KLCM will likely require resource management intervention or translocation to suitable habitat outside of North Key Largo.

**Stressor:**

**Exposure:**

**Response:**

**Consequence:**

**Narrative:**

**Stressor:**

**Exposure:**

**Response:**  
**Consequence:**  
**Narrative:**

**Stressor:**  
**Exposure:**  
**Response:**  
**Consequence:**  
**Narrative:**

### ***Recovery***

#### **Delisting Criteria:**

Delisting Recovery Criteria The Key Largo cotton mouse will be considered for delisting when all the following criteria have been met: 4 1. Five (5) additional populations are established or discovered within the historical range of the species that exhibit a stable or increasing population trend for multiple generations, and natural recruitment (Factor A). 2. The five (5) new populations should be located outside of Dagny Johnson Botanical Preserve State Park and Crocodile Lake National Wildlife Refuge and be connected to the extent that genetic diversity can be naturally maintained without translocations or captive breeding (Factor A, D, E). 3. Non-native species (e.g., Burmese pythons, tegus, free-roaming pets, black rats, fire ants) are reduced or eliminated to a degree that predation and competition are low enough for KLCM to remain viable for the foreseeable future (Factor C, D). 4. When, in addition to the above criteria, it can be demonstrated that habitat loss associated with sea level rise and development are diminished such that enough suitable habitat remains for KLCM to remain viable for the foreseeable future (Factor E) (USFWS 2019).

#### ***Conservation Measures and Best Management Practices:***

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## SPECIES ACCOUNT: *Peromyscus polionotus allophrys* (Choctawhatchee beach mouse)

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### *Species Taxonomic and Listing Information*

**Commonly-used Acronym:** CBM

**Listing Status:** Endangered; 06/06/1985; Southeast Region (R4) (USFWS, 2016)

### **Physical Description**

The Choctawhatchee beach mouse is variable in coloration. Bowen (1968) described four color morphs for this subspecies, varying from orange to yellow on the back, with varying amounts of white on the face and nose. The tail of the Choctawhatchee beach mouse is relatively longer than that of any other species of Gulf coast beach mouse (USFWS, 1987).

### **Taxonomy**

Since the listing of the CBM, further research concerning the taxonomic validity of the subspecies classification of beach mice has initiated and/or conducted. Preliminary results from these studies support the separation of beach mice from inland forms, and support currently accepted (Bowen 1968) taxonomy that each beach mouse group represents a unique and isolated subspecies (USFWS, 2007).

### **Historical Range**

Originally found on Gulf coast of Florida, USA, from East Pass, Choctawhatchee Bay, Okaloosa County, to Shell Island, Bay County (NatureServe, 2015).

### **Current Range**

Range has been reduced to about 1/5 of original size. As of the late 1980s, occupied habitat amounted to 6.5 km of beach dunes at Topsail Hill and 9.4 km on Shell Island, with a low but persistent population in a translocation area at Grayton Beach State Recreation Area (Holler 1992) (NatureServe, 2015).

### **Distinct Population Segments Defined**

No

### **Critical Habitat Designated**

Yes; 6/6/1985.

### **Legal Description**

On October 12, 2006, the U.S. Fish and Wildlife Service (Service), revised critical habitat for the Perdido Key beach mouse (*Peromyscus polionotus trissyllepsis*) and Choctawhatchee beach mouse (*Peromyscus polionotus allophrys*) and designated critical habitat for the St. Andrew beach mouse (*Peromyscus polionotus peninsularis*) under the Endangered Species Act of 1973, as amended (Act). In total, approximately 6,193 acres (ac) (2,506 hectares (ha)) are designated as critical habitat for the three subspecies. The action adds approximately 135 ac (44 ha) to the amount of currently designated critical habitat for the Perdido Key beach mouse and 1,629 ac (659 ha) to the area designated for the Choctawhatchee beach mouse.

**Critical Habitat Designation**

Five units are designated as critical habitat for the Choctawhatchee beach mouse: (1) Henderson Beach, (2) Topsail Hill, (3) Grayton Beach, (4) Deer Lake, and (5) West Crooked Island/Shell Island. Table 2 summarizes areas that meet the definition of critical habitat for the Choctawhatchee beach mouse, areas excluded, and areas designated as critical habitat.

CBM-1: Henderson Beach Unit CBM-1 consists of 96 ac (39 ha) in Okaloosa County, Florida. This unit encompasses essential features of beach mouse habitat within the boundary of Henderson Beach State Park from 0.5 mi (0.8 km) east of the intersection of Highway 98 and Scenic Highway 98 to 0.25 mi (0.4 km) west of Matthew Boulevard and the area from the MHWL north to the seaward extent of the maritime forest. This westernmost unit provides primary, secondary, and scrub dune habitat (PCEs 2 and 3). This unit is within the historic range of the subspecies; however, it was not known to be occupied at the time of listing and current occupancy is unknown because no recent efforts have been made to document beach mouse presence or absence. This unit is essential to the conservation of the species because it includes protected, high-elevation scrub habitat and may serve as a refuge during storm events and as an important source population if storms extirpate or greatly reduce local populations or populations to the east. This unit is managed by the Florida Park Service. Threats specific to this unit that may require special management considerations include habitat fragmentation, artificial lighting, presence of feral cats as well as other predators at unnatural levels, and high recreational use that may result in soil compaction, damage to dunes, or other decrease in habitat quality.

CBM-2: Topsail Hill Unit CBM-2 consists of 309 ac (125 ha) in Walton County, Florida. This unit encompasses essential features of beach mouse habitat within the boundary of Topsail Hill Preserve State Park, as well as adjacent private lands from 0.1 mi (0.2 km) east of Gulf Pines to 0.6 mi (0.9 km) west of the inlet of Oyster Lake and the area from the MHWL north to the seaward extent of human development or maritime forest. This unit provides primary, secondary, and scrub dune habitat and possesses all five PCEs. Its large, contiguous, high-quality habitat allows for natural movements and population expansion. Choctawhatchee beach mice were confirmed present in the unit in 1979 (Humphrey 1992, p. 79), were present at the time of listing, and are still present. Beach mice have been captured recently on private lands within the unit, east of the park (Service 2003a, p. 2). The population of Choctawhatchee beach mice inhabiting this unit appears to harbor unique genetic variation and displays a relatively high degree of genetic divergence considering the close proximity of this population to other populations (Wooten and Holler 1999, p. 27). This unit has portions with different ownership, purposes, and mandates. Threats specific to this unit that may require special management considerations include artificial lighting, presence of feral cats as well as other predators at unnatural levels, and high recreational use that may result in soil compaction, damage to dunes, or other decrease in habitat quality. Lands containing the features essential to the conservation of the Choctawhatchee beach mouse within the area covered under the HCP for the Stallworth Preserve (4 ac (2 ha)) are excluded from critical habitat designation under section 4(b)(2) of the Act.

CBM-3: Grayton Beach Unit CBM-3 consists of 179 ac (73 ha) in Walton County, Florida. This unit encompasses essential features of beach mouse habitat within the boundary of Grayton Beach State Park, as well as adjacent private lands and inholdings, from 0.3 mi (0.5 km) west of the inlet of Alligator Lake east to 0.8 mi (1.3 km) west of Seagrove Beach and the area from the MHWL

north to the seaward extent of human development or maritime forest. This unit provides primary, secondary, and scrub dune habitat (PCEs 2 and 3), habitat connectivity (PCE 4) and is essential to the conservation of the species because it contains a population needed for recovery. This unit also provides a relatively natural light regime (PCE 5). Beach mice were not detected in the unit in 1979 (Holler 1992b, p. 79); however, they were found to be present in 1995 after Hurricane Opal (Moyers et al. 1999, p. 211). While it seems likely that beach mice were present at the time of listing (and may have been present, but not detected, in 1979), we do not have data to confirm this assumption. Therefore, we consider this unit to be unoccupied at the time of listing. A program to strengthen and reestablish the population began in 1997 and 1998 and yielded a persistent population at the State park. Further relocations, specifically in the west portion of Grayton Beach State Park, are under consideration. Recent evidence of beach mice on park lands was documented in 2004 (Service 2004c, pp. 1–4). Beach mice are also known to currently occupy the private lands immediately east of the park. This unit has portions with different ownership, purposes, and mandates. Threats specific to this unit that may require special management considerations include hurricane impacts that may require dune restoration and revegetation, excessive open, unvegetated habitat due to recreational use or storm impacts that may require revegetation, artificial lighting, presence of feral cats as well as other predators at unnatural levels, and high recreational use that may result in soil compaction, damage to dunes, or other decrease in habitat quality. Lands containing the features essential to the conservation of the Choctawhatchee beach mouse within the area covered under the HCP for Watercolor (4 ac (2 ha)) are excluded from critical habitat designation under section 4(b)(2) of the Act.

CBM–4 consists of 49 ac (20 ha) in Walton County, Florida. This unit encompasses essential features of beach mouse habitat within the boundary of Deer Lake State Park as well as adjacent private lands from approximately 1 mi (1.6 km) east of the Camp Creek Lake inlet west to approximately 0.5 mi (0.8 km) west of the inlet of Deer Lake and the area from the MHWL north to the seaward extent of maritime forest or human development. This unit provides primary, secondary, and scrub dune habitat (PCEs 2 and 3), habitat connectivity to adjacent lands (PCE 4), and is essential to the conservation of the species. This unit also provides a relatively natural light regime (PCE 5). Because live-trapping efforts in this area have been limited to incidental trapping, and beach mice were not detected in 1998 (Auburn University 1999, p. 2.10–2.11), we consider this unit to be unoccupied at the time of listing. Choctawhatchee beach mice were translocated from Topsail Hill Preserve State Park to private lands adjacent to this unit in 2003 and 2005 (Service 2003a, pp. 1–2; 2005, pp. 1–2). Tracking within the adjacent State park lands have indicated expansion of the population into the park. This unit is essential to the conservation of the species because it contains a population needed for recovery. Threats specific to this unit that may require special management considerations include artificial lighting, presence of feral cats as well as other predators at unnatural levels, and high recreational use that may result in soil compaction, damage to dunes, or other decrease in habitat quality. Lands containing the features essential to the conservation of the Choctawhatchee beach mouse within the area covered under the HCP for Watersound (71 ac (29 ha)) are excluded from critical habitat designation under section 4(b)(2) of the Act. This excluded area is 0.5 mi west of the Camp Creek Lake inlet to 0.5 mi east of the Camp Creek Lake inlet.

CBM–5: West Crooked Island/Shell Island Unit CBM–5 consists of 1,771 ac (716 ha) in Bay County, Florida. This unit encompasses essential features of beach mouse habitat within the boundaries of St. Andrew State Park mainland from 0.1 mi (0.2 km) east of Venture Boulevard

east to the entrance channel of St. Andrew Sound, Shell Island east of the entrance of St. Andrew Sound east to East Pass, and West Crooked Island southwest of East Bay and east of the entrance channel of St. Andrew Sound, and areas from the MHWL north to the seaward extent of the maritime forest. Shell Island consists of State lands, Tyndall AFB lands, and small private inholdings. Choctawhatchee beach mice were known to inhabit the majority of Shell Island in 1987 (Holler 1992b, p. 79) and were again confirmed present in 1998 (Auburn University 1999, p. 2.18), 2002, and 2003 (Service 2002a, 2003b). Because beach mice inhabited nearly the entire suitable habitat on the island less than 2 years prior to listing and were reconfirmed after listing, we consider this area to be occupied at the time of listing. The West Crooked Island population is the result of a natural expansion of the Shell Island population after the two islands became connected in 1998 and 1999, a result of Hurricanes Opal and Georges (Service 2003b). (Shell Island was connected to the mainland prior to the 1930s when a navigation inlet severed the connection on the western end.) Beach mice were documented at St. Andrew State Park mainland as late as the 1960s (Bowen 1968, pp. 86–88), though no records of survey efforts exist again until Humphrey and Barbour (1981, pp. 841–843) and Meyers (1983, p. 10) at which time beach mice were not detected. Therefore, it seems likely that this area was not occupied at the time of listing. Current beach mouse population levels at this site are unknown, and live-trapping to document the absence of mice has not been conducted. Similar to the original designation, this Park is proposed as critical habitat because it has features essential to the Choctawhatchee beach mouse. It is also within the historic range of the mouse and is essential to the conservation of the species. This unit supports the easternmost population of Choctawhatchee beach mouse, with the next known population 22 miles (mi) (35 kilometers (km)) to the west. This unit provides primary, secondary, and scrub dune habitat and possesses all five PCEs. Portions of this unit are managed by the Florida Park Service, while the remaining areas are federally (Tyndall AFB) and privately owned. Threats specific to this unit that may require special management considerations include artificial lighting, presence of feral cats as well as other predators at unnatural levels, and high residential or recreational use that may result in soil compaction, damage to dunes, or other decrease in habitat quality.

**Primary Constituent Elements/Physical or Biological Features**

Critical habitat units are designated for Okaloosa, Walton, and Bay Counties, Florida. The primary constituent elements of critical habitat for the Choctawhatchee beach mouse are the habitat components that provide:

- (i) A contiguous mosaic of primary, secondary, and scrub vegetation and dune structure, with a balanced level of competition and predation and few or no competitive or predaceous nonnative species present, that collectively provide foraging opportunities, cover, and burrow sites;
- (ii) Primary and secondary dunes, generally dominated by sea oats (*Uniola paniculata*), that despite occasional temporary impacts and reconfiguration from tropical storms and hurricanes, provide abundant food resources, burrow sites, and protection from predators;
- (iii) Scrub dunes, generally dominated by scrub oaks (*Quercus* spp.), that provide food resources and burrow sites, and provide elevated refugia during and after intense flooding due to rainfall and/or hurricane-induced storm surge;
- (iv) Functional, unobstructed habitat connections that facilitate genetic exchange, dispersal, natural exploratory movements, and re-colonization of locally extirpated areas; and

(v) A natural light regime within the coastal dune ecosystem, compatible with the nocturnal activity of beach mice, necessary for normal behavior, growth, and viability of all life stages.

### **Special Management Considerations or Protections**

Critical habitat does not include man-made structures existing on the effective date of this rule and not containing one or more of the primary constituent elements, such as buildings, aqueducts, airports, driveways, and roads, and the land on which such structures are located.

The features essential to the conservation of beach mice in all of the areas designated may require special management considerations or protections due to threats to the subspecies or its habitat. Such management considerations and protections include: Management of non-native predators and competitors, management of non-native plants, protection of beach mice and their habitat from threats by road construction, urban and commercial development, heavy machinery, and recreational activities.

### ***Life History***

#### **Feeding Narrative**

Adult: Diet likely includes fruits and seeds of dune plants, especially sea oats and sea rocket, and invertebrates when available.; Food Habits: Invertivore (Adult, Immature), Granivore (Adult, Immature) Primarily nocturnal; probably most active on moonless or cloudy nights (Natureserve, 2015).

#### **Reproduction Narrative**

Adult: Based on other subspecies, the following may apply to ALLOPHRYS as well: Produces two or more litters per year. Gestation averages 23-24 days (nonlactating) or 28-29 days (lactating). Litter size averages 3-4. Young are weaned in about 18 days. Minimum age at conception is about 4-5 weeks. Apparently monogamous mating system. See Kirkland and Layne (1989) and Holler (1992).; Populations may exhibit large fluctuations. Dispersal probably is by juveniles; adults tend to be permanent residents of their home ranges, which probably range from less than one hectare to several hectares (see Holler 1992).; (Natureserve, 2015)

#### **Spatial Arrangements of the Population**

Adult: Clumped (NatureServe, 2015)

#### **Environmental Specificity**

Adult: Narrow (inferred from NatureServe, 2015)

#### **Tolerance Ranges/Thresholds**

Adult: Low (inferred from NatureServe, 2015)

#### **Site Fidelity**

Adult: High (inferred from NatureServe, 2015)

#### **Habitat Narrative**

Adult: Coastal sand dunes (high primary and secondary, lower interior) with sparse vegetation, including sea oats, bluestem, and bunch grass (PANICUM) on the primary and secondary dunes,

and scrubby oaks, dwarfed magnolia, and rosemary on the older dunes; not all of these habitat elements are present at each occupied site (Holler 1992). Nests are in burrows. Grassland/herbaceous; Sand/dune; Shrubland/chaparral. Burrowing in or using soil (NatureServe, 2015) Narrow environmental specificity, high ecological integrity, low tolerance range and high site fidelity are inferred based on the very specific habitat this species inhabits (NatureServe, 2015).

***Dispersal/Migration*****Motility/Mobility**

Adult: High (NatureServe, 2015)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory (NatureServe, 2015)

**Dispersal**

Adult: Low (inferred from NatureServe, 2015)

**Immigration/Emigration**

Adult: Unlikely (inferred from NatureServe, 2015)

**Dispersal/Migration Narrative**

Adult: Mice are highly mobile and NatureServe (2015) notes this species is non-migratory. Low dispersal and unlikely immigration/emigration are inferred based on species habitat and the isolated nature of the known populations (NatureServe, 2015).

***Population Information and Trends*****Population Trends:**

Decreasing (NatureServe, 2015)

**Resiliency:**

Low (inferred from NatureServe, 2015)

**Representation:**

Low (inferred from NatureServe, 2015)

**Redundancy:**

Low (inferred from NatureServe, 2015)

**Number of Populations:**

1 - 5 (NatureServe, 2015)

**Population Size:**

1 - 1000 total number of individuals (NatureServe, 2015)

**Population Narrative:**

NatureServe (2015) notes that there are between 1 and 5 populations and a total of between 1 and 1000 individuals. In addition NatureServe notes that the short-term trend for this species is increasing. Low resiliency, representation and redundancy are inferred based on limited number of populations and low numbers of individuals.

### ***Threats and Stressors***

**Stressor:** Development/Construction

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Threats include human destruction of habitat, residential development in privately owned portions of designated Critical Habitat, artificial lights, vehicle use on dunes and pedestrian use (USFWS 1990). (NatureServe, 2015; USFWS, 2007)

**Stressor:** Predation (USFWS, 2007; NatureServe, 2015)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** Feral cats and non-native wildlife (raccoons, red fox, hawks, etc) are a threat to this species (USFWS, 2007; NatureServe, 2015).

**Stressor:** Storms/Hurricanes (USFWS, 2007; NatureServe, 2015)

**Exposure:**

**Response:**

**Consequence:** Loss of Habitat, loss of populations

**Narrative:** Storms (particularly hurricanes) are a threat to this species low lying habitat (USFWS, 2007; NatureServe, 2015).

### ***Recovery***

#### **Reclassification Criteria:**

The species will be considered for downlisting to threatened when there are 3 distinct, self-sustaining populations in each of the critical habitat areas, and a minimum of 50% of the critical habitat is protected and occupied by mice (USFWS, 1987).

#### **Delisting Criteria:**

The Choctawhatchee beach mouse will be considered for delisting when all the following criteria have been met: 1. Populations inhabit all five (5) critical habitat units, and three (3) additional populations inhabit Habitat Conservation Plan (HCP) covered lands that directly connect to the critical habitat units. Populations exhibit stable or increasing trends, evidenced by natural recruitment and multiple age classes (Factor A). 2. Habitat connectivity and genetic diversity shall be maintained throughout the range to a level that does not require translocations or captive breeding (Factors A and E). 3. All designated CBM critical habitat under public ownership (Federal, State, and Local entities) is managed under a conservation mechanism that addresses beach mice (Factor A). 4. Non-native predator removal (specifically free-roaming/feral cats) shall be conducted to a degree that CBM will remain viable for the foreseeable future (Factor C, D). 5. When, in addition to the above criteria, it can be demonstrated that habitat loss associated with

climate change/sea-level rise and development are diminished such that enough suitable habitat remains in the foreseeable future for CBM to remain viable (Factor E) (USFWS 2019).

**Recovery Actions:**

- Protect habitat from further human encroachment. Conduct studies to determine optimal habitat needs and life history parameter for the three subspecies. Provide habitat protection on Federal and State-owned lands. Cooperate with landowners to protect privately owned habitat. Identify unprotected habitat important to beach mice, and take actions to protect it. Monitor activities planned for privately owned land (USFWS, 1987).
- Reestablish and/or supplement populations. Conduct genetic studies to estimate both degree of inbreeding and interrelatedness of the three subspecies. Identify areas where populations have been extirpated and need to be reestablished, or where existing populations show indications of loss of genetic variability and need to be supplemented. Identify populations from which mice may be removed for translocation or captive breeding. As appropriate, based on task 23, translocate beach mice directly in to predetermined areas. As appropriate, based on task 23, develop plans for captive breeding colonies of the three subspecies (USFWS, 1987).
- Develop an educational program for the public. Provide public with information about life history and distribution of beach mice. Inform public about need for careful sanitation around dwellings to reduce beach mouse predators. Seek public support in protecting dune vegetation, and in reporting violations of laws and regulations governing use of beaches and dunes. Urge close confinement of cats in vicinity of beach mouse populations (USFWS, 1987).
- Develop emergency procedures to provide protection to beach mouse habitat in case of off-shore oil spills (USFWS, 1987).

***Conservation Measures and Best Management Practices:***

- Recovery Coordinator. A full time beach mouse recovery coordinator position should be filled. Without such a position, few of the recommendations suggested can be accomplished for CBM as well other beach mouse subspecies. Furthermore, this position would allow for Field Office coordination and consistency with permitting aspects, monitoring and trapping protocols, permit compliance, research, and recovery activities such as translocations and outreach. This would also allow for more informed and consistent guidance presented to land managers and local governments (USFWS, 2015).
- Revise Recovery Plan. The 1987 Recovery Plan should be revised and updated to reflect the current status and threats to the CBM, and recovery criteria, objectives, and tasks should be developed or revised (USFWS, 2015).
- Population and Habitat Assessment program. A monitoring program should be developed and implemented for CBM. This plan should include clear goals and objectives that the data collected would be used to achieve. Aspects of this program may include habitat mapping; obtaining demographic, landscape, or dispersal data; estimating future population trends or the likelihood of extinction; assessing management options; developing criteria for recovery; or evaluating future research priorities. A monitoring program is necessary for several other recommendations listed, particularly the Emergency Response Plan, land acquisition, translocation, and habitat management projects (USFWS, 2015).

- Emergency Response Plan: A contingency plan should be developed to outline actions taken in case of severe threats to the persistence of CBM (i.e., forecasted category 5 hurricane, feral cat population increase, population crash) (Traylor-Holzer and Lacy 2007) (USFWS, 2015).
- Land Acquisition. Appropriate parcels for land acquisition should be identified using LIDAR data (to identify high-elevation habitat) and current knowledge of CBM movements and habitat use. With this knowledge, the Service would be prepared to work with partners to acquire lands when opportunities arise (USFWS, 2015).
- Corridor size persistence, HCP studies. Research should be conducted to investigate the effectiveness of corridors currently set aside in HCPs. Studies should determine the minimum dimensions needed by CBM to ensure movement of individuals and genetic exchange through corridors (USFWS, 2015).
- Translocation. Multiple core populations of CBM are crucial for their long-term persistence. A comprehensive translocation plan is needed to identify key sites, set criteria for when translocations are needed, consider genetic as well as demographic characteristics of the donor and recipient populations, and should include a assessment of the suitability of the recipient habitat (i.e., habitat quality, have feral cats and other threats been minimized or removed). Public-private partnerships and easements should also be explored (USFWS, 2015).
- Outreach/ Education. Opportunities to convey the importance of coastal dune habitat to the public should be sought and pursued. Additional "Share the Shore" signs have been purchased by the Service and will be distributed in the summer of 2007. In addition, an outreach/education program focused on the threats feral cats pose to wildlife should also be developed (USFWS, 2015).
- Hurricane response studies. One project is underway to determine how beach mice recolonize areas after storm events. Further research should be implemented to determine the response of beach mice to storm events. This may include placing transmitters on beach mice immediately prior to a hurricane event to determine whether (or to what extent) beach mice retreat to the scrub dunes, remain in their burrows, or perish. A study to investigate the effects of revegetation and habitat modification on beach mouse habitat use and foraging patterns following storm events should be conducted (USFWS, 2015).
- Fertilization, habitat quality improvement projects. Habitat restoration projects should be developed and implemented to improve the habitat quality of areas recovering from hurricane damage. Previous studies have shown that sand fencing and application of fertilizer have yielded greater vegetative cover and greater densities of beach mice (Boyd et al. 2004). The State greenhouse project should be continued to conduct research on cultivating and to produce commercially unavailable vegetation for dune restoration of CBM habitats. Additional research on the effects of artificial lighting on beach mice should be undertaken. The research should focus on the different types of "wildlife lighting lamps" and how they affect beach mouse breeding, foraging and movement behavior and home range (USFWS, 2015).

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## SPECIES ACCOUNT: *Peromyscus polionotus ammobates* (Alabama beach mouse)

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### *Species Taxonomic and Listing Information*

**Commonly-used Acronym:** ABM

**Listing Status:** Endangered; 06/06/1985; Southeast Region (R4) (USFWS, 2016)

### **Physical Description**

The Alabama beach mouse is a pale gray with a faint dark stripe running down the upper surface of the tail; the abdomen is white. Head and body length ranges from 68 to 88 mm (2.7 to 3.4 in); tail length from 42 to 60 mm (1.6 to 2.3 in); and hind foot from 17 to 19 mm (0.6 to 0.7 in)(USFWS, 1987).

### **Taxonomy**

Since the subspecies' listing, further research concerning the taxonomic validity of beach mice subspecies has been initiated and/or conducted. Preliminary results from these studies support the separation of beach mice from inland forms, and support the currently accepted (Bowen 1968) taxonomy that each beach mouse group represents a unique and isolated subspecies (Hoekstra and Vignieri per. comm. 2006 and 2008, ITIS 2008, Van Zant 2006). Moderate levels of genetic variation, and low dispersal rates and distances are supported in Swilling and Wooten 1998, Wooten and Holler 1998, and Van Zant 2006. Van Zant 2006 also asserts that ABM populations have clusters of similar genotypes, or genetical spatial structure, that reduces the rate of genetic decay in this species (USFWS, 2009).

### **Historical Range**

Its historic range extends from the tip of the Fort Morgan Peninsula in the west to Perdido Pass and Ono Island in the east in Baldwin County, Alabama (USFWS, 2009).

### **Current Range**

Current distribution is contained within 2,450 acres of frontal, tertiary and interior scrub habitat along an estimated 13 miles of Alabama coastline (USFWS, 2009). Disjunct tracts of sand dune system from Fort Morgan eastward to the western terminus of Alabama Highway 182, including the Perdue Unit of the Bon Secour National Wildlife Refuge, Baldwin County, Alabama. Specific sites include Ft. Morgan, Gulf Highlands, Pine Beach, Gulf States Park, and Romar Beach (NatureServe, 2015). With the monitoring data received since our last 5-year review in 2009, we witnessed a slow expansion of occupied ABM habitat from 2005 to 2012 as mice began to recolonize areas from which they had become extirpated from by Hurricanes Ivan and Katrina (Danielson and Falcy 2008, Service 2008 and 2011). By 2012, ABM were found in most pre-Ivan occupied habitats (B. Lynn, USFWS pers. obs. 2012). The ABM's current distribution lies within about 2,443 acres. Natural recolonization at GSP is unlikely because it had become isolated by a man-made canal (Little Lagoon Pass)(date built unknown) and high-density development in Gulf Shores. ABM had not naturally recolonized GSP following other tropical storm events (Holliman 1983, Holler and Rave 1991, Service 2004 and 2005a, Volkert 2005). Thus, ABM were reintroduced in 2010 to GSP with 22 individuals translocated from BSNWR and FMSP. Monitoring indicates, the reintroduction has been successful and ABM occupies all available

habitat in the State park and some immediately adjacent areas in the Cities of Gulf Shores and Orange Beach (EcoSolutions, Inc. 2014, Service 2014, Volkert 2017). (USFWS 2019).

**Distinct Population Segments Defined**

No

**Critical Habitat Designated**

Yes; 6/6/1985.

**Legal Description**

On January 30, 2007, the U.S. Fish and Wildlife Service (Service), revised critical habitat for the Alabama beach mouse (*Peromyscus polionotus ammobates*) under the Endangered Species Act of 1973, as amended (Act). The revised designation encompasses approximately 1,211 acres (ac) (490 hectares (ha)) of coastal dune and scrub habitat in Baldwin County, Alabama.

**Critical Habitat Designation**

Five units are designated as critical habitat for the ABM (from to east): (1) Fort Morgan, (2) Little Point Clear, (3) Gulf Highlands, (4) Pine Beach, and (5) Gulf State Park. They are described below as our best assessment, at this time, of the areas determined to be occupied by the ABM at the time of listing that contain one or more of the PCEs that may require special management, and those additional areas that were not occupied at the time of listing, but are essential for the conservation of the ABM because they contain one or more of the PCEs, support core ABM populations and habitat continuity, and are currently occupied.

Unit 1: Fort Morgan Unit 1 (Map 2) consists of 446 ac (180 ha) and encompasses ABM habitat in the Fort Morgan State Historic Site and private lands to the east. It is located at the extreme western edge of the ABM range and consists principally of habitat that was known to be occupied at the time of listing (50 FR 23990; Holliman 1983, p. 126) south of S.R. 180 (Fort Morgan Parkway), with the exception of a single line of high scrub dunes directly north of the roadway and within the historic site boundaries. Much of Unit 1 is existing critical habitat that was designated at the time of listing (June 6, 1985; 50 FR 23885). However, the actual Fort and associated structures and developed areas that were included in the original designation are not included in this critical habitat unit. The unit extends from mean high water line (MHWL) northward to the break between scrub dune habitat and either the maritime forest or human developed landscape (for example, grassy areas associated with Fort Morgan State Historic Site). The unit is bounded to the west by Mobile Bay, and to the east by Unit 2 (western property line of the "Bay to Breakers" residential development; see below). The Dunes development and several single family homes covered by Service-approved HCPs are excluded from this unit (see "Application of Exclusions Under Section 4(b)(2) of the Act" section). ABM occurrence in the unit over time is well documented (Holliman 1983, p. 126; 50 FR 23990; Rave and Holler 1992, pp. 349–350; Sneckenberger 2001, pp. 12–13 and 32–36), and mice have been captured here following Hurricanes Ivan and Katrina (Endangered Species Consulting Services 2004a, p. 2; Service 2005, p. 15). This unit contains the features essential to the conservation of the subspecies. Some areas of the unit contain a contiguous mix of primary and secondary dunes, interdunal swales, wetlands, and scrub dunes (PCE 1), whereas other areas contain high quality primary and secondary dune habitat (PCE 2). While no one portion of the designated unit contains all PCEs, all five PCEs are present within the unit. Natural areas of the Fort Morgan Historic Site are owned by the State of Alabama (Alabama State Historical Commission), but are

currently managed by the Refuge according to a cooperative agreement (Service 2005) (see “Application of Section 3(5)(A) and Exclusions Under Section 4(b)(2) of the Act” section for further detail on management). Threats in this unit that may require special management considerations include humangenerated refuse, and degraded habitat (from activities associated with recreational use).

**Unit 2: Little Point Clear** Unit 2 consists of 268 ac (108 ha) and includes east-to-west bands of ABM habitat and connections between habitat south of the Alabama Department of Environmental Management’s Coastal Construction Control Line (CCCL) (ADEM 1995, pp. 2–8 through 2–10) and along the roadway right-of-way for Fort Morgan Parkway. This Unit is bounded to the west by Unit 1 and extends eastward to the western edge of the Surfside Shores subdivision (western boundary of Unit 3). The CCCL varies in width but generally extends about 300 ft (91 m) landward of MHWL. The Fort Morgan Parkway right-of-way, which is managed by the State of Alabama (ADCNR) extends 160 ft (49 m) both south and north of the roadway centerline. The designation includes the southern sections of right-of-way and small portions of the northern right-of-way. In several places along the east west extent of this unit, additional parcels, either to the south of the Fort Morgan Parkway, or to the north of the CCCL, that contain the PCEs (see Primary Constituent Elements section) are included in the revised designation (see Map 3). Several areas covered by HCPs for single family and duplex development have been excluded. This unit was not part of the original (1985) critical habitat designation. This unit is a mix of Federal, State, local, and private ownership. This unit, while often being inundated during storm surge events (Service 2004a; pp. 12–13; ENSR 2004, pp. 3–5 through 4–1; ACOE 2001, Service 2005a, pp. 14–15), represents the last remaining natural habitat connections between ABM populations in and around Unit 1 and Unit 3, and provides an essential link between those populations (PCE 4). Portions of this unit south of the CCCL contain PCE 2 and some sections of the right-of-way contain PCE 3. While this area was identified as being within the range of the ABM (50 FR 23872, Holliman 1983, pp. 125–126; Dawson 1983, pp. 8–11), we have no records that ABM were present at the time of listing. However, pre-hurricane Ivan trapping has verified the presence of mice south of the CCCL (Meyers 1983, pp. 5, 12–21; 50 FR 23872; Endangered Species Consulting Services 2004b, p. 2) and along the right-of-way (Sneckenberger 2001, p. 13; Farris 2003). Because the unit is presently occupied and contains two of the PCEs, and because long-term beach mouse viability depends on the existence of more populations than were documented at the time of listing, it is essential to the conservation of the subspecies. Habitat south of the CCCL consists of primary and secondary dunes, while habitat along the right-of-way consists primarily of scrub that is often temporarily disturbed by utility line maintenance. Utility line work results in a sparsely vegetated, open scrub habitat that still provides forage and cover opportunities for mice in the area.

**Unit 3: Gulf Highlands** Unit 3 consists of 275 ac (111 ha) in the central portion of the Fort Morgan Peninsula. It includes portions of the Morgantown, Surfside Shores, and Cabana Beach subdivisions, as well as portions of the Beach Club West-Gulf Highlands development, BLM properties, and some properties along the Fort Morgan Parkway right-of-way. It is bounded to the west by Unit 2. The main portion of the unit generally stretches from MHWL landward to a natural border of wetlands to the north. This portion is bisected by ABM habitat associated with the Kiva Dunes, Plantation Palms, Beach Club, and Martinique developments and is excluded because of its HCPs (see “Application of Exclusions Under Section 4(b)(2) of the Act” section). The unit also contains an eastward continuation of ABM habitat adjacent to the Fort Morgan Parkway. This northern portion of Unit 3 is bounded to the west by Unit 2 and to the east by

wetlands and maritime forest along the S.R. 180 and points east. Like the right-of-way corridor in Unit 2, it generally extends from the centerline of Fort Morgan Parkway 160 ft (49 m) south though a few areas of habitat north of the road are also captured. Unit 3 serves as an expansion, to encompass scrub habitat, of critical habitat Zone 2 that was designated at the time of listing (50 FR 23872; June 6, 1985). This unit contains the features essential to the conservation of the subspecies; all five PCEs are present in varying amounts throughout this unit. This unit, combined with the neighboring Perdue Unit of the Refuge and several properties with conservation plans that are being excluded (see “Application of Exclusions Under Section 4(b)(2) of the Act” section), contains the largest assemblage of high elevation habitat within the range of the ABM (ACOE 2001, Plate 2–11; ENSR 2004, pp. 3–5 through 4–1; Service 2004a, pp. 9–12; Service 2004b, p. 6; Service 2005a, pp. 2–4). The largest tracts of contiguous habitat possessing a full gradient of ABM habitat (primary dunes landward to scrub dunes) are also found here. ABM occupancy is well documented both at the time of listing (Meyers 1983, pp. 5, 12–21; Holliman 1983, pp. 125– 126) and recently (Endangered Species Consulting Services, LLC and ENSR Corporation 2001, p. 22; Farris 2003). ABM were found here following Hurricane Ivan (Endangered Species Consulting Services 2004, p. 2; 2004d, p. 2). Threats that may require special management include habitat degradation and fragmentation, extensive recreational pressure, poststorm cleanups, artificial lighting, predation, and human-generated refuse.

**Unit 4: Pine Beach** This unit consists of 30 ac (12 ha) including a BLM property and 27 private inholdings within the Perdue Unit of the Refuge that are not managed under the Refuge’s Comprehensive Conservation Plan. The primary and secondary dunes within this unit were part of “Zone 2” of the original critical habitat designation, which extended from the mean high tide line of the Gulf of Mexico landward 500 ft (152 m). ABM are well documented from the area both recently (Rave and Holler 1992, pp. 349–350; Swilling et al. 1998, pp. 289– 294; Sneckenberger 2001, pp. 66–69; Service 2003, p. 1) and from the time of listing (Holliman 1983, p. 126; Meyers 1983, pp. 5, 12–21). This unit, along with adjacent Refuge lands and exclusions for single family homes covered by Service-approved HCPs (see “Application of Exclusions Under Section 4(b)(2) of the Act” section), contains the features essential to the conservation of the ABM because of its high elevation habitat and continuity between habitat types. It contains PCEs 2, 3, and 5, and when combined with the surrounding Refuge lands, it also includes PCEs 1 and 4. Threats that may require special management considerations on this unit may include artificial lighting from residences, human-generated refuse that may attract predators, habitat fragmentation from the design and construction of properties (and access routes) to inholdings, and primary and secondary dunefields impacted from recent storm events.

**Unit 5: Gulf State Park** Unit 5 consists of 192 ac (78 ha) of ABM habitat in Gulf State Park, immediately east of the City of Gulf Shores and west of the City of Orange Beach. This unit retains most critical habitat designated in the 1985 listing rule (Zone 3—all primary and secondary dunes south of State Route 182) (June 6, 1985; 50 FR 23872) and adds approximately 30 ac (12 ha) of scrub habitat located directly north of S.R. 182. It extends from MHWL northward to a natural boundary consisting of brackish wetlands and maritime forest. ABM habitat covered under the 2004 HCP and subsequent HCP–ITP modifications is excluded from the designation (see “Application of Exclusions Under Section 4(b)(2) of the Act” section). This unit contains a mix of scrub and primary and secondary dune habitat, and represents the last remaining sizable block of habitat on the eastern portion of the historic range of the subspecies. ABM were documented from the Park in the late 1960s (Linzey 1970, p. 81), but were presumed extirpated by the early 1980s (Holliman 1983, pp. 123– 126; Holler and Rave 1991, p. 22–25),

because of habitat isolation combined with the effects of tropical storms, predation (primarily from feral cats), and competition with house mice. This area was referred to as occupied in our final listing rule (June 6, 1985; 50 FR 23872). ABM were reintroduced to the park in 1998, and subsequent trapping confirmed their presence there (Sneckenberger S., Service, personal communication, 2005; Service 2003, p. 2). This unit was heavily impacted by Hurricane Ivan in 2004 (Service 2004a, pp. 5–6) and Hurricane Katrina in 2005 (Service 2005a, pp. 6–9), and recent trapping has not located mice (Volkert 2005, pp. 2–5). This unit contains PCEs 2 and 3 and, therefore, possesses the habitat features essential to the conservation of the subspecies. Because this unit contains several PCEs, because it is presently occupied, and because ABM recovery depends on more populations than were documented at the time of listing, it is essential to the conservation of the subspecies. This unit is State-owned and managed by the State Parks Division of the ADCNR. It has pressures from heavy recreational use and ABM habitat here has been severely impacted by recent hurricanes. Threats to ABM habitat include loss of dune topography and vegetation from habitat destruction, human-generated refuse that could attract predators, and artificial lighting. Habitat fragmentation also threatens ABM within this unit.

#### **Primary Constituent Elements/Physical or Biological Features**

Critical habitat units are designated for Baldwin County, Alabama. The primary constituent elements of critical habitat for the Alabama Beach Mouse are the habitat components that provide:

- (i) A contiguous mosaic of primary, secondary, and scrub vegetation and dune structure, with a balanced level of competition and predation and few or no competitive or predaceous nonnative species present, that collectively provide foraging opportunities, cover, and burrow sites.
- (ii) Secondary dunes, generally dominated by sea oats (*Uniola paniculata*), that despite occasional temporary impacts and reconfiguration from tropical storms and hurricanes, provide abundant food resources, burrow sites, and protection from predators.
- (iii) Scrub dunes, generally dominated by scrub oaks (*Quercus* spp.), that provide food resources and burrow sites, and provide elevated refugia during and after intense flooding due to rainfall and/or hurricane-induced storm surge.
- (iv) Unobstructed habitat connections that facilitate genetic exchange, dispersal, natural exploratory movements, and recolonization of locally extirpated areas.
- (v) A natural light regime within the coastal dune ecosystem, compatible with the nocturnal activity of beach mice, necessary for normal behavior, growth, and viability of all life stages.

#### **Special Management Considerations or Protections**

Critical habitat does not include manmade structures (such as buildings, aqueducts, airport runways, roads, other paved areas, and piers) and the land on which they are located existing within the legal boundaries on the effective date of this rule.

The features we are designating may require special management considerations or protections due to threats to the subspecies or its habitat. Such management considerations and protections include: management of nonnative predators and competitors, management of nonnative plants,

and protection of ABM and their habitat from threats by road construction, urban and commercial development, heavy machinery, and recreational activities.

### ***Life History***

#### **Feeding Narrative**

Adult: Eats fruits and seeds of dune plants, especially sea oats and sea rocket; feeds on invertebrates when seeds scarce (Matthews and Moseley 1990).; Food Habits: Invertivore (Adult, Immature), Granivore (Adult, Immature) Primarily nocturnal.; (Natureserve, 2015)

#### **Reproduction Narrative**

Adult: Breeds year round, but the percentage of reproductively active females is highest in fall and winter (71-84%) and lowest in summer (36%) (Rave and Holler 1992). Produces 2 or more litters per year. Gestation averages 23-24 days (nonlactating) or 28-29 days (lactating). Litter size averages 3-4 (A88USF04NA). Young weaned in about 18 days. Minimum age at conception 5 weeks. Very few live longer than 12 months (Rave and Holler 1992). Apparently monogamous mating system (B89KIR01NA).; Populations fluctuate seasonally and annually (Holler 1992); most abundant in winter and spring (Rave and Holler 1992). See Swilling et al. (1998) for information on population dynamics following Hurricane Opa (Natureserve, 2015). USFWS (2007) notes that the average lifespan is 9 months with some individuals living 12-20 months.

#### **Environmental Specificity**

Adult: Narrow/specialist (NatureServe, 2015)

#### **Tolerance Ranges/Thresholds**

Adult: Low (inferred from NatureServe, 2015)

#### **Site Fidelity**

Adult: High (inferred from NatureServe, 2015)

#### **Habitat Narrative**

Adult: Favors dunes with grass/shrub cover: primary dunes, interdune areas, secondary dunes, and scrub dunes; sites with all of these components are optimal (Matthews and Moseley 1990). In fact, although highest densities occur within primary dunes, scrub dunes are probably essential for the long-term survival of these mice since they provide higher elevation refuges during major storm events (USFWS 2000). Occupies underground burrows when inactive or rearing young; entrances are in clumps of grass or beneath sheltering vegetation (Matthews and Moseley 1990). Grassland/herbaceous; Sand/dune; Shrubland/chaparral Burrowing in or using soil (Natureserve, 2015). Low tolerance range and high site fidelity are inferred based on species specific habitat needs of this species and corresponding low number of population.

### ***Dispersal/Migration***

#### **Motility/Mobility**

Adult: High (Natureserve, 2015)

#### **Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migrant (Natureserve, 2015)

**Dispersal**

Adult: Low (inferred from Natureserve, 2015)

**Immigration/Emigration**

Adult: Unlikely (inferred from NatureServe, 2015)

**Dispersal/Migration Narrative**

Adult: Based on recaptures in traps, mean home range size was 3,586 square meters; mean dispersal distance of subadults was 160 m (= 2.4 home range diameters), but a significant number of mice dispersed more than 5 home range diameters (Swilling and Wooten 2002). Trapping data likely underestimate dispersal distance. Nonmigrant: N; Local migrant: N; Distant migrant: N; (Natureserve, 2015). Low dispersal and unlikely immigration and emigration are inferred based on low number of populations and specific habitat needs (NatureServe, 2015).

***Population Information and Trends*****Population Trends:**

Increasing (NatureServe, 2015)

**Species Trends:**

Improving; ABM habitat continues to recover following the devastating hurricanes of 2004 (Ivan) and 2005 (Katrina). These hurricanes destroyed 90-95% of the frontal (primary and secondary) dune habitat (Service 2011). Hurricane Isaac (2012) and Nate (2017) had some impact on frontal dunes, but any effects to beach mouse populations as a result of this storm have not been documented. Based on live trapping data, track tube data, site visits and personal observations, we have determined the overall species status as "improving" because the ABM appears to be at pre-2004 hurricane levels of population and habitat occupancy (USFWS 2019).

**Resiliency:**

Low (inferred from NatureServe, 2015)

**Representation:**

Low (inferred from NatureServe, 2015)

**Redundancy:**

Low (inferred from NatureServe, 2015)

**Number of Populations:**

1 - 5 populations (NatureServe, 2015)

**Population Size:**

1 - 1000 total individuals (Natureserve, 2015)

**Population Narrative:**

NatureServe (2015) notes that the short-term population trend is increasing, the number of populations is one to five and the total number of individuals is estimated at between 1 and

1,000. Low resiliency, representation and redundancy are inferred based on low number of populations and specific habitat requirements (NatureServe, 2015).

### ***Threats and Stressors***

**Stressor:** Human Activity/Construction (USFWS, 2009)

**Exposure:**

**Response:**

**Consequence:** Habitat Loss/Fragmentation

**Narrative:** Habitat has been lost as a result of construction activities and recreational use. Threats include competition from house mice (MUS) and predation by cats (see Rave and Holler 1992). (NatureServe, 2015; USFWS, 2009). Artificial lighting is also an issue, as is vehicle use on beaches and pedestrian traffic.

**Stressor:** Predation

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Predation by feral and loose housecats is a major threat to this species along with predation by non-native wildlife (raccoons, red fox, hawks, etc) (USFWS, 2009; NatureServe, 2015).

**Stressor:** Storms/Hurricanes (USFWS, 2009)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Storms and hurricanes are a major threat to this species low lying/coastal habitat (USFWS, 2009).

**Stressor:** Competitors

**Exposure:** Native hispid cotton rats, house mice (*Mus musculus*), Norway rats (*Rattus norvegicus*), and black rats (*Rattus rattus*)

**Response:** Mortality.

**Consequence:** Population decline.

**Narrative:** Conservation measures for reducing impacts from invasive species are routinely included in Endangered Species Act consultations on the Fort Morgan Peninsula. Some of these measures include: discouraging use of hay or pine straw bales in landscaping or habitat restoration that spread fire ants and non-native plants, equipment used in ground-disturbing activities in cogon grass areas must be washed off-site before returning, prohibiting outdoor cats, and requiring rodent-proof garbage receptacles. These conservation measures help reduce new invasions; however, invasive species may need direct intervention to reduce existing impacts (USFWS 2019).

**Stressor:** Climate change

**Exposure:** Sea level rise

**Response:** May alter habitat

**Consequence:** Population decline

**Narrative:** Efforts to relate sea level rise with beach loss along Alabama's coast have been attempted by the Service, resulting in estimates up to 1 m of beach inundated for every 1 cm rise in sea level (i.e., 1 inch sea level rise ~ 8.3 ft of beach width lost by inundation (Service files). Therefore, we are concerned global climate change and sea level rise could have adverse effects on coastal ecosystems and their associated wildlife populations, including ABM. About half of the 55-mile open-water shoreline along southern Alabama has been receding 2-5 feet per year in recent decades (Bush et al. 2001). The receding shoreline appears to be a physical response to a combination of natural events and human-caused activities such as tropical storm erosion, inland erosion, development practices, sea level rise, and basic barrier island dynamics. The rate of shoreline retreat from sea level rise is considered a function of the slope of the inundated land and the rate of sea-level rise. In coastal areas with gentle slopes, a very small increase in sea level would cause more substantial island migration (Bush et al. 2001). Estimates of sea level rise along the Gulf coast range between 38 and 60 cm (15 and 24 in) during the next century (Titus and Narayanan 1995 and 1996, Wigley 1999, Davenport 2007). However, such implications for coastal change are far from clear and would likely be influenced by a number of locally varying factors, such as slope, elevation and underlying structure of the shoreline, sand availability and transport, erosion rate, and storm frequency, duration and magnitude (Emanuel 2005, Trenberth 2005, Webster et al. 2005, and Landsea 2005) (USFWS 2019).

## **Recovery**

### **Reclassification Criteria:**

The species will be considered for downlisting to threatened when there are 3 distinct, self-sustaining populations in each of the critical habitat areas, and a minimum of 50% of the critical habitat is protected and occupied by mice (USFWS, 1987).

### **Delisting Criteria:**

The ABM will be considered for delisting when the following criteria are met: 1. The existing two (2) ABM populations exhibit stable or increasing trends, evidenced by natural recruitment and multiple age classes (Factor A, C, D, E). 2. Habitat connectivity and genetic diversity shall be maintained to a level that does not require translocations, or captive breeding (Factor A, C, D, E). 3. A mosaic of suitable habitat consisting of primary, secondary, tertiary, and interior scrub dunes is created, protected, and managed as needed for the species to remain viable for the foreseeable future (Factor E). 4. When in addition to the above criteria, it can be demonstrated that habitat loss associated with sea-level rise and development are diminished such that enough suitable habitat remains in the foreseeable future for ABM to remain viable (Factor E) (USFWS 2019b).

### **Recovery Actions:**

- Protect habitat from further human encroachment. Conduct studies to determine optimal habitat needs and life history parameter for the three subspecies. Provide habitat protection on Federal and State-owned lands. Cooperate with landowners to protect privately owned habitat. Identify unprotected habitat important to beach mice, and take actions to protect it. Monitor activities planned for privately owned land (USFWS, 1987).
- Reestablish and/or supplement populations. Conduct genetic studies to estimate both degree of inbreeding and interrelatedness of the three subspecies. Identify areas where populations have been extirpated and need to be reestablished, or where existing populations show indications of loss of genetic variability and need to be supplemented.

- Identify populations from which mice may be removed for translocation or captive breeding. As appropriate, based on task 23, translocate beach mice directly in to predetermined areas. As appropriate, based on task 23, develop plans for captive breeding colonies of the three subspecies (USFWS, 1987).
- As part of the revision of the 1987 Recovery Plan, a contingency plan should be developed to outline actions taken in case of severe threats to the persistence of ABM (e.g., Category 4-5 hurricanes). This emergency response plan should be developed with the aid of the captive breeding feasibility workshop's findings (i.e., temporary emergency action if large storm forecasted and population deemed at serious risk) (CBSG 2007). Currently, supplemental feeding of ABM under extreme circumstances (e.g., major loss of forage due to storm surge and/or salt spray) will be considered. Large scale efforts for frontal dune restoration after a major hurricane landfall similar to the recent Phase I Alabama Dune Restoration Project or the similar dune restoration funding Congress provided after the 2004 and 2005 hurricane season should be developed (<http://www.gulfspillrestoration.noaa.gov/sites/default/files/wp-content/uploads/2012/04/AlabamaDuneRestorationF.pdf>). An oil spill contingency plan was developed based on the recent 2010 Deepwater Horizon Event (USFWS 2019).
  - Appropriate parcels for land acquisition should be identified using LIDAR data and storm surge models (for high-elevation habitat identification) and current knowledge of ABM movements and habitat use (e.g., lands at Fort Morgan that are being leased to the Service). Land could potentially be purchased through a variety of means, if appropriate during the fiscal year, including section 6 land acquisition grants, the State of Alabama's Forever Wild program, or through the GCP in-lieu fee mitigation program (USFWS 2019).
  - Opportunities to convey the importance of coastal dune habitat to the public should be continued and expanded. Outreach should focus on the larger coastal ecosystem and role of the beach mouse in this ecosystem, instead of adopting a single-species focus. Efforts should stress the importance of healthy environments for both people (through the protection of infrastructure and aesthetics) and beach mice. In addition, an outreach/education program focused on the threats that feral cats pose to wildlife should also be developed (USFWS 2019).
  - Develop methods for estimating ABM population parameters in scrub and beach dune habitats with various levels of human development. In addition, conduct research to determine dispersal potentials between local populations in beach/scrub habitats and in response to tropical cyclone events. Research objectives are to quantify the relative importance of various habitats to ABM, and identify the habitat parameters or conditions necessary for ABM persistence and movement between habitat patches. Test methods to improve or create ABM habitats, particularly in scrub dunes, and document responses by invasive species such as cogon grass. The ability to create habitat could increase the quantity or quality of existing habitat, particularly high elevation habitat. For example, Gulf State Park is highly susceptible to storm surge effect because the park lacks high elevation habitat. Studies to determine if high elevation habitat can be created would increase the chances of ABM persisting at this site. Danielson and Falcy (2008) suggested that: (1) "preemptive" (i.e., pre-hurricane event) habitat management efforts in scrub may be more beneficial to local population viability than "post-facto" (i.e., post-hurricane event) management efforts in frontal dunes habitat; and (2) cotton rats appear to outcompete beach mice in some microhabitats which may be important during post-hurricane periods. These two issues should be explored further. Conduct research to determine whether or not

diseases and/or parasites are significant threats to ABM and if wet/dry weather patterns are a factor in ABM population trends (USFWS 2019).

- The development of a conservation strategy will identify baseline conditions, potential impacts, expected species responses, conservation objectives, and management options for the conservation (including long-term survival and recovery) of the ABM. Management options in the Strategy would contribute to the overall goal of protecting and improving ABM habitats and movement corridors to provide adequate feeding, breeding and sheltering needs across its range. Maintaining adequate numbers, genetic diversity, and distributions within core ABM populations (e.g., Ft. Morgan, Perdue Unit, and eventually GSP) will allow the species to persist over the long-term and core populations to recover from stochastic events (e.g., hurricanes, flooding, disease) (USFWS 2019)
- Habitat restoration projects should continue to be developed and implemented to improve the habitat quality of areas recovering from hurricane damage. Boyd et al. 2004 showed that sand fencing and application of fertilizer have yielded greater vegetative cover and greater densities of beach mice (Boyd et al. 2004). Recent dune restoration research suggests there is no benefit to using sand fencing or fertilizer in addition to vegetative cover if planted at the proper time of year (Debbie Miller per. comm. 2009). Following Hurricane Ivan (2004) and the 2005 Hurricane Season, the Service was successful in securing emergency habitat restoration funds from Congress. These funds were used to re-establish dunes at GSP and on the BSNWR. They were also used to establish a cooperative agreement with the Baldwin County Soil and Water Conservation District to restore ABM habitat on private lands on a cost-share basis. Thus far, this program has assisted over 100 coastal landowners, many of which are along the Fort Morgan Peninsula. Such efforts are of paramount importance to ABM recovery and generating public support for ABM conservation efforts and should be continued (USFWS 2019).

***Conservation Measures and Best Management Practices:***

- **Revise the 1987 ABM Recovery Plan:** The revision of the 1987 Recovery Plan should be completed to reflect the current status and threats to the ABM, and measurable recovery criteria, objectives and tasks should be developed (USFWS, 2009).
- **Emergency Response Plan:** A contingency plan should be developed to outline actions taken in case of severe threats to the persistence of ABM (e.g., Category 4-5 hurricanes). This emergency response plan should be developed with the aid of the captive breeding feasibility workshop's findings (i.e., temporary emergency action if large storm forecasted and population deemed at serious risk) (CBSG 2007). Supplemental feeding of ABM under extreme circumstances (e.g., major loss of forage due to storm surge and/or salt spray) should be considered (USFWS, 2009).
- **Land Acquisition:** Appropriate parcels for land acquisition should be identified using LIDAR data and storm surge models (for high-elevation habitat identification) and current knowledge of ABM movements and habitat use (e.g., lands at Fort Morgan that are being leased to the Service). Land could potentially be purchased through a variety of means, including section 6 land acquisition grants, the State of Alabama's Forever Wild program, or through an in-lieu fee mitigation program (USFWS, 2009).
- **Outreach/ Education:** Opportunities to convey the importance of coastal dune habitat to the public should be continued and expanded. Outreach should focus on the larger coastal ecosystem and role of the beach mouse in this ecosystem, instead of adopting a single-species focus. Efforts should stress the importance of healthy environments for both people (through the protection of

infrastructure and aesthetics) and beach mice. In addition, an outreach/education program focused on the threats that feral cats pose to wildlife should also be developed (USFWS, 2009).

- **Additional Research: Corridor Size, Persistence, Habitat Values, and Habitat Mitigation/Enhancement:** Develop methods for estimating ABM population parameters in scrub and beach dune habitats with various levels of human development. In addition, conduct research to determine dispersal potentials between local populations in beach/scrub habitats and in response to tropical cyclone events. Research objectives are to quantify the relative importance of various habitats to ABM, and identify the habitat parameters or conditions necessary for ABM persistence and movement between habitat patches. Test methods to improve or create ABM habitats, particularly in scrub dunes, and document responses by invasive species such as cogon grass. The ability to create habitat could increase the quantity or quality of existing habitat, particularly high elevation habitat. For example, Gulf State Park is the only remaining public parcel within ABM habitat where the mice no longer persist; however, the site lacks high elevation habitat. Studies to determine if high elevation habitat can be created would increase the chances of ABM persisting at this site. Danielson and Falcy (2008) suggested that: (1) “preemptive” (i.e., pre-hurricane event) habitat management efforts in scrub may be more beneficial to local population viability than “post-facto” (i.e., post-hurricane event) management efforts in frontal dunes habitat; and (2) cotton rats appear to outcompete beach mice in some microhabitats which may be important during post-hurricane periods. These two issues should be explored further. Conduct research to determine whether or not diseases and/or parasites are significant threats to ABM and if wet/dry weather patterns are a factor in ABM population trends (USFWS, 2009).
- **Develop an Overarching Conservation Strategy for the ABM:** The subspecies is restricted to suitable areas within about 2,450 acres of coastal habitat, and there are presently no acceptable options for mitigation. The development of a conservation strategy will identify baseline conditions, potential impacts, expected species responses, conservation objectives, and management options for the conservation (including long-term survival and recovery) of the ABM. Management options in the Strategy would contribute to the overall goal of protecting and improving ABM habitats and movement corridors to provide adequate feeding, breeding and sheltering needs across its range. Maintaining adequate numbers, genetic diversity, and distributions within core ABM populations (e.g., Ft. Morgan, Perdue Unit, and eventually GSP) will allow the species to persist over the long-term and core populations to recover from stochastic events (e.g., hurricanes, flooding, disease) (USFWS, 2009).
- **Re-establishment of Sustainable ABM Population at Gulf State Park:** GSP, west of Perdido Pass, is at the easternmost extent of the ABM’s range. This local population is small and isolated, and has been extirpated three times in the last three decades, most recently by Hurricane Ivan in 2004 (Holliman 1983, Holler and Rave 1991, Service 2004 and 2005a, Volkert 2005). Nonetheless, GSP is important to the conservation of the subspecies by helping establish multiple local populations of ABM over a wider range which is crucial for the subspecies long-term persistence (Shaffer and Stein 2000, Oli et al. 2001, Danielson 2005). For example, in 1986, the last remaining population of Perdido Key beach mice was located in Gulf State Park east of Perdido Pass in Alabama. Following translocation 1986-1988 to Gulf Islands National Seashore, this source population at GSP was subsequently lost following Hurricane Opal in 1995. Plans should be developed to translocate ABM to GSP in conjunction with control of feral cats and other threats. However, if ABMs are translocated to GSP in the future, it should be recognized that it is unlikely they would survive after hurricanes until sufficient high elevation storm refugia becomes available at this location (USFWS, 2009).
- **Fertilization, habitat quality improvement projects:** Habitat restoration projects should continue to be developed and implemented to improve the habitat quality of areas recovering from hurricane

damage. Boyd et al. 2004 showed that sand fencing and application of fertilizer have yielded greater vegetative cover and greater densities of beach mice (Boyd et al. 2004). Recent dune restoration research suggests there is no benefit to using sand fencing or fertilizer in addition to vegetative cover if planted at the proper time of year (Debbie Miller per. comm. 2009). Following Hurricane Ivan (2004) and the 2005 Hurricane Season, the Service was successful in securing emergency habitat restoration funds from Congress. These funds were used to re-establish dunes at GSP and on the BSNWR. They were also used to establish a cooperative agreement with the Baldwin County Soil and Water Conservation District to restore ABM habitat on private lands on a cost-share basis. Thus far, this program has assisted over 100 coastal landowners, many of which are along the Fort Morgan Peninsula. Such efforts are of paramount importance to ABM recovery and generating public support for ABM conservation efforts and should be continued (USFWS, 2009).

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## SPECIES ACCOUNT: *Peromyscus polionotus niveiventris* (Southeastern beach mouse)

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### *Species Taxonomic and Listing Information*

**Commonly-used Acronym:** SEBM

**Listing Status:** Endangered; 05/12/1989; Southeast Region (R4) (USFWS, 2016)

### **Physical Description**

A large subspecies relative to other forms of the polionotus complex. Ten adult southeastern beach mice averaged 139 mm (5.42 in) in total length and 52 mm (2.03 in) in tail length (Osgood 1909). The southeastern beach mouse is slightly darker than the Anastasia Island beach mouse, but paler than inland populations of *P. polionotus* (USFWS, 1993).

### **Taxonomy**

The southeastern beach mouse was described by Chapman (1889) as *Hesperomys niveiventris*. The type locality is Oak Lodge, opposite Micco, Brevard County, Florida. Bangs (1898) subsequently placed it in the genus *Peromyscus*, and Osgood (1909) relegated it to subspecific rank under *P. polionotus* (USFWS, 1993).

### **Historical Range**

Originally occurred on the Atlantic coast of Florida, USA, from Ponce Inlet, Volusia County, south to Hollywood Beach, Broward County (NatureServe, 2015).

### **Current Range**

Presently known from six sites in Brevard, Indian River, and St. Lucie Counties (NatureServe, 2015).

### **Distinct Population Segments Defined**

No

### **Critical Habitat Designated**

No;

### **Life History**

#### **Feeding Narrative**

Adult: Eats fruits and seeds of dune plants; feeds on invertebrates when seeds scarce (Matthews and Moseley 1990). Primarily nocturnal.; (NatureServe, 2015)

#### **Reproduction Narrative**

Adult: May breed all year. Much breeding activity November-January. On Merritt Island, the proportion of females lactating peaked in August (61%) and again in January (64%), with a major decline in September (25%); few lactating females were found from mid-winter through mid-summer (Stout 1992). Produces two or more litters per year. Gestation averages 23-24 days (nonlactating) or 28-29 days (lactating). Litter size averages 3-4 (USFWS 1988). Young are

weaned in about 18 days. Minimum age at conception is five weeks. Apparently has a monogamous mating system (Kirkland and Layne 1989).; On Merritt Island, maximum density was 64/ha (March-April); lower densities have been recorded elsewhere (Stout 1992).; (NatureServe, 2015)

**Tolerance Ranges/Thresholds**

Adult: Low (inferred from NatureServe, 2015)

**Site Fidelity**

Adult: High (inferred from NatureServe, 2015)

**Habitat Narrative**

Adult: Sea oats zone and associated dune systems with grasses, open sandy areas, and scattered shrubs; coastal strand and coastal scrub (e.g., oak-rosemary-saw palmetto) on the Canaveral Peninsula (Stout 1992). On Cape Canaveral, flourishes on powerline rights-of-way subjected to brush hogging of weedy plants (Stout 1992). Rare or absent in areas dominated by woody vegetation more than 2 m in height (Stout 1992). Nests in underground burrows. Burrowing in or using soil (NatureServe, 2015). Ecological integrity of the population and tolerance ranges are inferred based on specific habitat requirements (NatureServe, 2015).

**Dispersal/Migration****Motility/Mobility**

Larvae: Nonmigrant: Y; Local migrant: N; Distant migrant: N; (NatureServe, 2015)

Adult: High (NatureServe, 2015)

**Migratory vs Non-migratory vs Seasonal Movements**

Larvae: Nonmigrant: Y; Local migrant: N; Distant migrant: N; (NatureServe, 2015)

Adult: Non-migrant (NatureServe, 2015)

**Dispersal**

Larvae: Nonmigrant: Y; Local migrant: N; Distant migrant: N; (NatureServe, 2015)

Adult: Low (inferred from NatureServe, 2015)

**Immigration/Emigration**

Adult: Unlikely (inferred from NatureServe, 2015)

**Dispersal/Migration Narrative**

Larvae: Nonmigrant: Y; Local migrant: N; Distant migrant: N; (NatureServe, 2015)

Adult: Mice are highly mobile by nature and this species is non-migratory (NatureServe, 2015). Low dispersal and Unlikely immigration/emigration are inferred based on the low number of populations and habitat specificity.

**Additional Life History Information**

Larvae: Nonmigrant: Y; Local migrant: N; Distant migrant: N; (NatureServe, 2015)

### ***Population Information and Trends***

#### **Population Trends:**

Decreasing (NatureServe, 2015)

#### **Resiliency:**

Low (Inferred from NatureServe, 2015)

#### **Representation:**

Low (Inferred from NatureServe, 2015)

#### **Redundancy:**

Low (Inferred from NatureServe, 2015)

#### **Number of Populations:**

1-20 (NatureServe, 2015)

#### **Population Size:**

1 - 1000 individuals (NatureServe, 2015)

#### **Population Narrative:**

NatureServe (2015) notes that the short-term population trend is declining from 10-30%. In addition, NatureServe notes that there are between 1-20 populations and 1-1,000 individuals. Low resiliency, representation and redundancy are inferred based on low populations and habitat specificity (NatureServe, 2015)

### ***Threats and Stressors***

**Stressor:** Predators (NatureServe, 2015; USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** Predation by cats and other non-native wild animals is known to be a threat to this species (NatureServe, 2015; USFWS, 1993)

**Stressor:** Human encroachment (NatureServe, 2015; USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Development, public use, artificial lighting and vehicle use are some of the known threats (NatureServe, 2015; USFWS, 1993).

**Stressor:** Competition (NatureServe, 2015; USFWS, 2007)

**Exposure:**

**Response:**

**Consequence:** Loss of food source/decreased numbers

**Narrative:** Competition by other Mus species is thought to be a threat to this species

**Stressor:** Storms/Hurricanes (USFWS, 2007; NatureServe, 2015)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Coastal erosion is responsible for the loss of dune environment along the Atlantic coast, particularly during winter storms, tropical storms, and hurricanes (USFWS, 2007; NatureServe, 2015)

### **Recovery**

#### **Reclassification Criteria:**

1. The continued viability of the beach mouse populations at the northern and southern ends of Anastasia Island must be assured. Natural population fluctuations must be shown to remain within limits adequate to avoid extinction from chance events or genetic deterioration (e.g., inbreeding depression or excessive loss of heterozygosity). Accordingly, each population of the mouse should support a breeding population of 500 if the subspecies is to be considered for reclassification (USFWS, 1993).
2. At least two more viable populations should be established. These populations should be within the mainland portion of the historic range of the subspecies. However, the only site with any potential for this appears to be the coastal portion of Guana River State Park, managed by the Florida Department of Natural Resources. As discussed, efforts are currently underway to reestablish beach mice at this site. It is uncertain if sufficient dune habitat exists at the site, but the area is basically managed in a manner compatible with the existence of beach mice. Guana River State Park includes a longer beach than those at Anastasia State Recreation Area and Fort Matanzas National Monument, possibly providing greater protection from storm damage (USFWS, 1993).
3. All populations should be monitored for at least 5 consecutive years to assure that condition 1 is met before considering reclassification (USFWS, 1993).

#### **Delisting Criteria:**

1. Viable populations are maintained on the five public land areas where the subspecies currently occurs. Each population should not fluctuate below an effective breeding size of 500 individuals (USFWS, 1993).
2. Five additional viable populations are established throughout the historic range of the subspecies. If acquisition of the Archie Carr National Wildlife Refuge is completed, this area may provide reintroduction sites for one or more populations. The primary purpose of the Refuge is to protect nesting beaches for the loggerhead sea turtle; this goal is compatible with the maintenance of suitable habitat for beach mice (USFWS, 1993).
3. These populations should be monitored for at least 5 years before considering delisting. If delisted, these populations will continue to be periodically monitored as required by the Act (USFWS, 1993).

**Recovery Actions:**

- 1. Protect beach mouse habitat. Use provisions of ESA to protect beach mice. Protect beach mouse habitat on private lands. Implement or encourage specific management actions (USFWS, 1993).
- 2. Monitor beach mice. Both subspecies should be monitored to assure that further declines in range and numbers do not occur without recovery actions being taken. Monitoring will also provide information on sites from which to select animals for reintroduction. Both trapping and sign should be used in monitoring these subspecies (USFWS, 1993).
- 3. Reestablish populations. Identify recipient sites. Identify donor populations. Release mice into new sites. Monitor introduced populations (USFWS, 1993).
- 4. Initiate captive propagation. Identify donor site for breeding stock. Establish breeding colony. Identify and prepare recipient sites. Reintroduce mice. Monitor success of new populations (USFWS, 1993).
- Educate public. The general public regularly uses beach areas in and adjacent to beach mouse habitat for recreational purposes. Public support for beach mouse recovery should therefore be encouraged. The public should understand that continued existence of beach mice is an indication that healthy beach and dune systems are being maintained. Responsible agencies should produce brochures, signs, and other materials to educate the public about the ecological role of beach mice in beach and dune communities. The public should be informed of recreational practices that are compatible with the continued existence of beach mice (USFWS, 1993).

**Conservation Measures and Best Management Practices:**

- Revise the current recovery plan to include updated objective and measurable recovery criteria. Currently, the recovery plan includes both the Anastasia Island beach mouse and the SEBM. Individual plans should be developed for these two subspecies to address the specific recovery actions relating to each subspecies (USFWS, 2008).
- Provide funding and technical support for further research on: (a) The effects of prescribed burning and other management tools within the dune habitat at all sites that currently have SEBM populations. Continue working with public land managers to increase management on their sites. (b) Improve the management of coastal strand/scrub habitat at MINWR/KSC, CCAFS, ACNWR, SISP, and PINWR to expand the available habitat for SEBM. It should be supported by research to appropriately address the ecological requirements of SEBM to achieve habitat restoration needs (e.g., prescribed fire and mechanical treatment of the vegetative component in the coastal strand/scrub and surveys or planting as needed of necessary food resources). Funding should be provided to support habitat restoration projects. (c) Continue genetic sampling of different populations. Goals for genetic sampling should be defined and a protocol established to achieve these goals. Such sampling can tell us if inbreeding depression is occurring. This information can also help the Service determine what constitutes a stable population for SEBM recovery. (d) Perform a population viability analysis to estimate the probability of survival of SEBM populations of differing effective breeding size (USFWS, 2008).
- Develop an emergency response plan to outline actions taken in case of severe threats to the persistence of SEBM (i.e., forecasted category 5 hurricane, feral cat population increase, population crash) (Traylor-Holzer and Lacy 2007) (USFWS, 2008).
- Develop and implement a long-term monitoring program for SEBM throughout its current and historic range. This plan should include goals and objectives such as habitat mapping; obtaining demographic, landscape, or dispersal data; estimating future population trends or the likelihood of

extinction; assessing management options; and evaluating future research priorities. A monitoring program is necessary for several other recommendations listed, particularly the Emergency Response Plan, land acquisition, translocation, and habitat management projects (USFWS, 2008).

- Develop a translocation plan to identify key sites, set criteria for when translocations are needed, consider genetic as well as demographic characteristics of the donor and recipient populations, and include an assessment of the suitability of the recipient habitat (i.e., habitat quality, food resources present, minimization or removal of feral cats and other threats). Public-private partnerships and easements should also be explored. Future translocation of SEBM should be considered at ACNWR, SISP, FPSP, and HSNWR if it can be shown that there is suitable habitat (e.g., the coastal strand/scrub habitat) to support additional mice and potential threats have been removed (USFWS, 2008).
- Continue to educate the public at the public parks about the importance of dune habitat. In addition, an outreach/education program should be developed and focused on the threats feral and house cats pose to wildlife (USFWS, 2008).
- Enforce the use of crossovers in areas with suitable beach mouse habitat to reduce impacts to the dunes. Restore habitats with native plant species that are also food sources for SEBM (USFWS, 2008).
- Continue feral cat removal and control from areas of suitable SEBM habitat (USFWS, 2008).

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## **SPECIES ACCOUNT: *Peromyscus polionotus peninsularis* (St. Andrew beach mouse)**

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### ***Species Taxonomic and Listing Information***

**Commonly-used Acronym:** SABM

**Listing Status:** Endangered; 12/18/1998; Southeast Region (R4) (USFWS, 2016)

### **Physical Description**

The St. Andrew beach mouse's fur is a pale, buff/brown color on its head and back with extensive pure white coloration on its underparts, sides, feet, face, and tail (Howell 1939). They have two distinct rump color patterns, tapered or squared (Bowen 1968). Their average size is: head and body length, 2.95 in (75 mm); tail length, 2.05 in (52 mm); and hind foot length, 0.73 in (18.5 mm) (James 1992) (USFWS, 2010).

### **Taxonomy**

The sub-specific classification of beach mice was based on the geographic variations in pelage characteristics and skeletal measurements. These variations were thought to be genetically based (Bowen 1968). Wooten and Holler (1999) conducted genetic analyses, using microsatellite data to look at the historic relationship of the two known populations of St. Andrew beach mouse (St. Joseph Peninsula State Park and Crooked Island) and the population of Choctawhatchee beach mouse (*P. p. allopheys*) found on Shell Island in Bay County. Their results indicated a rather complex genetic relationship between these populations. When comparing the alleles of the three populations they found the St. Joseph Peninsula State Park population's alleles were unique and its allele frequencies are substantially different from that of the Choctawhatchee beach mouse population on Shell Island. They found, however, that all the alleles of the St. Andrew beach mouse population on Crooked Island were found in both of the other two populations. Furthermore, they found two of the alleles were "uniquely shared" with either the St. Andrew beach mouse population on St. Joseph Peninsula State Park or the Choctawhatchee beach mouse population on Shell Island. These results revealed that beach mice inhabiting Crooked Island East historically may have had some genetic exchange with beach mice inhabiting Shell Island (Choctawhatchee beach mice). However, the analyses did not suggest that beach mice inhabiting Crooked Island East should not be classified as St. Andrew beach mice. Recent genetic research, based on DNA sequencing, suggests that Crooked Island East was historically inhabited by Choctawhatchee beach mice, contradicting the accepted historic ranges of the subspecies. These findings have yet to be peer reviewed (USFWS, 2010).

### **Historical Range**

Formerly occurred from Crooked Island East, Bay County, Florida, southeast to Indian Peninsula, Gulf County, Florida. (NatureServe, 2015)

### **Current Range**

More recently restricted to east end of Crooked Island East, Bay County, and distal half of St. Joseph Peninsula, Gulf County. Trapping during 1992-1994 suggests that the Crooked Island East population is now extirpated (J. A. Gore, N. R. Holler). Extant populations are limited to the northern 20 km of the St. Joseph Peninsula: St. Joseph Peninsula State Park (well established

and widely distributed) and private property on the Gulf side of St. Joseph Peninsula south of the park boundary to within 3 km of Cape San Blas (Gore, in litt. 1994; USFWS 1998). In 1997 and 1998, the subspecies was reintroduced, apparently successfully on East Crooked Island (USFWS 1998) (NatureServe, 2015).

**Distinct Population Segments Defined**

No

**Critical Habitat Designated**

Yes; 10/12/2006.

**Legal Description**

On October 12, 2006, the U.S. Fish and Wildlife Service (Service), revised critical habitat for the Perdido Key beach mouse (*Peromyscus polionotus trissyllepsis*) and Choctawhatchee beach mouse (*Peromyscus polionotus allopshys*) and designated critical habitat for the St. Andrew beach mouse (*Peromyscus polionotus peninsularis*) under the Endangered Species Act of 1973, as amended (Act). In total, approximately 6,193 acres (ac) (2,506 hectares (ha)) were designated as critical habitat for the three subspecies.

**Critical Habitat Designation**

Three units are designated as critical habitat for the St. Andrew beach mouse: (1) East Crooked Island, (2) Palm Point, and (3) St. Joseph Peninsula. Table 4 provides the approximate area encompassed within each critical habitat unit determined to meet the definition of critical habitat for the St. Andrew beach mouse. Urban development and the lack of available suitable habitat limited the designation to these areas. The large size and high quality of habitat in SABM-1 is considered to alleviate some of the need for additional units or independent populations.

**SABM-1: East Crooked Island Unit** SABM-1 consists of 826 ac (335 ha) in Bay County, Florida. This unit encompasses essential features of beach mouse habitat on East Crooked Island from the entrance of St. Andrew Sound to 1 mi (1.6 km) west of Mexico Beach, and the area from the MHWL to the seaward extent of the maritime forest (not including Raffield Peninsula). Beach mouse habitat in this unit consists of primary, secondary, and scrub dune habitat and possesses all five PCEs. St. Andrew beach mice were known to inhabit the unit in 1986 and 1989 (James 1992, pp. 88–90), though the population was presumably extirpated after 1989 due to impacts from hurricanes. The East Crooked Island population was reestablished with donors from St. Joseph State Park in 1997. This unit was occupied at the time of listing. Recent live-trapping confirms present occupation of mice (Moyers and Shea 2002, p. 3; Service 2002b). This unit maintains connectivity along the island and this unit is essential to provide a donor population following storm events. The majority of this unit is federally owned (Tyndall AFB), while the remaining habitat is privately owned. Threats specific to this unit that may require special management considerations include artificial lighting, presence of feral cats as well as other predators at unnatural levels, and high recreational and military use that may result in soil compaction, damage to dunes, or other decrease in habitat quality.

**SABM-2: Palm Point** SABM-2 consists of 162 ac (65 ha) of private lands in Gulf County, Florida. This unit encompasses habitat from Palm Point 1.25 mi (2.0 km) northwest of the inlet of the Gulf County Canal to the southeastern boundary of St. Joe Beach and the area from the MHWL to the seaward extent of the maritime forest. We consider beach mice to have been present in this unit

at the time of listing, because St. Andrew beach mice were documented in the area by Bowen (1968, pp. 86–88). Since St. Andrew beach mouse habitat is limited to only two other areas, protecting this mainland site located within the species' historic range is needed for the subspecies' long-term persistence. As other viable opportunities are limited or nonexistent, this unit is essential to reduce the threats of stochastic events to this subspecies. Furthermore, as this unit is on the mainland, it is somewhat buffered from the effects of storm events. This area provides frontal and scrub dune habitat (PCEs 2 and 3), but may provide limited connectivity between habitats. Threats specific to this unit that may require special management considerations include habitat fragmentation, habitat loss, artificial lighting, presence of feral cats as well as other predators at unnatural levels, and high residential use that may result in soil compaction, damage to dunes, or other decrease in habitat quality.

SABM–3: St. Joseph Peninsula SABM–3 consists of 1,502 ac (607 ha) in Gulf County, Florida. This unit encompasses essential features of beach mouse habitat within the boundary of St. Joseph Peninsula State Park (Park) as well as south of the Park to the peninsula's constriction north of Cape San Blas (also known as the "stumphole" region) and area from the MHWL to the seaward extent of the maritime forest. Beach mouse habitat in this unit consists of primary, secondary, and scrub dune habitat, and provides a relatively contiguous expanse of habitat within the historic range of the St. Andrew beach mouse. This unit possesses all five PCEs and was occupied at the time of listing. St. Andrew beach mice were known to inhabit this unit in 1986 and 1987 (James 1992, pp. 88–90), 1989, 1992, 1993, and 1994 (Gore 1994, pp. 2–5). In addition, recent tracking efforts suggest that mice continue to occupy private lands south of the Park. The Park alone does not provide sufficient habitat to allow for population expansion along the peninsula, which may be necessary for a population anchored by the tip of a historically dynamic peninsula. A continuous presence of beach mice along the peninsula is the species' best defense against local and complete extinctions due to storm events. The population of St. Andrew beach mice inhabiting this unit appears to possess unique genetic variation, and displays greater than expected genetic divergence from other populations (Wooten and Holler 1999, pp. 65–66). Portions of this unit are managed by the Florida Park Service, while the remaining area is privately owned. Threats specific to this unit that may require special management considerations include artificial lighting, habitat fragmentation and habitat loss, presence of feral cats as well as other predators at unnatural levels, and high recreational use that may result in soil compaction, damage to dunes, or other decrease in habitat quality. The population inhabiting this unit may also be particularly susceptible to hurricanes due to its placement within St. Joseph Bay (the peninsula is a thin barrier peninsula with a northsouth orientation).

#### **Primary Constituent Elements/Physical or Biological Features**

Critical habitat units are designated for Bay and Gulf Counties, Florida. The primary constituent elements of critical habitat for the St. Andrew beach mouse are the habitat components that provide:

- (i) A contiguous mosaic of primary, secondary, and scrub vegetation and dune structure, with a balanced level of competition and predation and few or no competitive or predaceous nonnative species present, that collectively provide foraging opportunities, cover, and burrow sites;
- (ii) Primary and secondary dunes, generally dominated by sea oats (*Uniola paniculata*), that despite occasional temporary impacts and reconfiguration from tropical storms and hurricanes, provide abundant food resources, burrow sites, and protection from predators;

(iii) Scrub dunes, generally dominated by scrub oaks (*Quercus* spp.), that provide food resources and burrow sites, and provide elevated refugia during and after intense flooding due to rainfall and/or hurricane-induced storm surge;

(iv) Functional, unobstructed habitat connections that facilitate genetic exchange, dispersal, natural exploratory movements, and re-colonization of locally extirpated areas; and

(v) A natural light regime within the coastal dune ecosystem, compatible with the nocturnal activity of beach mice, necessary for normal behavior, growth, and viability of all life stages.

### **Special Management Considerations or Protections**

Critical habitat does not include man-made structures existing on the effective date of this rule and not containing one or more of the primary constituent elements, such as buildings, aqueducts, airports, driveways, and roads, and the land on which such structures are located.

The features essential to the conservation of beach mice in all of the areas we are designating may require special management considerations or protections due to threats to the subspecies or its habitat. Such management considerations and protections include: Management of non-native predators and competitors, management of non-native plants, protection of beach mice and their habitat from threats by road construction, urban and commercial development, heavy machinery, and recreational activities.

### ***Life History***

#### **Feeding Narrative**

Adult: The frontal dunes provide a more diverse and higher energy food resource for the beach mouse, although the food is cyclic in its availability. Scrub dunes provide a more stable, but less diverse, food source and are believed to provide a food source for times when food resources in the frontal dune systems are low (Sneckenberger 2001). Diets are driven by the availability of food within the habitat and food item shifts both seasonally and yearly (Moyers 1996). No diet studies have been conducted on St. Andrew beach mice specifically; however, studies have been conducted on other beach mice subspecies along the northern Gulf Coast (Blair 1951, Ehrhart 1978, Holler 1992, Moyers 1996). Moyers (1996) found that the diets of Perdido Key beach mouse (*P. p. trissyllepsis*), Alabama beach mouse, and Santa Rosa beach mouse were similar (Moyers 1996). Bluestem (*Schizachyrium scoparium*) and sea oats (*Uniola paniculata*) were most frequently visited (Blair 1951); however, Moyer (1996) found beach mice showed no preference to any food item, instead their selection of food items appeared to be based on availability. The seeds of sea oats and bluestem and the fruits of dune spurge (*Chamaesyce bombensis*), ground cherry (*Physalis angustifolia*), and evening primrose (*Oenothera humifusa*) are utilized in autumn, while sea rocket (*Cakile lanceolata*), dune toadflax (*Linaria floridana*), and evening primrose make up the spring diet (Moyers 1996). Furthermore, insects, primarily Coleoptera beetles (Holler 1992), fire ants (*Solenopsis invicta*) and harvester ants (*Pogonomyrmex badius*) (Moyers 1996) have been found to make up a part of the beach mouse diet (Ehrhart 1978, Moyers 1996) (USFWS, 2010).

#### **Reproduction Narrative**

Adult: Studies suggest that *P. polionotus* are generally monogamous (Foltz 1981, Lynn 2000b), apparently forming pair bonds for life (Blair 1951). It appears, however, that some paired males may also mate with unpaired females (S. Sneckenberger, Service, pers. comm., 2005). Breeding activity is greatest during the fall and winter months (Blair 1951, Rave and Holler 1992). Female *P. polionotus* can become sexually mature as early as around 30 days old (Clark 1938). Gestation ranges from 23 to 24 days or 25 to 31 days for lactating females (Whitaker and Hamilton 1998). Litters average 3-4 in size, but can range from one to five individuals, with litter size tending to be positively correlated to female size (Caldwell and Gentry 1965a). Over a lifetime, under laboratory conditions, a female beach mouse can produce 80 young or more (Bowen 1968). Longevity. Rave and Holler (1992) found, of the mice they trapped, 63% of the mice lived 4 months or less, 37% lived 5 months or longer, and 21 individual mice lived 12-20 months beyond first capture. There is no significant difference in survival rates between males and females. However, mice that disperse from their natal grounds persisted significantly longer (males:  $138 \pm 19$  days; females:  $125 \pm 18$  days) than mice that remain in their natal grounds (males:  $96 \pm 10$  days; females:  $92 \pm 8$  days) (Swilling 2000) (USFWS, 2010).

**Spatial Arrangements of the Population**

Adult: Clumped (NatureServe, 2015)

**Environmental Specificity**

Adult: Narrow (inferred from NatureServe, 2015)

**Tolerance Ranges/Thresholds**

Adult: Low (inferred from NatureServe, 2015)

**Site Fidelity**

Adult: High (inferred from NatureServe, 2015)

**Habitat Narrative**

Adult: Coastal Dune and Coastal Strand. Occurs in well-developed high front dunes where the dominant plant cover is sea oats; also occurs on older and higher back dunes, where burrows often are at the base of blow-outs held up by roots of live oak shrubs (sea oats and rosemary also may be present); inhabits low front dunes and lower back dunes covered with bunch grass and beach grass (*PANICUM*) on Crooked Island (James 1992). Burrowing in or using soil (NatureServe, 2015) Narrow environmental specificity, high ecological integrity, low tolerance range and high site fidelity are inferred based on the very specific habitat this species inhabits (NatureServe, 2015).

**Dispersal/Migration****Motility/Mobility**

Adult: High (NatureServe, 2015)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migrant (NatureServe, 2015)

**Dispersal**

Adult: Low (inferred from NatureServe, 2015)

**Immigration/Emigration**

Adult: Unlikely (inferred from NatureServe, 2015)

**Dispersal/Migration Narrative**

Adult: Mice are highly mobile and NatureServe (2015) notes this species is non-migratory. Low dispersal and unlikely immigration/emigration are inferred based on species habitat and the isolated nature of the known populations (NatureServe, 2015).

***Population Information and Trends*****Population Trends:**

Decreasing (NatureServe, 2015)

**Resiliency:**

Low (inferred from Natureserve, 2015)

**Representation:**

Low (inferred from Natureserve, 2015)

**Redundancy:**

Low (inferred from Natureserve, 2015)

**Number of Populations:**

1 - 5 (NatureServe, 2015)

**Population Size:**

1-1000 total individuals (NatureServe, 2015)

**Population Narrative:**

Low resiliency, representation and redundancy are inferred from NatureServe (2015) based on low number of populations and restricted range and habitat. In addition, NatureServe notes that the short-term population trend is a decline of 10-30%, the number of populations is 1-5 and the total population is between 1 and 1,000.

***Threats and Stressors***

**Stressor:** Human impacts (NatureServe, 2015)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Threats include destruction of habitat via residential and commercial development, military exercises, vehicular and pedestrian traffic. (Natureserve, 2015). Threats also include artificial illumination and military exercises (USFWS, 2010).

**Stressor:** Storms (NatureServe, 2015)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Erosion caused by tropical storms and changes in water current patterns (about 50% of occupied habitat within the St. Joseph Peninsula was lost in October 1995 due to Hurricane Opal) is also a threat (NatureServe, 2015).

**Stressor:** Predation (NatureServe, 2015)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** Predation by cats (apparently not now a major problem but could become one) and predation by non-native natural predators such as red fox and coyotes (also feral hogs) (NatureServe, 2015; USFWS, 2010).

### ***Recovery***

**Reclassification Criteria:**

The St. Andrew beach mouse will be considered for downlisting to threatened status when the following measures are achieved: (USFWS, 2010)

1. Stable or increasing population trends are maintained at St. Joseph Peninsula State Park and East Crooked Island on Tyndall Air Force Base over a 10-year period based on data obtained from accepted, standardized, monitoring methods. (USFWS, 2010)
2. An additional viable or self sustaining population is reestablished at St. Joe Beach that shows a stable or increasing trend, after the initial repopulation of unoccupied habitat, over a 10-year period based on data obtained from accepted, standardized, monitoring methods. (USFWS, 2010)
3. At least 87% of designated St. Andrew beach mouse critical habitat is protected and under a management plan that addresses conservation of beach mice. The plans, at a minimum, address the following: a) Impact of commercial/residential development and recreational use, including that of pedestrians and motorized vehicles, to beach mouse habitat. b) Impact of shoreline erosion to beach mouse habitat. c) Impact of artificial lighting on beach mouse habitat. d) Control of feral cats and hogs in beach mouse habitat. (USFWS, 2010)
4. In areas with known populations of beach mice (Tyndall Air Force Base's property at East Crooked Island, St. Joseph Peninsula State Park, and their respective adjacent private lands), non-native predators, including free roaming cats and cat colonies, are controlled at levels in which they do not pose a threat to beach mice (USFWS, 2010).
5. County or local government, within the range of the St. Andrew beach mouse, have regulations or other protection mechanisms that: a) Minimize impacts to dunes in beach mouse habitat due to recreational use. b) Prohibit free-roaming cats and cat colonies. c) Minimize impacts of commercial and residential developments in primary, secondary, and scrub dunes. Measures include minimizing footprints; preserving connectivity between primary, secondary and scrub dunes; using native landscaping; and constructing boardwalks over dunes for beach access. d) Minimize impacts of artificial lighting in beach mouse habitat by requiring sea turtle

lighting, in areas visible from the beach and wildlife lighting, in areas not visible from the beach (USFWS, 2010).

6. An emergency response plan is prepared to prevent extirpation of any population of St. Andrew beach mice from tropical storms/hurricanes and other disasters (USFWS, 2010).

7. If determined to be necessary, an Action Plan is prepared to address the potential threat of cross-breeding with Choctawhatchee beach mice from West Crooked Island (USFWS, 2010).

8. House mice are controlled in areas with known populations of beach mice at levels in which they do not pose a threat to the population(s) (USFWS, 2010).

**Recovery Actions:**

- The St. Andrew beach mouse will be considered for delisting when all the downlisting criteria have been met and the following delisting criteria are achieved (USFWS, 2010).
- 1. Stable or increasing population trends are maintained at St. Joseph Peninsula State Park, East Crooked Island on Tyndall Air Force Base, and St. Joe Beach over a 20-year period based on data obtained from accepted, standardized, monitoring methods (USFWS, 2010).
- 2. An additional viable population is reestablished at Cape San Blas, Eglin Air Force Base, and has a stable or increasing population trend over a 10- year period based on data obtained from standardized monitoring methods (USFWS, 2010).
- 3. All designated St. Andrew beach mouse critical habitat on public land is protected and under a management plan that addresses conservation of beach mice, priority is given to those lands that provide connectivity. The plans, at a minimum, manage for the following: a) Impact of commercial/residential development and recreational use, including that of pedestrians and motorized vehicles, to beach mouse habitat. b) Impact of shoreline erosion to beach mouse habitat. c) Impact of artificial lighting on beach mouse habitat. d) Control of feral cats and hogs, including free ranging cats in beach mouse habitat (USFWS, 2010).
- 4. Within all critical habitat that is protected and under a management plan, non-native predators, including free roaming cats and cat colonies, are controlled at levels in which they do not pose a threat to beach mice (USFWS, 2010).
- 5. County or local government regulations or other protection mechanisms as set forth in the downlisting criteria have adequate compliance and enforcement (USFWS, 2010).
- 6. House mice continue to be deemed a minimal or no threat to St. Andrew beach mouse populations (USFWS, 2010).

***Conservation Measures and Best Management Practices:***

- The conservation of St. Andrew beach mice on rapidly developing private lands will need to be addressed. These lands also include areas designated as critical habitat for the subspecies. Development can be addressed in a variety of ways including: working with the State and local governments on the siting of structures and facilities and landscaping with native vegetation within St. Andrew beach mouse habitat, regulatory requirements, and education of property owners. Public land managers are under pressure to manage natural resources while providing for other uses of the resource such as military training and recreation. These public land managers will need to balance these often competing mandates to ensure the conservation of the St. Andrew beach mouse (USFWS, 2009).

- Another necessary action includes the control of free ranging pet and feral cats and other predators on public and private lands. This can be accomplished through local animal control organizations, the established state-federal land partnership, and implementation of best management practices on private and public lands (adequate refuse management, predator proof trash receptacles, and landscaping with native plant species) (USFWS, 2009).
- Conservation of St. Andrew beach mouse (and other wildlife) should be included in local emergency response plans. The plans could incorporate best management practices for debris clean up, responder and public access to affected areas, and infrastructure repair or rebuild. Additional perturbations on already stressed St. Andrew beach mouse habitat by storm passage could have significant effects on the recovery of the species following emergency events (USFWS, 2009).
- Other actions to facilitate recovery include the implementation of consistent range-wide monitoring of the beach mouse. A St. Andrew beach mouse recovery plan needs to be completed which should include plans for long-term monitoring, role of captive breeding and translocations of mice into unoccupied habitat where reasons for the mouse's extirpation have been addressed or ameliorated (USFWS, 2009).

## References

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## SPECIES ACCOUNT: *Peromyscus polionotus phasma* (Anastasia Island beach mouse)

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### *Species Taxonomic and Listing Information*

**Commonly-used Acronym:** AIBM

**Listing Status:** Endangered; 05/12/1989; Southeast Region (R4) (USFWS, 2016)

### **Physical Description**

A large subspecies relative to other forms of the polionotus complex. Ten adult Anastasia island beach mice from the type locality averaged 138.5 millimeters (mm) (5.40 inches (in)) in total length and 53 mm (2.07 in) in tail length (Osgood 1909). Howell (unpublished ins, ca. 1940) described the coloration as light ochraceous buff dorsally, white underparts, a unicolor tail, and indistinct white markings on the nose and face (USFWS, 1993).

### **Taxonomy**

The Anastasia Island beach mouse was described in 1898 by Bangs as a full species, *Peromyscus phasma*. Osgood (1909) relegated it to subspecific standing under *P. polionotus*. The type locality is Point Romo, Anastasia Island, St. Johns County, Florida. (USFWS, 1993).

### **Historical Range**

Historic range: mouth of St. Johns River at Jacksonville (Duval County) to southern end of Anastasia Island (St. Johns County), Florida (NatureServe, 2015).

### **Current Range**

Formerly occupied two adjacent barrier islands on Florida's east coast but currently restricted to the northernmost and southernmost ends of Anastasia Island (Frank and Humphrey 1992). In the early 1990s, a second population was being established within the historic range at Guana River State Park on an adjacent island several kilometers to the north (Frank 1992); as of early 1995, the reintroduction was going well (Tardona 1995) (NatureServe, 2015). At the time of listing in 1989, AIBM were known to occur along the 14.5 miles of Anastasia Island in St. Johns County, Florida within ASP and FMNM and in the dunes and swales in between. ASP continues to provide 3.5 miles of suitable habitat and undeveloped coastline to support AIBM at the north end of Anastasia Island (FDEP 2016). AIBM continue to occupy the narrow coastal dune habitat area between ASP and FMNM on private lands as well as several St. Johns County Parks (10 miles) (FWC, 2019; Miller 2019; Kropp and Dupree 2015; and Doonan pers. comm.). The width of this occupied habitat varies; Frank and Humphrey (1992) described an idealized cross section of dune topography for Anastasia Island which was approximately 500 feet wide, but most of the dune and swale habitat along the central section of Anastasia Island is much narrower due to the residential development of St. Augustine Beach, Butler Beach and Crescent Beach. AIBM continues to occupy the one mile of undeveloped coastal dune and swale habitat along the ocean and inlet shorelines at FMNM (FWC, 2019; Kropp and Dupree 2015). Habitat at FMNM is similar to ASP; however, beyond the primary dunes the habitat becomes woody, contains dense swales, and is bordered by oak forest to the west (Frank and Humphrey 1992; NPS 2012). Due to the interconnected habitat, Anastasia Island appears to support one population AIBM were reintroduced into historical habitat at GTMNERR in 1992-1993. Fifty-five mice (27 females and

28 males) were trapped at ASP (37) and FMNM (18) and placed in soft-release enclosures at GTMNERR (Frank 1995). This population was augmented again in 2000 (21 males and 12 females) trapped at ASP (33 total) (Bard pers. comm.). The last beach mouse was captured in 2006 and trapping ended in 2012 after 6 years with no captures (Marcum, pers. comm.) due to the probability of extirpation. There has not been any monitoring at this location since 2012 until a 2018 camera trap survey was conducted but it did not detect AIBM (USFWS 2018). It is possible AIBM have gone undetected due to the dynamic nature of small mammal populations and we currently consider the status of this population as unknown but likely extirpated. The 4.2 miles of undeveloped coastal habitat at GTMNERR provides a very narrow dune system for AIBM to use, as does the dune habitat north of GTMNERR along the remainder of the residential beaches to Mickler's Landing and very little dune and swale habitat south of the GTMNERR to St. Augustine Inlet (USFWS 2019)

**Distinct Population Segments Defined**

No

**Critical Habitat Designated**

Yes;

***Life History*****Feeding Narrative**

Adult: Eats fruits and seeds of dune plants, especially sea oats and sea rocket; feeds on invertebrates when seeds scarce (Matthews and Moseley 1990).; Food Habits: Invertivore (Adult, Immature), Granivore (Adult, Immature) Primarily nocturnal.; (NatureServe, 2015)

**Reproduction Narrative**

Adult: May breed all year. Much breeding activity occurs November-January. Produces 2 or more litters per year. Gestation averages 23-24 days (nonlactating) or 28-29 days (lactating). Litter size averages 3-4 (USFWS 1988). Young are weaned in about 18 days. Minimum age at conception is 5 weeks. Apparently monogamous mating system (Kirkland and Layne 1989). Density in high quality habitat 2-90/ha (mean around 30/ha) (Frank 1992).; (NatureServe, 2015)

**Spatial Arrangements of the Population**

Adult: Clumped (NatureServe, 2015)

**Environmental Specificity**

Adult: Narrow (inferred from NatureServe, 2015)

**Tolerance Ranges/Thresholds**

Adult: Low (inferred from NatureServe, 2015)

**Site Fidelity**

Adult: High (inferred from NatureServe, 2015)

**Habitat Narrative**

Adult: Beach dune and coastal strand habitats. Occurs in a narrow strip of sand dunes along the eastern side of Anastasia Island (Frank and Humphrey 1992). Favors beaches with grass/shrub

cover. Sleeps and gives birth in underground burrows; entrances are in clumps of grass or beneath sheltering vegetation (Matthews and Moseley 1990). (NatureServe, 2015) Narrow environmental specificity, high ecological integrity, low tolerance range and high site fidelity are inferred based on the very specific habitat this species inhabits (NatureServe, 2015).

***Dispersal/Migration*****Motility/Mobility**

Adult: High (NatureServe, 2015)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migrant (NatureServe, 2015)

**Dispersal**

Adult: Low (inferred from NatureServe, 2015)

**Immigration/Emigration**

Adult: Unlikely (inferred from NatureServe, 2015)

**Dispersal/Migration Narrative**

Adult: Mice are highly mobile and NatureServe (2015) notes this species is non-migratory. Low dispersal and unlikely immigration/emigration are inferred based on species habitat and the isolated nature of the known populations (NatureServe, 2015).

***Population Information and Trends*****Population Trends:**

short term trend = Increasing (NatureServe, 2015)

**Species Trends:**

Stable. At the time of listing in 1989, AIBM were distributed along the length of Anastasia Island, from the northern end at St. Augustine Inlet, Anastasia State Park (ASP), to the southern end at Matanzas Inlet, Fort Matanzas National Monument (FMNM). AIBM distribution in the coastal dunes and swales along the entire length of Anastasia Island continues today. Since the 2007 Review, there has been a decline in captures during trapping of the primary dunes at the northern section of ASP near the St. Augustine Inlet (FDEP 2016). This decline generally has corresponded to a net loss of primary dune habitat along the northern third of ASP. A 2011 track tube survey confirmed AIBM presence in the restored primary dune habitat at the south end of ASP, an area previously prone to overwash (Kropp and Dupree 2015). This is consistent with the observed habitat restoration and stability gained within the central and southern sections of the park since 2007. These ASP habitat conditions appear to be a function of one or more of the local, coastal navigation/shoreline/dune stabilization projects over the past twenty years, which include maintenance dredging of St. Augustine Inlet, dredging the inlet's ebb tidal shoal, and beach renourishment and dune stabilization efforts south of St. Augustine Inlet. Track tube and trapping surveys in 2011 confirmed that St. Augustine Beach, Butler Beach, and Crescent Beach dune habitats south of ASP and north of FMNM continue to be occupied (Kropp and Dupree 2015; Doonan, pers. comm.). These track tube surveys also indicated that FMNM continues to be occupied. Post Hurricane Mathew (2016) and Irma (2017) track tube monitoring indicate

AIBM continue to occupy the coastal dunes along the entire length of Anastasia Island (FWC 2019). We do not know the current status of the reintroduced population north of St. Augustine Inlet at the Guana-Tolomato-Matanzas National Estuarine Research Reserve (GTMNERR) which has over 4 miles of primary and secondary dunes and coastal scrub habitat located beyond this steep dune system. In 1992, 55 AIBM from ASP and FMNM were released in the coastal dunes of GTMNERR and appeared to flourish. In 2000 an additional 33 AIBM from ASP were released at GTMNERR. We reported in 2007 that this population was in decline and there had not been any captures since the summer of 2006. With no additional captures after 6 years, monitoring was discontinued in 2012 (Marcum, pers. comm.) due the probability that AIBM were extirpated from the GTMNERR. The effort to reintroduce mice into the historic range north of St. Augustine Inlet may have been unsuccessful. Due to the dynamic nature of small mammal populations, beach mice could have gone undetected and we consider the status of this small population as unknown but likely extirpated. In October 2016, Hurricane Matthew moved north along the Atlantic coast of Florida causing a substantial storm surge, erosion and destruction along Anastasia Island's dune habitats. Hurricane Irma, which made landfall and traversed along Florida's Gulf Coast in September 2017, also caused a storm surge event and additional damage to Anastasia Island's coastal habitats. A multi-agency monitoring effort to better understand impacts from the hurricanes and improve recovery efforts was initiated in January 2018 and completed in March 2019 (FWC 2019) and is summarized as follows. Track tube stations were installed along the length of Anastasia Island; 131 track tubes were place at ASP, FMNM and at 4 St. Johns County (SJC) properties in between. Live-trapping was conducted along transects in suitable habitats at ASP (4 transects) and FMNM (3 transects) in June, September, and December 2018 to confirm species identification; opportunist trapping was done on the SJC properties. Initial track tube detection rates for AIBM were 32% at ASP and 36% at FMNM. Detection rates reached highs of 91% at ASP and 68% at FMNM in March, then declined through the fall before increasing again in December. Trapping data confirmed that AIBM was the only rodent species present in dune habitats at all sites. Results and observations indicate that AIBM and the coastal dune habitats they depend upon have been recovering from Hurricanes Mathew and Irma and that AIBM continue to occupy the coastal dunes along the entire length of Anastasia Island. As there was extensive loss and damage to fore dune habitats, a restoration strategy that is site-specific for ASP and FMNM has been developed and these strategies can be applied on county and private properties along the central section of Anastasia Island (Miller 2019). The AIBM distribution along Anastasia Island remains stable and the reintroduced population at GTMNERR is unknown and we believe they are likely extirpated (USFWS 2019).

**Resiliency:**

Low (inferred from NatureServe, 2015)

**Representation:**

Low (inferred from NatureServe, 2015)

**Redundancy:**

Low (inferred from NatureServe, 2015)

**Number of Populations:**

1 - 5 (NatureServe, 2015)

**Population Size:**

1 - 1000 individuals (NatureServe, 2015)

**Population Narrative:**

NatureServe (2015) notes that the short term trend for this species is increasing. In addition NatureServe notes that there are 1-5 known populations with the total number of individuals estimated between 1 and 1,000. Low resiliency, representation and redundancy are inferred from NatureServe (2015) based on low number of populations and restricted range and habitat.

**Threats and Stressors**

**Stressor:** Predation (NatureServe, 2015)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** Beach mouse populations may be regulated by predation by house cats (Frank and Humphrey 1992, Frank 1992), populations of which are introduced/augmented through development. (NatureServe, 2015). Beach mice have a number of natural non-native predators including the coachwhip, corn snake, pygmy rattlesnake, Eastern diamondback rattlesnake, short-eared and great-horned owl, great blue heron, northern-harrier, loggerhead shrike, gray fox, striped skunk, long-tailed weasel, raccoon, bobcat, ghost crabs, red fox and coyotes (USFWS, 2007).

**Stressor:** Development/Construction/Beach driving (NatureServe, 2015)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Development has degraded and fragmented much of the remaining habitat (Frank and Humphrey 1992) (NatureServe, 2015). Beach driving is also mentioned as a cause of human made habitat destruction (USFWS, 2007)

**Stressor:** Storms/hurricanes

**Exposure:**

**Response:**

**Consequence:** Loss of habitat/loss of individuals

**Narrative:** Vulnerable to extinction that could be caused by severe hurricanes (Frank 1992). (NatureServe, 2015)

**Stressor:** Climate change

**Exposure:** Sea level rise

**Response:** May alter habitat

**Consequence:** Population decline

**Narrative:** Sea level rise is a long-term threat to AIBM and all coastal dependent species based on numerous prediction models. According to the Third National Climate Assessment, release May 2014, sea level rise and increasing storm surge events are occurring and are impacting coastal species and ecosystems (Melillo et al. 2014 and Wolf 2014). It is expected that low-lying coastal habitat will be affected most severely by sea level rise. Models such as the Sea Level Rise Affecting Marshes Model (SLAMM) can be used to project different levels of rise such as a 6-foot rise would remove significant amounts of habitat within ASP and FMNM. The varying and

dynamic elements of climate science are inherently long term, complex, and interrelated. At present, the science is not exact enough to precisely predict when and where climate impacts will occur. Although we may know the direction of change, it may not be possible to predict its precise timing or magnitude. Future planning will include guidance and use scenario planning to develop management strategies that account for potential environmental changes, given the future uncertainties in climatic conditions (USFWS 2019).

### ***Recovery***

#### **Reclassification Criteria:**

1. The continued viability of the beach mouse populations at the northern and southern ends of Anastasia Island must be assured. Natural population fluctuations must be shown to remain within limits adequate to avoid extinction from chance events or genetic deterioration. Accordingly, each population of the mouse should support a breeding population of 500 if the subspecies is to be considered for reclassification (USFWS, 1993).
2. At least two more viable populations should be established. These populations should be within the mainland portion of the historic range of the subspecies (USFWS, 1993).
3. All populations should be monitored for at least 5 consecutive years to assure that condition 1 is met before considering reclassification (USFWS, 1993).

#### **Delisting Criteria:**

The Anastasia Island beach mouse shall be considered for delisting when the following criteria are met: 1. The three (3) Anastasia Island Resiliency Units (RU) exhibit stable or increasing demographic and/or occupancy trends as compared to historic levels, and exhibit natural recruitment. (addresses Factors A, C and E) 2. Establish two (2) Resiliency Units of AIBM through reintroduction between St. Augustine Inlet and the St. Johns River that exhibit stable or increasing demographic trends and are comparable to the ASP and FMNM RUs, and exhibit natural recruitment. (addresses Factors A, C and E) 3. When in addition to the above criteria, it can be demonstrated that despite habitat loss associated with sea level rise and development within all of the RUs, sufficient suitable habitat remains for AIBM to remain viable into the foreseeable future. (addresses Factors A, C and E) (viable per criterions 1 and 2) (USFWS 2019b)

#### **Recovery Actions:**

- Protect beach mouse habitat. Use provisions of the ESA to protect beach mice. Protect beach mouse on private lands. Implement or encourage specific management actions (USFWS, 1993).
- Monitor beach mice. Both subspecies should be monitored to assure that further declines in range and numbers do not occur without recovery actions being taken. Monitoring will also provide information on sites from which to select animals for reintroduction. Both trapping and sign should be used in monitoring these subspecies (USFWS, 1993).
- Reestablish populations. Identify recipient sites. Identify donor populations. Release mice into new sites. Monitor introduced populations (USFWS, 1993).
- Initiate captive propagation. Identify donor site for breeding stock. Establish breeding colony. Identify and prepare recipient sites. Reintroduce mice. Monitor success of new populations (USFWS, 1993).

- Educate public. The general public regularly uses beach areas in and adjacent to beach mouse habitat for recreational purposes. Public support for beach mouse recovery should therefore be encouraged. The public should understand that continued existence of beach mice is an indication that healthy beach and dune systems are being maintained. Responsible agencies should produce brochures, signs, and other materials to educate the public about the ecological role of beach mice in beach and dune communities. The public should be informed of recreational practices that are compatible with the continued existence of beach mice (USFWS, 1993).
- Revise the current recovery plan to define objective measurable criteria (both reclassification and delisting criteria), better address the five factors, and update ecological information for the AIBM. Currently, the recovery plan includes both the AIBM and the Southeastern beach mouse. Individual plans should be developed for these two subspecies to address the specific recovery actions and recovery criteria relating to each subspecies (USFWS 2019).
- Continue fostering a working partnership with partners and stakeholders: Florida Fish and Wildlife Conservation Commission, Florida Department of Environmental Protection's Anastasia State Park and Division of Beaches and Shores, U.S. Army Corps of Engineers, Fort Matanzas National Monument, Guana-Tolomoto-Matanzas National Estuarine Research Reserve, and St. Johns County and beach front communities of South Ponte Vedra, Vilano Beach, St. Augustine Beach, Butler Beach and Crescent Beach for recovery of Anastasia Island beach mouse and all beach mice subspecies (USFWS 2019).
- Develop an emergency response plan to outline and update actions to be taken in case of severe threats to the persistence of AIBM (i.e., forecasted category 5 hurricane, feral cat population increase, population crash) (Traylor-Holzer and Lacy 2007). An emergency action plan has been developed for ASP, which provides a protocol for the live trapping and removal of mice from the park in case of a hurricane (FDEP 2016) (USFWS 2019).
- Improve the management of AIBM habitat at ASP, FMNM, GTMNERR and St. Johns County Parks to expand and or improve the available habitat and travel corridors for AIBM. Enforce the use of crossovers in areas with suitable beach mouse habitat to reduce impacts to the dunes. Restore and manage habitats with native plant species that are also food sources for AIBM. Continue to educate the public at the public parks about the importance of the dune habitat (USFWS 2019).
- Develop and implement a monitoring program for AIBM, including a survey to determine presence/absence of the reintroduced population at GTMNERR. This plan should include some goals and objectives such as habitat mapping; obtaining demographic, landscape, or dispersal data; estimating future population trends or the likelihood of extinction; assessing management options; or evaluating future research priorities. A monitoring program is necessary for several other recommendations listed, particularly the Emergency Response Plan, land acquisition, translocation, and habitat management projects (USFWS 2019).
- A comprehensive translocation plan is needed to identify key sites, set criteria for when translocations are needed, consider genetic as well as demographic characteristics of the donor and recipient populations, and should include an assessment of the suitability of the recipient habitat (i.e., habitat quality, have feral cats and other threats been minimized or removed) (USFWS 2019).
- Continue genetic sampling and conduct a taxonomic assessment of the species and its subspecies. A priority is to determine genetic diversity across Anastasia Island and set goals for improving. Sampling can also tell us if inbreeding depression is occurring. This

information will help the Service determine what constitutes a stable population for AIBM recovery (USFWS 2019).

- Perform a population viability analysis to estimate the probability of survival of beach mice populations of differing effective breeding size (USFWS 2019).
- Remove feral cats from areas of suitable AIBM habitat. Develop an outreach/education program focused on the threats feral and free-ranging cats pose to wildlife (USFWS 2019).
- Use top down construction techniques for new, repair or replacement of dune walkovers (USFWS 2019).

***Conservation Measures and Best Management Practices:***

- Revise current recovery plan to include updated objectives and measurable recovery criteria (USFWS, 2007).
- Provide funding and technical support for further research on: The effects of prescribed burning and other management tools (e.g. removal of Wax myrtle) on AIBM. Continue working with public land managers to increase management of their sites. Improve the management of coastal strand habitat at GTMNEER-Guana River to expand all available habitat for AIBM. Continue genetic sampling of different populations. Perform population viability analysis to estimate probability of survival of animal populations of differing effective breeding size (USFWS, 2007).
- Develop an emergency response plan to outline actions taken in case of severe threats to the persistence of AIBM (USFWS, 2007).
- Develop and implement a monitoring program for AIBM (USFWS, 2007).
- Discuss with FMNM on how to better monitor beach mouse populations and manage the habitat, and address the threats for AIBM (USFWS, 2007).
- Develop a translocation plan to identify key sites, set criteria for when translocations are needed, consider genetic as well as demographic characteristics of the donor and recipient populations and include an assessment of the suitability of the recipient habitat. Public-private partnerships should also be explored (USFWS, 2007).
- Continue to educate the public (USFWS, 2007).
- Enforce the use of crossovers in areas with suitable beach mouse habitat to reduce impacts to the dunes (USFWS, 2007).
- Continue feral cat removal and control from areas of suitable AIBM habitat (USFWS, 2007).

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## **SPECIES ACCOUNT: *Peromyscus polionotus trissyllepsis* (Perdido Key beach mouse)**

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### ***Species Taxonomic and Listing Information***

**Commonly-used Acronym:** PKBM

**Listing Status:** Endangered; June 6, 1985; Southeast Region (R4) (USFWS, 2016)

### **Physical Description**

A small pale mouse with a white belly, white feet, white to pale grayish-brown tail that lacks a dorsal stripe, and membranous ears; dorsum is grayish fawn to wood brown with a distinct very pale yellow hue; most of the head and the lower sides are white; total length 115-140 mm, tail 45-55 mm, hind foot 16.2-18.2 mm, ear 13.4-15 mm (Holler 1992, Layne 1978). Length 13 cm (NatureServe, 2015).

### **Taxonomy**

Paler than subspecies AMMOBATES and ALLOPHRYS (more orange-brown or yellow brown). Pigmentation does not extend down the thighs as in some ALLOPHRYS. Darker and slightly larger than the western population of subspecies LEUCOCEPHALUS found on Santa Rosa Island to the east, and much paler than the uniform brownish-fawn subspecies POLIONOTUS to the north (Layne 1978) (NatureServe, 2015).

### **Historical Range**

Coastal dunes between Perdido Bay, Alabama, and Pensacola Bay, Florida (NatureServe, 2015).

### **Current Range**

Gulf State Park at the western end of Perdido Key in Alabama and Gulf Islands National Seashore at the eastern end of Perdido Key in Florida (reintroduced; Holler et al. 1989); not known to occur in the intervening area, though mouse tracks have been observed (may be MUS MUSCULUS). Possibly contiguous with subspecies POLIONOTUS prior to construction of the intracoastal ship canal, but perhaps even then the population was isolated by a band of poorly drained soils (Holler 1992) (NatureServe, 2015).

### **Distinct Population Segments Defined**

No

### **Critical Habitat Designated**

Yes; 6/6/1985.

### **Legal Description**

On October 12, 2006, the U.S. Fish and Wildlife Service (Service), revised critical habitat for the Perdido Key beach mouse (*Peromyscus polionotus trissyllepsis*) and Choctawhatchee beach mouse (*Peromyscus polionotus allophrys*) and designated critical habitat for the St. Andrew beach mouse (*Peromyscus polionotus peninsularis*) under the Endangered Species Act of 1973, as amended (Act). In total, approximately 6,193 acres (ac) (2,506 hectares (ha)) were designated as critical habitat for the three subspecies. This action adds approximately 135 ac (44 ha) to the

amount of currently designated critical habitat for the Perdido Key beach mouse and 1,629 ac (659 ha) to the area designated for the Choctawhatchee beach mouse.

### **Critical Habitat Designation**

Five units are designated as critical habitat for the Perdido Key beach mouse: (1) Gulf State Park, (2) West Perdido Key, (3) Perdido Key State Park, (4) Gulf Beach, and (5) Gulf Islands National Seashore. Table 1 provides the approximate area (acres and hectares) encompassed within each critical habitat unit determined to meet the definition of critical habitat for the Perdido Key beach mouse. Since the Perdido Key beach mouse would be extinct without the aid of reestablishment programs (Holler et al. 1989, pp. 398, 403; Service 2004b, p. 1–2), most of the habitat remaining within the subspecies' historic range (Perdido Key) is considered essential for its continued existence.

**PKBM–1: Gulf State Park Unit** PKBM–1 consists of 115 ac (46 ha) in southern Baldwin County, Alabama, on the westernmost region of Perdido Key. This unit encompasses essential features of beach mouse habitat within the boundary of Gulf State Park from the west tip of Perdido Key at Perdido Pass east to approximately 1.0 mi (1.6 km) west of where the Alabama-Florida State line bisects Perdido Key and the area from the MHWL north to the seaward extent of the maritime forest. This unit was occupied by the species at the time of listing. Perdido Key beach mice were known to inhabit this unit during surveys in 1979 and 1982, and by 1986 this was the only known existing population of the subspecies (Humphrey and Barbour 1981, pp. 841–842; Holler et al. 1989, p. 398). This population was the donor site for the reestablishment of Perdido Key beach mice into Gulf Islands National Seashore in 1986. This project ultimately saved Perdido Key beach mice from extinction as the population at Gulf State Park was considered extirpated in 1998 due to tropical storms and predators (Auburn University 1999, p 1.9). Beach mouse habitat in this unit consists of primary, secondary, and scrub dune habitat (PCEs 2 and 3), and some areas of connectivity (PCE 4). Because scrub habitat is separated from the frontal dunes by a highway in some areas, the population inhabiting this unit can be especially vulnerable to hurricane impacts, and therefore, further linkage to scrub habitat or habitat management would improve connectivity. This unit is managed by the Alabama Department of Conservation and Natural Resources and provides a relatively natural light regime (PCE 5). Threats specific to this unit that may require special management considerations include artificial lighting, presence of feral cats and other predators at unnatural levels, and high recreational use that may result in soil compaction, damage to dunes, or other decrease in habitat quality.

**PKBM–2: West Perdido Key Unit** PKBM–2 consists of 114 ac (46 ha) in southern Escambia County, Florida, and 33 ac (13 ha) in southern Baldwin County, Alabama. This unit encompasses essential features of beach mouse habitat from approximately 1.0 mi (1.6 km) west of where the Alabama-Florida State line bisects Perdido Key east to 2.0 mi (3.2 km) east of the State line and areas from the MHWL north to the seaward extent of human development or maritime forest. This unit consists of private lands and ultimately includes essential features of beach mouse habitat between Gulf State Park (PKBM–1) and Perdido Key State Park (PKBM–3). Beach mouse habitat in this unit consists of primary, secondary, and scrub dune habitat (PCEs 2 and 3). Habitat fragmentation and other threats specific to this unit are mainly due to development. Consequently, threats to this unit that may require special management considerations include habitat fragmentation and habitat loss, artificial lighting, presence of feral cats as well as other predators at unnatural levels, excessive foot traffic and soil compaction, and damage to dune vegetation and structure. This area was not known to be occupied at the time of listing. Beach

mouse presence was confirmed in 2005 through observations of beach mouse burrows and tracks (Sneckenberger 2005), and this unit is adjacent to contiguous, occupied beach mouse habitat (PKBM-3). Therefore, we have determined this unit to be currently occupied. This unit is essential to the conservation of the species because it provides essential connectivity (PCE 4) between two more contiguous habitat patches (Perdido Key State Park and Gulf State Park), provides habitat for expansion, natural movements, and recolonization, and is therefore essential to the conservation of the species. Specifically, this unit may have historically provided for the recolonization of Gulf State Park (PKBM-1) or may facilitate similar recolonization in the future as the habitat recovers from recent hurricane events.

**PKBM-3: Perdido Key State Park Unit** PKBM-3 consists of 238 ac (96 ha) in southern Escambia County, Florida. This unit encompasses essential features of beach mouse habitat within the boundary of Perdido Key State Park from approximately 2.0 mi (3.2 km) east of the Alabama-Florida State line to 4.0 mi (6.4 km) east of the State line, and the area from the MHWL north to the seaward extent of the maritime forest. Beach mouse habitat in this unit consists of primary, secondary and scrub dune habitat (PCEs 2 and 3) and some areas of connectivity (PCE 4). Live-trapping efforts in this area were limited in the past. Perdido Key beach mice were known to inhabit this unit in 1979, though the population was impacted by Hurricane Frederic (1979) and no beach mice were captured during extensive surveys in 1982 and 1986 (Humphrey and Barbour 1981, pp. 841-843; Holler et al. 1989, p. 400); therefore, the unit was considered unoccupied at the time of listing. In 2000, a relocation program began to reestablish mice at Perdido Key State Park, and this project is considered a success. This unit is essential to the conservation of the species because it contains a reestablished population of beach mice that is needed for recovery. Improving or restoring habitat connections would increase habitat quality and provide more connectivity for dispersal, exploratory movements, and population expansion. This unit is managed by the Florida Park Service and provides a natural light regime (PCE 5). Threats specific to this unit that may require special management considerations include artificial lighting, presence of feral cats as well as other predators at unnatural levels, and high recreational use that may result in soil compaction, damage to dunes, or other decrease in habitat quality.

**PKBM-4: Gulf Beach Unit** PKBM-4 consists of 162 ac (66 ha) in southern Escambia County, Florida. This unit includes features essential to the conservation of beach mouse habitat between Gulf Islands National Seashore and Perdido Key State Park from approximately 4.0 mi (6.4 km) east of the Alabama-Florida State line to 6.0 mi (9.7 km) east of the State line and areas from the MHWL north to the seaward extent of human development or maritime forest. This unit consists of private lands. Beach mouse habitat in this unit consists of primary, secondary, and scrub dune habitat (PCEs 2 and 3). Habitat fragmentation and other threats specific to this unit are mainly due to development. Consequently, threats to this unit that may require special management considerations include habitat fragmentation and habitat loss, artificial lighting, presence of feral cats as well as other predators at unnatural levels, excessive foot traffic and soil compaction, and damage to dune vegetation and structure. While the unit was not known to be occupied at the time of listing, presence of beach mice has recently been confirmed as a result of live-trapping efforts in conjunction with permitting (Service 2004a). This unit includes high-elevation scrub habitat and serves as a refuge during storm events and as an important source population if storms extirpate or greatly reduce local populations. This unit is essential to the conservation of the species because it provides essential connectivity between two populations (Gulf Islands

National Seashore and Perdido Key State Park) and provides essential habitat for expansion, natural movements, and recolonization (PCE 4).

PKBM-5: Gulf Islands National Seashore Unit PKBM-5 consists of 638 ac (258 ha) in southern Escambia County, Florida, on the easternmost region of Perdido Key. This unit encompasses essential features of beach mouse habitat within the boundary of Gulf Islands National Seashore-Perdido Key Area (also referred to as Johnson Beach) from approximately 6.0 mi (9.7 km) east of the Alabama-Florida State line to the eastern tip of Perdido Key at Pensacola Bay and the area from the MHWL north to the seaward extent of the maritime forest. Beach mouse habitat in this unit consists mainly of primary and secondary dune habitat but provides the longest contiguous expanse of frontal dune habitat within the historic range of the Perdido Key beach mouse. Perdido Key beach mice were known to inhabit this unit in 1979, though the population was impacted by Hurricane Frederic (1979) and no beach mice were captured during extensive surveys in 1982 and 1986 (Humphrey and Barbour 1981, pp. 841–843; Holler et al. 1989, p. 400); therefore, the unit was considered unoccupied at the time of listing. In 1986, Perdido Key beach mice were reestablished at this unit as a part of Service recovery efforts. This reestablishment project was identified as the most urgent recovery need for the mouse (Service 1987, p. 12; Holler et al. 1989, p. 398). The project is considered a success. In 2000 and 2001, Perdido Key beach mice captured from this site served as donors to re-establish beach mice at Perdido Key State Park (PKBM- 3). PKBM-5, in its entirety, possesses all five PCEs and is essential to the conservation of the species because it provides habitat for a population that is needed for recovery. However, most of this unit consists of frontal dunes, making the population inhabiting this unit particularly threatened by storm events. Threats specific to this unit that may require special management considerations include artificial lighting, presence of feral cats and other predators at unnatural levels, and high recreational use that may result in soil compaction, damage to dunes, or other decrease in habitat quality. Gulf Islands National Seashore is managed by the National Park Service.

#### **Primary Constituent Elements/Physical or Biological Features**

Critical habitat units are designated for Escambia County, Florida, and Baldwin County, Alabama. The primary constituent elements of critical habitat for the Perdido Key beach mouse are the habitat components that provide:

- (i) A contiguous mosaic of primary, secondary, and scrub vegetation and dune structure, with a balanced level of competition and predation and few or no competitive or predaceous nonnative species present, that collectively provide foraging opportunities, cover, and burrow sites;
- (ii) Primary and secondary dunes, generally dominated by sea oats (*Uniola paniculata*), that despite occasional temporary impacts and reconfiguration from tropical storms and hurricanes, provide abundant food resources, burrow sites, and protection from predators;
- (iii) Scrub dunes, generally dominated by scrub oaks (*Quercus* spp.), that provide food resources and burrow sites, and provide elevated refugia during and after intense flooding due to rainfall and/or hurricane-induced storm surge;
- (iv) Functional, unobstructed habitat connections that facilitate genetic exchange, dispersal, natural exploratory movements, and re-colonization of locally extirpated areas; and

(v) A natural light regime within the coastal dune ecosystem, compatible with the nocturnal activity of beach mice, necessary for normal behavior, growth, and viability of all life stages.

**Special Management Considerations or Protections**

Critical habitat does not include man-made structures existing on the effective date of this rule and not containing one or more of the primary constituent elements, such as buildings, aqueducts, airports, driveways, and roads, and the land on which such structures are located.

The features essential to the conservation of beach mice in all of the areas designated may require special management considerations or protections due to threats to the subspecies or its habitat. Such management considerations and protections include: Management of non-native predators and competitors, management of non-native plants, protection of beach mice and their habitat from threats by road construction, urban and commercial development, heavy machinery, and recreational activities.

***Life History*****Feeding Narrative**

Adult: Eats fruits and seeds of dune plants, probably including such species as sea oats, bluestem, sea rocket, PANICUM, and PHYSALIS; feeds also on invertebrates (Moyers and Holler 1992) (NatureServe, 2015).

**Reproduction Narrative**

Adult: May breed all year. Much breeding activity occurs November-January. Numerous pregnant and/or lactating females have been found in spring and fall (Holler 1992). Produces 2 or more litters per year. Gestation averages 23-24 days (nonlactating) or 28-29 days (lactating). Litter size averages 3-4. (USFWS 1988). Young are weaned in about 18 days. Minimum age at conception is 5 weeks. Apparently monogamous mating system. (Kirkland and Layne 1989) (NatureServe, 2015).

**Spatial Arrangements of the Population**

Adult: Clumped (NatureServe, 2015)

**Environmental Specificity**

Adult: Narrow (inferred from NatureServe, 2015)

**Tolerance Ranges/Thresholds**

Adult: Low (inferred from NatureServe, 2015)

**Site Fidelity**

Adult: High (inferred from NatureServe, 2015)

**Habitat Narrative**

Adult: Dry, sandy, sparsely vegetated frontal coastal dunes of medium height, with no or very few secondary dunes lying inland; vegetation of inhabited dunes includes mainly sea oats and bluestem at moderate density; scrub dunes are lacking in presently occupied habitat (Holler 1992). Young are born in underground burrows (NatureServe, 2015). Narrow environmental

specificity, high ecological integrity, low tolerance range and high site fidelity are inferred based on the very specific habitat this species inhabits (NatureServe, 2015).

***Dispersal/Migration*****Motility/Mobility**

Adult: High (NatureServe, 2015)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory (NatureServe, 2015)

**Dispersal**

Adult: Low (inferred from NatureServe, 2015)

**Immigration/Emigration**

Adult: Unlikely (inferred from NatureServe, 2015)

**Dispersal/Migration Narrative**

Adult: Mice are highly mobile and NatureServe (2015) notes this species is non-migratory. Low dispersal and unlikely immigration/emigration are inferred based on species habitat and the isolated nature of the known populations (NatureServe, 2015).

***Population Information and Trends*****Population Trends:**

Increasing (NatureServe, 2015)

**Resiliency:**

Low inferred from (NatureServe, 2015; USFWS, 2014)

**Representation:**

Low inferred from (NatureServe, 2015; USFWS, 2014)

**Redundancy:**

Low inferred from (NatureServe, 2015; USFWS, 2014)

**Number of Populations:**

1 - 5 (NatureServe, 2105)

**Population Size:**

1-1000 total individuals (NatureServe, 2015)

**Population Narrative:**

NatureServe (2015) notes that there are between 1 and 5 populations and a total of between 1 and 1000 individuals. In addition NatureServe notes that the short-term trend for this species is increasing. Low resiliency, representation and redundancy are inferred based on limited number of populations and low numbers of individuals.

***Threats and Stressors***

**Stressor:** Human encroachment (USFWS, 2014)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Due to coastal development, from the PKBMs historic range of 16.9 miles of coastal dune habitat, and estimate of nine miles of habitat with moderate fragmentation remains. Less than 1,711 acres of PKBM habitat remains in its entirety, portions of which include heavily fragmented habitat on private lands (USFWS, 2014).

**Stressor:** Predation (USFWS, 2014)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** Beach mice have a number of natural non-native predators including the coachwhip, corn snake, pygmy rattlesnake, Eastern diamondback rattlesnake, short-eared and great-horned owl, great blue heron, northern-harrier, loggerhead shrike, gray fox, striped skunk, long-tailed weasel, raccoon, bobcat, ghost crabs, red fox, coyotes free roaming and feral cats (USFWS, 2014).

**Stressor:** Tropical storms and hurricanes (USFWS, 2014)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat and death of individuals

**Narrative:** Tropical storm events affect beach mouse population densities in various habitats (USFWS, 2014).

**Stressor:** Artificial lighting (USFWS, 2014)

**Exposure:**

**Response:**

**Consequence:** Increased predation/limits foraging

**Narrative:** Artificial lighting increases the risk of predation and influences beach mouse foraging patterns and natural movements as it increases their perceived risk of predation (USFWS, 2014).

**Stressor:** Sea level rise (USFWS, 2014)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Sea level rise is an increasing threat to PKBM and all other coastal dependent species based on numerous prediction models. Significant loss of PKBM habitat range wide becomes apparent around 6 ft. of sea level rise (USFWS, 2014).

***Recovery***

**Reclassification Criteria:**

The species will be considered for downlisting to threatened when there are 3 distinct, self-sustaining populations in each of the critical habitat areas, and a minimum of 50% of the critical habitat is protected and occupied by mice (USFWS, 1987).

**Delisting Criteria:**

The Perdido Key beach mouse will be considered for delisting when all the following criteria have been met: 1. Populations inhabiting all five (5) critical habitat units exhibit stable or increasing trends, evidenced by natural recruitment and multiple age classes (Factor A). 2. Habitat connectivity and genetic diversity shall be maintained throughout the range to a level that does not require translocations or captive breeding (Factors A and E). 3. All designated PKBM critical habitat under public ownership (Federal, State, and Local entities) is managed under a conservation mechanism that addresses beach mice (Factor A). 4. Non-native predator removal (specifically free-roaming/feral cats) shall be conducted to a degree that PKBM will remain viable for the foreseeable future (Factor C, D). 5. When, in addition to the above criteria, it can be demonstrated that habitat loss associated with climate change/sea-level rise and development are diminished such that enough suitable habitat remains in the foreseeable future for CBM to remain viable (Factor E) (USFWS 2019)

**Recovery Actions:**

- Protect habitat from further human encroachment. Conduct studies to determine optimal habitat needs and life history parameter for the three subspecies. Provide habitat protection on Federal and State-owned lands. Cooperate with landowners to protect privately owned habitat. Identify unprotected habitat important to beach mice, and take actions to protect it. Monitor activities planned for privately owned land (USFWS, 1987).
- Reestablish and/or supplement populations. Conduct genetic studies to estimate both degree of inbreeding and interrelatedness of the three subspecies. Identify areas where populations have been extirpated and need to be reestablished, or where existing populations show indications of loss of genetic variability and need to be supplemented. Identify populations from which mice may be removed for translocation or captive breeding. As appropriate, based on task 23, translocate beach mice directly in to predetermined areas. As appropriate, based on task 23, develop plans for captive breeding colonies of the three subspecies (USFWS, 1987).
- Develop an educational program for the public. Provide public with information about life history and distribution of beach mice. Inform public about need for careful sanitation around dwellings to reduce beach mouse predators. Seek public support in protecting dune vegetation, and in reporting violations of laws and regulations governing use of beaches and dunes. Urge close confinement of cats in vicinity of beach mouse populations (USFWS, 1987).
- Develop emergency procedures to provide protection to beach mouse habitat in case of off-shore oil spills (USFWS, 1987).

***Conservation Measures and Best Management Practices:***

- Additional Biologist. A second biologist position should be filled to aid in the identified recovery actions. Another biologist could assist in the heavy workload for all beach mice and allow for in-office coordination and consistency with Section 7 and 10 permitting aspects, monitoring and trapping, permit compliance, research, and recovery activities such as translocations and outreach. This position could be an entry level or student trainee position and should work under the lead Recovery biologist. Without such a position, few of the recommendations suggested can be accomplished for PKBM and other beach mouse subspecies (USFWS, 2015).

- Revise Recovery Plan. The recovery plan should be updated to define objective measurable criteria and better address the five factors (USFWS, 2015).
- Population and Habitat Assessment program. The track tube monitoring program has been developed since our last 5-year review and should continue to be implemented for PKBM. Funding for this action is critical. The development of a habitat mapping tool has also been initiated and will soon be peer reviewed and put to use to see landscape connectivity and potential dispersal routes on Perdido Key. An updated PVA needs to be done to estimate future population trends and the likelihood of extinction. This would be beneficial to do while the population is high (USFWS, 2015).
- Emergency Response Plan. A contingency plan to outline actions taken in case of severe threats to the persistence of PKBM (i.e., forecasted category 5 hurricane, feral cat population increase, population crash) (Traylor-Holzer and Lacy 2007) has been initiated since our last 5-year review (USFWS, 2015).
- Finalization and implementation is recommended. This plan is associated with the PKBM captive breeding program (USFWS, 2015).
- Land Acquisition. Appropriate parcels for land acquisition have been identified using LIDAR data (to identify high-elevation habitat) and current knowledge of PKBM movements and habitat use. The Service should keep this list as relevant as possible to the current landscape and needs of PKBM. Recently, Escambia County purchased a parcel with grant money from the Service. This practice should continue as intact coastal dune habitat is limited (USFWS, 2015).

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## SPECIES ACCOUNT: *Pteropus mariannus mariannus* (Mariana fruit bat, Mariana flying fox)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered 08/27/1984 downlisted to Threatened; 01/06/2005; Pacific Region (R1) (USFWS, 2016)

### **Physical Description**

The Mariana fruit bat is a medium-sized fruit bat in the family Pteropodidae that weighs 0.66 to 1.15 pounds. Males are slightly larger than females. The underside (abdomen) is black to brown with gray hair interspersed that creates a grizzled appearance. The shoulders (mantle) and sides of the neck are bright golden brown, but may be paler in some individuals. The head varies from brown to dark brown. The well-formed, rounded ears and large eyes give the face a canine appearance.

### **Current Range**

The Mariana fruit bat is a subspecies endemic to the Mariana archipelago (Guam and the CNMI), where it was historically present on every island except Uracas (Wiles et al. 1989, p. 69). The fruit bat is currently thought to be extirpated from Tinian (USFWS 2009c, pp. 269-272; USFWS 2014d, pp. 2-3).

### **Critical Habitat Designated**

Yes; 10/28/2004.

### **Legal Description**

On October 28, 2004, the U.S. Fish and Wildlife Service (Service) designated critical habitat for the Mariana fruit bat (*Pteropus mariannus mariannus*) pursuant to the Endangered Species Act, as amended (Act or ESA) (69 FR 62944 - 62990). Approximately 376 acres (ac) (152 hectares (ha)) on the island of Guam were designated for the Mariana fruit bat.

### **Critical Habitat Designation**

Lands designated as critical habitat for the Mariana fruit bat occurs in one unit on Guam. Designated critical habitat includes land under Federal, Commonwealth, and private ownership, with Federal lands being managed by the Department of the Interior. All of the designated critical habitat on Guam currently is occupied by the Mariana fruit bat.

This unit consists of approximately 376 ac (152 ha) of land in the fee simple portion of the Guam National Wildlife Refuge. The vegetation in this unit consists of coastal, limestone, and secondary forests composed of native and introduced plant species and contains the full range of primary constituent elements needed for the conservation of the Mariana fruit bat. This area is important because it contains areas used for foraging by the only known Mariana fruit bat colony on Guam. This area also contains roosting and foraging sites used by bats since 1981 (see Wiles et al. 1995 for details). This unit also encompasses essential conservation areas identified in the Mariana fruit bat recovery plan (USFWS 1990a).

### **Primary Constituent Elements/Physical or Biological Features**

The critical habitat unit for the Mariana fruit bat is designated for the Territory of Guam. Within this area, the primary constituent elements required by the Mariana fruit bat for the biological needs of foraging, sheltering, roosting, and rearing of young are found in areas supporting limestone, secondary, ravine, swamp, agricultural, and coastal forests composed of native or introduced plant species. These forest types provide the primary constituent elements of:

(i) Plant species used for foraging, such as *Artocarpus* sp. (breadfruit), *Carica papaya* (papaya), *Cycas circinalis* (fadang), *Ficus* spp. (fig), *Pandanus tectorius* (kafu), *Cocos nucifera* (coconut palm), and *Terminalia catappa* (talisai); and

(ii) Remote locations, often within 328 ft (100 m) of clifflines that are 260 to 590 ft (80 to 100 m) tall, with limited exposure to human disturbance; land that contains mature fig, *Mammea odorata* (chopak), *Casuarina equisetifolia* (gago), *Macaranga thompsonii* (pengua), *Guettarda speciosa* (panao), *Neisosperma oppositifolia* (fagot), and other tree species that are used for roosting and breeding.

### **Special Management Considerations or Protections**

Critical habitat does not include existing features and structures within the boundaries of the mapped units, such as buildings, roads, aqueducts, antennas, water tanks, agricultural fields, paved areas, lawns, and other urban landscaped areas not containing one or more of the primary constituent elements.

Excluded from designation (see “Exclusions from Critical Habitat”) are 10,838 ac (4,386 ha) of Air Force lands, 7,977 ac (3,228 ha) of Navy lands, 2,989 ac (1,210 ha) of Government of Guam lands, and 1,941 ac (785 ha) of private lands in northern and southern Guam that were proposed as critical habitat in the October 15, 2002, proposed rule (67 FR 63738), leaving a final designation of 376 ac (816 ha). Although Air Force, Navy, Government of Guam, and private lands are excluded from final critical habitat designation, they still contribute to the conservation of the Mariana fruit bat.

### ***Life History***

#### **Feeding Narrative**

Adult: The diet of the fruit bat is comprised of fruits, nectar, pollen, and some leaves (Wiles and Fujita 1992, pp. 26-31; Wiles and Johnson 2004, p. 591

#### **Reproduction Narrative**

Adult: Within colonies, fruit bats typically group themselves into harems (one male and 2-15 females) or bachelor groups (predominantly males; Wiles 1987a, pp. 93-94; J. Boland, unpubl. data). Unlike most *Pteropus* species, mating and the presence of nursing young have been observed in Mariana fruit bats throughout the year on Guam and Rota (Wiles 1987a, pp. 93-94; CNMI 2010, p. 12; CNMI 2011, p. 12; J. Boland, unpubl. data). Data is limited for age of sexual maturity, reproductive rates, length of gestation, and lifespan of Mariana fruit bats. Female bats of the family Pteropodidae generally have a gestation period of 4.6- 6.3 months and one offspring per year (Pierson and Rainey 1992, pp. 1-17). Many *Pteropus* species typically do not give birth before 18 months of age (Pierson and Rainey 1992, pp. 1-17; McIlwee and Martin 2002, p. 76). Based on these reproductive traits, several authors have suggested that *Pteropus*

bats have a low maximum population growth rate and thus a slow rate of recovery when a population is diminished (Pierson and Rainey 1992, p. 1-17; McIlwee and Martin 2002, p. 76).

**Habitat Narrative**

Adult: It uses several forest types for foraging, roosting, and breeding, including native primary and secondary limestone forest, volcanic (or ravine) forest, old coconut plantations, and groves of *Casuarina equisetifolia* (Glass and Taisacan 1988, pp. 6–13; Worthington et al. 2001, pp. 137–138; Wiles and Johnson 2004, pp. 589–591). Most fruit bats roost during the day in maternity colonies at sites to which they show a high level of fidelity (unless disturbed). A small proportion of fruit bats, usually males, roost alone or in small groups called bachelor colonies.

***Dispersal/Migration*****Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory

***Population Information and Trends*****Population Trends:**

Stable or declining

**Population Size:**

~ 6,000

**Population Narrative:**

The total population of the Mariana fruit bat is estimated to be approximately 6,000 animals (USGS 2010, p. 36; CNMI 2011, p. 6). Surveys suggest populations are stable or declining throughout most of their range (Table 5). A notable exception to the declining trend is the island of Rota, where the population has increased since 2008 (CNMI 2008, p. 11; CNMI 2011, p. 6). The population increase on Rota is due to a recent decrease in illegal hunting at roost sites of fruit bat maternity colonies, and the decrease in illegal hunting can be attributed to an increase in enforcement of wildlife regulations that began in 2009 (CNMI 2010, pp. 7-9). The fruit bat population on Rota is estimated at approximately 2600 (CNMI 2011; p. 6). Although comprehensive surveys have not been conducted on Saipan, there have been no confirmed observations of maternity colonies in recent years, and the island-wide population is expected to be less than 50 individuals (T. Willsey, CNMI DLNR, pers. comm. 2014). The population of fruit bats on Guam is estimated to be less than 30 bats (SWCA 2013, pp. 19-22; DON 2013b, pp. 11-15). The most recent and last colony to exist on Guam was at Pati Point, but recent surveys indicate that this colony no longer exists (Figure 5) (SWCA 2013, pp. 13). On July 3, 2014, a survey was conducted on Andersen Air Force Base which resulted in 10 observations of bats; analyses are still in progress to determine duplicate observations and detection probability given the amount of area surveyed on the Base (DON 2014c).

***Threats and Stressors***

**Stressor:** Loss or degradation of habitat

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** • Loss or degradation of habitat: o Human development is a factor in habitat loss on all inhabited southern islands and on northern islands with military activity. O Feral ungulates and Philippine sambar deer (*Rusa marianna*) degrade habitat on many of the Mariana Islands. The successful eradication of feral ungulates from Sarigan and Anatahan suggests that similar projects may succeed on other islands. However, once grazing and browsing pressure is removed, the potential invasion of native forest by alien plants may be a more difficult and long-term recovery issue.

**Stressor:** Human disturbance

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** • Human disturbance: o Illegal hunting is a threat to Mariana fruit bats throughout its range. Although law enforcement activity has increased since 2009 (CNMI 2008, 2009a-b, 2010), illegal hunting of fruit bats on Rota continues and will likely resume to historical levels unless consistent, effective law enforcement efforts in tandem with education and outreach programs continue. Fruit bats appear to be extirpated from Tinian and are declining on Saipan and Guam, and illegal hunting is thought to have greatly contributed to the decimation/decline of those populations (Wiles and Payne 1986; Wiles and Glass 1990; Sheeline 1991; Stinson et al. 1992; Wiles 1992; Esselstyn et al. 2006). As with Rota, recovery of the fruit bat on human-inhabited islands will not likely be possible without strong education programs combined with effective control of illegal hunting.

**Stressor:** Nonnative snake predation

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** • Nonnative snake predation – The brown treesnake is thought to prey on non-volant young left at the roost during the night, thus preventing the recruitment of young bats into the breeding population. Effective control of brown treesnakes must be achieved before fruit bat population on Guam can recover. The interdiction, control, and ultimate eradication of brown treesnakes in the archipelago are the focus of major, ongoing projects, and the fruit bat is likely to benefit from these efforts in the long term. This prognosis would change drastically if the brown treesnake were to become established widely throughout the archipelago.

**Stressor:** Stochastic events

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** • Stochastic events – Typhoons and volcanic eruptions result in mortality, reduced population viability, and habitat loss. Natural disasters can be especially damaging to the viability of smaller fruit bat populations (e.g., on Guam, Saipan, Aguiguan, and Maug). The significant loss of habitat on Anatahan after the volcanic eruption in 2003 resulted in the loss of a substantial fruit bat population that has not yet recovered.

**Recovery**

**Delisting Criteria:**

Before the Mariana fruit bat is considered for delisting, the Service proposes that stable or increasing populations should exist on three of the five southern islands (Saipan, Tinian, Aguiguan, Rota, and Guam), and six of the northern islands where Mariana fruit bats have persisted historically (Anatahan, Sarigan, Guguan, Alamagan, Pagan, Agrihan, Asuncion, and Maug; USFWS 2009d, pp. 37-39). Of the six northern islands that require stable or increasing fruit bat numbers, two of these must include Pagan, Anatahan, or Agrihan. Since publication of the draft revised recovery plan in 2009, new information on the Mariana fruit bat has resulted in changes to how we look at recovery for the species. We now consider recovery in terms of stable or increasing subpopulations of sufficient size distributed across Guam and the Mariana Islands. To meet recovery objectives, stable or increasing fruit bat subpopulations should at a minimum be distributed on the islands that currently have extant populations (USFWS in review). The final version of the Mariana fruit bat recovery plan is currently in review, and recovery criteria stated here may change upon completion of the final plan. Of the six northern islands, the only evidence for a possibly increasing population is on Asuncion (USGS 2010, p. 33). Of the five southern islands, only Rota has achieved an increasing population. Although a conservation area containing some important habitat for fruit bats was recently established on Rota (USFWS 2011, pp. 1), there is not currently enough protected fruit bat habitat on Rota, Guam, Tinian, or Saipan to support substantial population recovery on any of those islands. Even if sufficient habitat is set aside in conservation to support recovery of populations, controlling illegal hunting may continue to be a challenge that limits recovery of the species.

**Recovery Actions:**

- New management actions (adapted from USFWS 2014d, pp. 4-5):
  - Monitoring and analysis of population viability – Technical assistance was obtained in 2008 to analyze fruit bat survey data from Rota and refine survey methods and the existing monitoring program (CNMI 2008, 2009a-b, 2010).
- Law enforcement and compliance – On the island of Rota, the Service and the CNMI Division of Fish and Wildlife (DFW) have increased law enforcement actions since 2009. With support from Service law enforcement and federal discretionary funds, CNMI Conservation Officers have participated in nine fruit bat-related arrests on Rota, all resulting in convictions. Enforcement actions have contributed to a decrease in illegal hunting, and approximate doubling of the fruit bat population on Rota.
- Development of monitoring protocol – Experts were consulted to review and refine survey methods for fruit bats to develop standardized, quantitative monitoring that permits data comparison at multiple timescales. Standard operating procedures were developed for CNMI DFW (CNMI 2009a) and a monitoring protocol was developed for Service for fruit bat surveys in the Northern Mariana Islands (Mildenstein and Boland 2010).
- Habitat and natural process management and restoration – The Mariana Crow Conservation Area was established on Rota through an MOA between the CNMI and the Service (USFWS 2011). This area encompasses 444 hectares (1097 acres) and contains some high-quality foraging and roosting habitat for fruit bats.
- Outreach and education – Discussions were initiated with CNMI DFW and CNMI Public School System (PSS) to develop outreach and education materials and opportunities to curb illegal hunting. Several education and outreach programs were funded by the Service, Bat Conservation International, Disney, and Lube Bat Conservancy, and these programs were implemented on Rota through a local non-profit. An education curriculum was developed with the CNMI PSS, but has not yet been implemented.

***Conservation Measures and Best Management Practices:***

- • Outreach and education – Decrease illegal hunting by developing and supporting outreach and education programs that emphasize the value of and need to protect fruit bats and other native plant and wildlife species in the Marianas.
- • Law enforcement and compliance – Decrease illegal hunting by continuing to provide technical and financial assistance to CNMI DFW enforcement officers to facilitate apprehension and prosecution of poachers.
- • Ungulate monitoring and control
  - o Decrease habitat loss by eradicating feral ungulates on islands where they exist, and preventing their introduction on other islands where fruit bat recovery is desired.
  - o Decrease habitat loss by controlling deer in areas of high-quality fruit bat habitat.
- • Habitat and natural process management and restoration
  - o Improve habitat through support of native forest restoration, especially on Guam, Saipan, and Tinian.
  - o Set aside enough high-quality habitat including in-perpetuity protection of conservation areas to support the recovery of fruit bat populations on three of the five southern islands.
- • Human interaction monitoring and management
  - o Limit military training in areas occupied by fruit bats to activities that will not disturb fruit bats or their habitat.
  - o Limit urban development in areas occupied by or potentially used for roosting and foraging by fruit bats.
- • Population monitoring and viability analysis – Continue monitoring fruit bat numbers on Anatahan to understand the fluctuation of numbers in response to volcanic activity.
- • Population monitoring and viability analysis – Hire and ensure consistent employment of a full-time, resident DFW or Service biologist who is charged with monitoring the fruit bat population on Rota according to established protocols (CNMI 2009a, Appendix 1).
- • Predator / herbivore monitoring and control
  - o Development and implementation of large-scale, long-term methods for brown treesnake control that will reduce the brown treesnake population on a landscape level on Guam.
  - o Continue and increase efforts to prevent introduction of brown treesnake populations on other Mariana Islands.

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Designation of Critical Habitat for the Mariana Fruit Bat and Guam Micronesian Kingfisher on Guam and the Mariana Crow on Guam and in the Commonwealth of the Northern Mariana Islands

Final Rule. 69 FR 62944 - 62990. October 28, 2004.

## SPECIES ACCOUNT: *Pteropus tokudae* (Little Mariana fruit Bat)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered/Presumed extinct; Proposed for Delisting

#### **Physical Description**

The little Mariana fruit bat is much smaller than the Mariana fruit bat and adults have body measurements of: head-body length, 140 to 151 mm; forearm length, 94 to 95 mm; wingspan, 650 to 709 mm; and body weight, 152 gr (Tate 1934, Perez 1972, K. Koopman, 1985 ). The abdomen and wings are brown to dark brown but with few whitish hairs present (Tate 1934). The mantle and sides of the neck vary from brown to pale gold. The top of the head is grayish to yellowish brown while the throat and chin are dark brown (USFWS, 1990)

#### **Taxonomy**

The little Mariana fruit bat was first described as the species *Pteropus tokudae* by Tate (1934) (USFWS, 2009).

#### **Historical Range**

Both species of fruit bat probably once occurred throughout Guam in forested areas that formerly occupied most of the island (USFWS, 1990). Estimates of historical populations are not available, In 1920, Crampton (1921) reported bats to be “not an uncommon” sight as they flew over forest during the daytime.

#### **Current Range**

This species was endemic to Guam, but has not been observed since 1968 and is now thought to be extinct. Individuals were never collected for captive breeding, and therefore recovery is not possible (USFWS, 2015).

#### **Critical Habitat Designated**

No;

#### ***Life History***

#### **Feeding Narrative**

Adult: The fruit bat feeds on a wide variety of plant material but is primarily frugivorous (Marshall 1983, 1985). Wiles (1983b) gathered information on the foods of Mariana fruit bats by direct observations, finding food remains (discarded fruit, chewed pellets of fruit pulp) of bats feeding, examining feces, interviewing island residents, and reviewing published accounts (Safford 1905, 1910, Linsley 1934, Stone 1970, Perez 1972, Wheeler 1979a). Twenty-two species of plants are known to be used as food sources by fruit bats in the Marianas. These include fruit of 17 species, flowers of 7 species and the leaf of 1 species. It appears that favored foods include the fruits of breadfruit (*Artocarpus mariannensis* and *A. altilis*), papaya (*Carica a a a fadan C cas circinalis*), figs, kafu (*Pandanus tectorius*) and talisai (*Terminalia catappa*) and the flowers of kapok (*Ceiba pentandna*), coconut (*Cocos nucifera*), and gaogao (*Erythrina varie~ata*). The stems of leaves and tips of small twigs on breadfruit are also often eaten. Fruit bats have also been observed to feed on fagot and da’ok (*Calophyllum inophyllum* ) on other islands. Both of these plants are found on Guam (USFWS, 1990).

**Reproduction Narrative**

Adult: The only information available on breeding in the little Mariana fruit bat was gathered from a single female shot by hunters on June 5, 1968 (Perez 1972, 1985). This animal was with a juvenile that was able to fly away (USFWS, 1990).

**Habitat Narrative**

Adult: Only a single reference exists on the habitat used by little Mariana fruit bats. The animal killed by hunters in 1968 was taken below Tarague Point (Perez 1972) in an area of mature limestone forest (USFWS, 1990).

***Dispersal/Migration******Population Information and Trends*****Population Trends:**

No information found - Thought to be extinct (USFWS, 2009)

***Threats and Stressors***

**Stressor:** Habitat destruction (USFWS, 2009)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The documented threats to the Mariana fruit bat were considered likely to have affected the little Mariana fruit bat similarly. While large stands of relatively intact native forest can still be found on military lands and in the rugged interior areas of northern and southern Guam, some of these areas may be further fragmented and degraded by development activities and road building in the coming years (U.S. Air Force 2006b; Daleno 2007; U.S. Navy 2007a, b). Much of the remaining forest also has been severely degraded by Philippine deer (*Cervus mariannus*), feral pigs (*Sus scrofa*), and feral Asiatic water buffalo (*Bubalus bubalis*), all of which were introduced to Guam in the 1600s and 1700s (Conry 1988; Wiles et al. 1999). These introduced ungulates cause significant damage to native vegetation on Guam by consuming seeds, fruits, and foliage, ingesting or trampling seedlings, and promoting the spread of introduced weeds (Wiles et al. 1999; Wiles 2005). Philippine deer and feral pigs are found throughout Guam. On Andersen Air Force Base, densities of Philippine deer and feral pigs were estimated at 1.8 deer per hectare (0.8 deer per acre) and 0.4 pigs per hectare (0.2 pigs per acre), some of the highest ungulate densities recorded in the world (Knutson and Vogt, unpubl. manuscript 2003). Feral Asiatic water buffalo are found predominately on the Ordnance Annex and surrounding non-Navy lands in southern Guam, where the population is estimated to number fewer than 100 animals (A. Brooke, U.S. Navy, pers. comm. 2008). Efforts to control Asiatic water buffalo on Navy lands have been underway since 1996 and the population has been reduced from approximately 300 animals to 100 animals (A. Brooke, pers. comm. 2008). The Navy has also been working on developing an ungulate management plan for deer and pigs (A. Brooke, pers. comm. 2008). In addition, the Air Force is proposing to fence approximately 254 hectares (628 acres) from pig and deer incursions and to remove ungulates from these areas to offset impacts associated with two projects on Andersen Air Force Base (U.S. Air Force 2006a, b).

However, additional work is still needed to help offset the impact of these species on the remaining forests (USFWS, 2009).

**Stressor:** Predation (USFWS, 2009)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** By 1988, the brown treesnake had eliminated most of the native birds on the island (Wiles et al. 2003), as well as many other native and exotic animal species (Fritts and Rodda 1998). All but two of Guam's native bird species (the yellow bittern [*Ixobrychus sinensis*] and Mariana swiftlet [*Collocalia bartschi*]) have shown patterns of decline coinciding with the expansion of the snake's range across the island. These patterns of decline indicated an inverse relationship between populations of snakes and birds (Savidge 1987), presumably due to nest predation by brown treesnakes. Brown treesnakes are thought to prey on non-volant young Mariana fruit bats (*Pteropus mariannus mariannus*) (Wiles 1987, Wiles et al. 1995; Wiles 1996); treesnakes may have played a role in the decline of the little Mariana fruit bat (USFWS, 2009).

### ***Recovery***

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## SPECIES ACCOUNT: *Puma (=Felis) concolor coryi* (Florida panther)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; Southeast Region (R4) (USFWS, 2015)

#### **Physical Description**

An adult Florida panther is unspotted and typically rusty reddish-brown on the back, tawny on the sides, and pale gray underneath. Adult males can reach a length of 7 feet (ft) (2.1 meters [m]) from their nose to the tip of their tail and may exceed 161 pounds (lbs) (73 kilograms [kg]) in weight. However, adult males usually average around 116 lbs (52.6 kg) and are about 24 to 28 inches (in) (60 to 70 centimeters [cm]) tall at shoulder height (Roelke 1990). Female panthers are smaller with an average weight of 75 lbs (34 kg) and length of 6 ft (1.8 m) (Roelke 1990). Florida panther kittens are gray with dark brown or blackish spots and five bands around the tail. The spots gradually fade as the kittens grow older and are almost unnoticeable by the time they are 6 months old. At this age, their bright blue eyes slowly turn to the light-brown straw color of the adult (Belden 1988). Three external characteristics: a right angle crook at the terminal end of the tail, a whorl of hair or cowlick in the middle of the back, and irregular, white flecking on the head, nape, and shoulders – not found in combination in other subspecies of *Puma* (Belden 1986), were commonly observed in Florida panthers through the mid-1990s. The kinked tail and cowlicks were considered manifestations of inbreeding (Seal 1994); whereas the white flecking was thought to be a result of scarring from tick bites (Maehr 1992; Wilkins et al. 1997). Four other abnormalities prevalent in the panther population prior to the mid-1990s were cryptorchidism (one or two undescended testicles), low sperm quality, atrial septal defects (the opening between two atria in the heart fails to close normally during fetal development), and immune deficiencies; and these were suspected to be the result of low genetic variability (Roelke et al. 1993).

#### **Taxonomy**

The Florida panther was first described by Charles B. Cory in 1896 as *Felis concolor floridana* (Cory 1896). The type specimen was collected in Sebastian, Florida. Bangs (1899), however, believed the Florida panther was restricted to peninsular Florida and could not intergrade with other *Felis* sp. Therefore, he assigned it full specific status and named it *Felis coryi* since *Felis floridana* had been used previously for a bobcat (*Lynx rufus*). Culver et al. (2000) examined genetic diversity within and among the described subspecies of *Puma concolor* using three groups of genetic markers and proposed a revision of the genus to include only six subspecies, one of which encompassed all puma in North America including the Florida panther. They determined the Florida panther was one of several smaller populations that had unique features. Specifically, the number of polymorphic microsatellite loci and amount of variation were lower, and it was highly inbred. The degree to which the scientific community accepted the results of Culver et al. (2000) and the proposed change in taxonomy is not resolved (Service 2008). The Florida panther remains listed as a subspecies, and continues to receive protection pursuant to the Act.

#### **Historical Range**

See Current

#### **Current Range**

Although generally considered unreliable, sightings of panthers regularly occur throughout the southeast. Nonetheless, a reproducing population of panthers has not been documented to occur outside of south Florida for at least 30 years despite an extensive search effort (Belden et al. 1991; McBride et al. 1993; Clark et al. 2002). Survey reports and more than 70,000 locations of radio-collared panthers recorded between 1981 and 2004 clearly define the panther's current breeding range. Reproduction is known only in the Big Cypress Swamp and Everglades physiographic region in Collier, Lee, Hendry, Miami-Dade, and Monroe Counties, south of the Caloosahatchee River (Belden et al. 1991). There is no evidence of female panthers or successful panther reproduction currently occurring north of the Caloosahatchee River (Nowak and McBride 1974; Belden et al. 1991; Land and Taylor 1998; Land et al. 1999; Shindle et al. 2000; McBride 2002; Belden and McBride 2005). In 1973, McBride captured one female in Glades County (Nowak and McBride 1974). This was the last time a female panther was identified north of the Caloosahatchee River.

**Distinct Population Segments Defined**

No. (USFWS, 2015)

**Critical Habitat Designated**

No;

***Life History*****Feeding Narrative**

Adult: Primary panther prey species are white-tailed deer and feral hog (*Sus scrofa*) (Maehr et al. 1990a; Dalrymple and Bass 1996). Generally, feral hogs constitute the greatest biomass consumed by panthers north of the Alligator Alley section of I-75, while white-tailed deer are the greatest biomass consumed to the south (Maehr et al. 1990a). Secondary prey species include raccoons (*Procyon lotor*), nine-banded armadillos (*Dasypus novemcinctus*), marsh rabbits (*Sylvilagus palustris*) (Maehr et al. 1990a), and American alligators (*Alligator mississippiensis*) (Dalrymple and Bass 1996). No seasonal variation in diet has been detected. Maehr et al. (1990a) rarely observed domestic livestock in scats or kills of the Florida panther, although cattle were readily available in the study area. Little information on the feeding frequency of the Florida panther is available. However, the feeding frequency of the Puma is likely similar to the feeding frequency of the Florida panther. Ackerman et al. (1986) reported a resident adult male puma generally consumes one deer-sized prey every 8 to 11 days. Moreover, a female puma will consume one deer-sized prey item every 14 to 17 days for a resident female and one deer-sized prey item every 3.3 days for a female with three 13-month-old kittens.

**Reproduction Narrative**

Adult: Survivorship and causes of mortality – Benson et al. (2009) analyzed survival and cause-specific mortality of subadult and adult Florida panthers. They found sex and age influenced panther survival, as females survived better than males, and older adults (=10 years) survived poorly compared with younger adults. Genetic ancestry strongly influenced annual survival of subadults and adults after introgression, as F1 generation admixed panthers survived longer than pre-introgression panthers and non-F1 admixed individuals (Benson et al. 2009). Female panthers are considered adult residents if they are older than 18 months, have established home ranges, and have bred (Maehr et al. 1991). Land et al. (2004) reported 23 of 24 female

panthers first captured as kittens survived to become residents and 18 (78.3 percent) produced litters; 1 female was too young to determine residency. Male panthers are considered adult residents if they are older than 3 years and have established a home range that overlaps with females. Thirty-one (31) male panthers were captured as kittens and 12 (38.7 percent) of these cats survived to become residents (Jansen et al. 2005). "Successful male recruitment may depend on the death or home range shift of a resident adult male" (Maehr et al. 1991). Turnover in the breeding population is low with documented mortality in radio-collared panthers being greatest in subadult and non-resident males (Maehr et al. 1991; Shindle et al. 2003). Den sites of female panthers have been visited since 1992 and the kittens tagged with passive integrated transponder chips. Annual survival of these kittens has been determined to be  $0.328 \pm 0.072$  (standard error) (Hostetler et al. 2009). There was no evidence survival rate differed between male and female kittens or was influenced by litter size. Hostetler et al. (2009) found kitten survival generally increased with degree of admixture with introduced Texas pumas and decreased with panther abundance. Kitten survival is lowest during the first 3 months of their lives (Hostetler et al. 2009). Mortality records have been kept by the FWC beginning on February 13, 1972, for uncollared panthers and February 10, 1981, for radio-collared panthers. Through January 16, 2015, 404 mortalities have been documented. Vehicle strikes are the leading cause of documented panther mortalities. A total of 225 Florida panthers have been hit by vehicles (FWC 2013). These collisions resulted in 216 panther fatalities and 9 non-fatal injuries. The number of panther/ vehicle collisions recorded per year is positively correlated with the annual panther count (McBride et al. 2008). Intraspecific aggression is also a significant cause of panther mortality for panthers (70 documented cases) and is more common for males than females (Benson et al. 2009). However, instances where a male kills a female have been documented (Benson et al. 2009). In most cases the defense of a territory is the most likely cause for the intraspecific aggression, although the defense of kittens or a kill is also a suspected cause (Shindle et al. 2003).

#### **Habitat Narrative**

Adult: Noss and Cooperrider (1994) considered the landscape implications of maintaining viable panther populations. Assuming a male home range size of 137,599 ac (55,685 ha) (Maehr 1990), an adult sex ratio of 50:50 (Anderson 1983), and some margin of safety, they determined a reserve network as large as 15,625 to 23,438 mi<sup>2</sup> (40,469 to 60,703 km<sup>2</sup>) would be needed to support an effective population size of 50 individuals (equating to an actual adult population of 100 to 200 panthers [Ballou et al. 1989]). However, to provide for long-term persistence based on an effective population size of 500 individuals (equating to 1,000 to 2,000 adult panthers [Ballou et al. 1989]), could require as much as 156,251 to 234,376 mi<sup>2</sup> (404,687 to 607,031 km<sup>2</sup>). This latter acreage corresponds to roughly 60 to 70 percent of the Florida panther's historical range. Although it is uncertain whether this much land is needed for panther recovery, it does provide some qualitative insight into the importance of habitat conservation across large landscapes for achieving a viable panther population (Noss and Cooperrider 1994). Radio-collar data and ground tracking indicate that panthers use the mosaic of habitats available to them as resting and denning sites, hunting grounds, and travel routes. The majority of telemetry locations (Belden 1986; Belden et al. 1988; Maehr 1990; Maehr et al. 1991; Maehr 1992; Smith and Bass 1994; Kerkhoff et al. 2000; Comiskey et al. 2002; Cox et al. 2006; Kautz et al. 2006; Land et al. 2008) and natal den sites (Benson et al. 2008) were within or close to forested cover types, particularly cypress swamp, pinelands, hardwood swamp, and upland hardwood forests. Global Positioning System data has shown panthers (n = 12) use all habitats contained within their home ranges by selecting for forested habitat types and using all others

in proportion to availability (Land et al. 2008). Kautz et al. (2006) found the smallest class of forest patches (i.e., 9 to 26 ac [3.6 to 10.4 ha]) were the highest ranked forest patch sizes within panther home ranges. The diverse woody flora of forest edges probably provides cover suitable for stalking and ambushing prey (Belden et al. 1988; Cox et al. 2006). Also, dense understory vegetation comprised of saw palmetto provides some of the most important resting and denning cover for panthers (Maehr 1990; Benson et al. 2008). Shindle et al. (2003) estimated 73 percent of panther dens were in saw palmetto thickets. Even though some suitable panther habitat remains in south-central Florida, it is widely scattered and fragmented (Belden and McBride 2005). Thatcher et al. (2006) used a statistical model in combination with a geographic information system (GIS) to develop a multivariate landscape-scale habitat model based on the Mahalanobis distance statistic (D2) to evaluate habitats in south central Florida for potential expansion of the Florida panther population. They identified four potential habitat patches: the Avon Park Bombing Range area, Fisheating Creek/Babcock-Webb Wildlife Management Area (WMA), eastern Fisheating Creek, and the Duette Park/ Manatee County area. These habitat patches are smaller and more isolated compared with the current Florida panther range, and the landscape matrix where these habitat patches exist provides relatively poor habitat connectivity among the patches (Thatcher et al. 2006, 2009). Major highways and urban or agricultural development isolate these habitat patches, and they are rapidly being lost to the same development that threatens southern Florida (Belden and McBride 2005).

### ***Dispersal/Migration***

#### **Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory

#### **Dispersal**

Adult: Disperse over wide areas to establish home range

#### **Dispersal/Migration Narrative**

Adult: Dispersal – Panther dispersal begins after a juvenile becomes independent from its mother and continues until it establishes a home range. Dispersal distances are greater for males than females. The maximum dispersal distance recorded for a young male was 139.2 mi (224.1 km) over a 7-month period followed by a secondary dispersal of 145 mi (233 km). Comiskey et al. (2002) found males disperse an average distance of 25 mi (40 km) and females typically remain in or disperse short distances from their natal ranges. Female dispersers are considered philopatric because they usually establish home ranges less than one average home range width from their natal range (Maehr et al. 2002a). Maehr et al. (2002a) reported all female dispersers (n = 9) were successful at establishing a home range whereas only 63 percent of males (n = 18) were successful. Dispersing males usually go through a period as transient (non-resident) subadults, moving through the fringes of the resident population and often occupying suboptimal habitat until an established range becomes vacant (Maehr 1997). Most panther dispersal occurs south of the Caloosahatchee River. However, panthers have been documented north of the Caloosahatchee River many times since February 1972 through field signs (e.g., tracks, urine markers, scats), camera-trap photographs, carcasses from vehicle-related mortalities, telemetry from radio-collared animals (Land and Taylor 1998; Land et al. 1999; Shindle et al. 2000; Maehr et al. 2002b; Belden and McBride 2005), captured animals (one of which was radio collared), and one skeleton. The Caloosahatchee River, a narrow (295-328 ft [90-100 m]), channelized river, is probably not a significant barrier to panther movements.

Western subspecies of *Puma* are known to cross wide, swift-flowing rivers up to a mile in width (Seidensticker et al. 1973; Anderson 1983). However, the combination of the river, SR 80, and land uses along the river seems to have somewhat restricted panther dispersal northward (Maehr et al. 2002b). Documented physical evidence of at least 15 uncollared male panthers has been confirmed north of the river since 1972, but neither female panthers nor reproduction have been documented in this area since 1973 (Belden and McBride 2005).

**Home range dynamics and movements** – Panthers require large areas to meet their needs. Numerous factors influence panther home range size, including: habitat quality, prey density, and landscape configuration (Belden 1988; Comiskey et al. 2002). Home range sizes of six radio-collared panthers monitored between 1985 and 1990 averaged 128,000 ac (51,800 hectares [ha]) for resident adult males and 48,000 ac (19,425 ha) for resident adult females; transient males had a home range of 153,599 ac (62,160 ha) (Maehr et al. 1991). Comiskey et al. (2002) examined the home range size for 50 adult panthers (residents greater than 1.5 years old) monitored in south Florida from 1981 to 2000 and found resident males had a mean home range of 160,639 ac (65,009 ha) and females had a mean home range of 97,920 ac (39,627 ha). Beier et al. (2003) found home range size estimates for panthers reported by Maehr et al. (1991) and Comiskey et al. (2002) to be reliable. Annual minimum convex polygon home range sizes of 52 adult radio-collared panthers monitored between 1998 and 2002 ranged from 15,360 to 293,759 ac (6,216 to 118,880 ha), averaging 89,600 ac (36,260 ha) for 20 resident adult males and 44,160 ac (17,871 ha) for 32 resident adult females (Land et al. 1999, 2002; Shindle et al. 2000, 2001). The most current estimate of home-range sizes (minimum convex polygon method) for established, non-dispersing, adult, radio-collared panthers averaged 29,056 ac (11,759 ha) for females (n = 11) and 62,528 ac (25,304 ha) for males (n = 11) (Lotz et al. 2005). The average home range was 35,089 ac (14,200 ha) for resident females (n = 6) and 137,143 ac (55,500 ha) (n = 5) for males located at Big Cypress National Preserve (BICY) (Jansen et al. 2005). Home ranges of resident adults tend to be stable unless influenced by the death of other residents. Activity levels for Florida panthers are greatest at night with peaks around sunrise and after sunset (Maehr et al. 1990b). The lowest activity levels occur during the middle of the day. Female panthers at natal dens follow a similar pattern with less difference between high and low activity periods. Telemetry data indicate panthers typically do not return to the same resting site day after day, with the exception of females with dens or panthers remaining near kill sites for several days. The presence of physical evidence such as tracks, scats, and urine markers, confirms panthers move extensively within home ranges, visiting all parts of the range regularly in the course of hunting, breeding, and other activities (Maehr 1997; Comiskey et al. 2002). Males travel widely throughout their home ranges to maintain exclusive breeding rights to females. Females without kittens also move extensively within their ranges (Maehr 1997). Panthers are capable of moving large distances in short periods of time. Nightly panther movements of 12 mi (20 km) are not uncommon (Maehr et al. 1990b).

**Intraspecific interactions** – Adult females and their kittens interact more frequently than any other group of panthers. Interactions between adult male and female panthers last from 1 to 7 days and usually result in pregnancy (Maehr et al. 1991). Aggressive interactions between males often result in serious injury or death. Independent subadult males have been known to associate with each other for several days and these interactions do not appear to be aggressive in nature. Aggression between males is the most common cause of male mortality and an important determinant of male spatial and recruitment patterns based on radio-collared panthers (Maehr et al. 1991; Shindle et al. 2003). In the absence of direct field observations/measurements, Harrison (1992) suggested landscape corridors for wide-ranging predators should be half the width of an average home range size. Following Harrison's (1992) suggestion, corridor widths for Florida

panthers would range from 6.1 to 10.9 mi (9.8 to 17.6 km) depending on whether the target animal was an adult female or a transient male. Beier (1995) suggested that corridor widths for transient male puma in California could be as small as 30 percent of the average home range size of an adult Florida panther; however, topography in California is dramatically different from that in Florida. Without supporting empirical evidence, Noss (1992) suggests regional corridors connecting larger hubs of habitat should be at least 1.0 mi (1.6 km) wide. Beier (1995) makes specific recommendations for very narrow corridor widths based on short corridor lengths in a California setting of wild lands completely surrounded by urban areas; he recommended that corridors with a length less than 0.5 mi (0.8 km) should be more than 328 ft (100 m) wide, and corridors extending 0.6 to 4 mi (1 to 7 km) should be more than 1,312 ft (400 m) wide. The Dispersal Zone encompasses 44 mi<sup>2</sup> (113 km<sup>2</sup>) with a mean width of 3.4 mi (5.4 km). Although it is not adequate to support a single panther, the Dispersal Zone is strategically located and expected to function as an important landscape linkage to south-central Florida (Kautz et al. 2006). Transient male panthers currently use this zone as they disperse northward into south-central Florida.

### ***Population Information and Trends***

#### **Number of Populations:**

6 - 80 (NatureServe, 2015)

#### **Population Size:**

1 - 1000 individuals (NatureServe, 2015)

#### **Population Narrative:**

The 1989 population was estimated at about 30 (census) to 50 (extrapolation) in the wild (Seal et al. 1989). As of 1989, 24 individuals had radio collars and about 10-25 had evaded capture or occupied private lands (Seal et al. 1989). The only confirmed, extant panthers are in the south Florida Big Cypress-Everglades area; probably functions as one population, though will treat occupied areas as separate EORs until evidence for merging. (NatureServe, 2015)

### ***Threats and Stressors***

#### **Stressor:**

#### **Exposure:**

#### **Response:**

#### **Consequence:**

**Narrative:** A major limiting factor in southern Florida appears to be availability of suitable habitat (Belden et al. 1988). Habitat loss and alteration have resulted from logging, drainage, oil field activity, housing development, citrus agriculture, and road construction. Though continuing urbanization and agricultural development of habitat has been regarded as the greatest threat, mercury contamination apparently is another major threat, especially where panthers feed most extensively on raccoons (Jordan 1990, Roelke et al. 1992). Collisions with vehicles (most common around the Fakahatchee Strand) are the greatest known mortality factor, but these are relatively easy to document (Maehr 1992). Threatened also by loss of genetic variability, which evidently is responsible for recently observed congenital heart defects, reproductive abnormalities (high incidence of unilateral cryptorchidism) (Barone et al. 1994), and possible immune deficiencies. Hunting of deer (a primary food resource) may be an additional threat. (NatureServe, 2015)

**Recovery****Recovery Actions:**

- The recovery objectives identified in the final third revision of the Florida Panther Recovery Plan (Service 2008) are to: (1) maintain, restore, and expand the Florida panther population and its habitat in south Florida and, if feasible, expand the known occurrence of Florida panthers north of the Caloosahatchee River to maximize the probability of the long-term persistence of this metapopulation; (2) identify, secure, maintain, and restore habitat in potential reintroduction areas within the panther's historic range, and to establish viable populations of the panther outside south and south-central Florida; and (3) facilitate panther conservation and recovery through public awareness and education. The Service's goal for Florida panther conservation in south Florida is to locate, preserve, and restore lands containing sufficient area and appropriate land cover types to ensure the long-term survival of a population of 80 to 100 individuals (adults and subadults) south of the Caloosahatchee River. The Service proposes to achieve this goal through land management partnerships with private landowners, through coordination with private landowners during review of development proposals, and through land management and acquisition programs with Federal, State, local, private, and Tribal partners. Based on an average density of 31,923 ac per panther as determined by Kautz et al. (2006), the acreages of lands necessary to achieve this goal are 2,553,840 ac for 80 panthers and 3,192,300 ac for 100 panthers.

**Conservation Measures and Best Management Practices:**

- Panthers, because of their wide-ranging movements and extensive spatial requirements, are particularly sensitive to habitat fragmentation (Harris 1984). Habitat fragmentation can result from road construction, urban development, and agricultural land conversions. Habitat protection has been identified as being one of the most important elements to achieving panther recovery. While efforts have been made to secure habitat, continued action is needed to obtain additions to and inholdings for public lands, assure linkages are maintained, restore degraded and fragmented habitat, and obtain the support of private landowners for maintaining property in a manner that is compatible with panther use. Conservation lands used by panthers are held and managed by a variety of entities including the Service, NPS, Seminole Tribe of Florida, Miccosukee Tribe of Indians of Florida, FWC, Florida Department of Environmental Protection (DEP), Florida Division of Forestry (FDOF), Water Management Districts, non-governmental organizations, counties, and private landowners.

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## SPECIES ACCOUNT: *Rangifer tarandus caribou* (Woodland caribou)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered/proposed downlisting of Selkirk Mtn. Pop. and Southern mountain pop. to threatened; 01/14/1983; Pacific Region (R1) (USFWS, 2016)

### **Physical Description**

Caribou are a hoofed mammal. They have large, concave hoofs that spread widely to support the animal in snow and soft tundra. The feet also function as paddles when caribou swim. Caribou are the only member of the deer family (Cervidae) in which both sexes grow antlers. Antlers of adult bulls are large and massive; those of adult cows are much shorter and are usually more slender and irregular. In late fall, caribou are clove-brown with a white neck, rump, and feet and often have a white flank stripe. The hair of newborn calves is generally reddish-brown (USFWS, 2016).

### **Historical Range**

The historical range of woodland caribou (*Rangifer tarandus caribou*) included southeastern Alaska and much of Canada, and extended south into northern United States from Washington to New England. This subspecies is still widespread across much of boreal Canada (COSEWIC 2002) (USFWS, 2016).

### **Current Range**

The current range includes the southern Yukon; southwestern Northwest Territories; northern, west-central and southeastern British Columbia; extreme northeastern Washington; extreme northern Idaho; west-central and northern Alberta; boreal portions of Saskatchewan and Manitoba; and the boreal and arctic portions of Ontario, Quebec, and Newfoundland and Labrador (Cichowski et al. 2004) (USFWS, 2016).

### **Critical Habitat Designated**

Yes; 11/28/2012.

### **Legal Description**

On November 28, 2012, the U.S. Fish and Wildlife Service designated critical habitat for the southern Selkirk Mountains population of woodland caribou (*Rangifer tarandus caribou*) under the Endangered Species Act. In total, approximately 30,010 acres (12,145 hectares) was designated as critical habitat. The critical habitat is located in Boundary County, Idaho, and Pend Oreille County, Washington.

### **Critical Habitat Designation**

The Selkirk Mountains Critical Habitat Unit consists of 30,010 ac (12,145 ha) in Boundary County, Idaho and Pend Oreille County, Washington. Lands within this unit are at 5,000 ft (1,520 m) and higher in elevation. These lands are under Federal ownership, within the Colville and Idaho Panhandle National Forests. The Selkirk Mountains Critical Habitat Unit was occupied at the time of both the emergency listing on January 14, 1983 (48 FR 1722), and the final listing in 1984 (49 FR 7390; February 29, 1984), and is essential to the conservation of the species. This area also contains the PBFs essential to the conservation of the southern Selkirk Mountains population of woodland caribou and which may require special management considerations or protection.

**Primary Constituent Elements/Physical or Biological Features**

A critical habitat unit is designated for Boundary County, Idaho, and Pend Oreille County, Washington. Within this area, the primary constituent elements of the physical and biological features essential to the conservation of the southern Selkirk Mountains population of woodland caribou consist of five components:

- (i) Mature to old-growth western hemlock (*Tsuga heterophylla*)/western red cedar (*Thuja plicata*) climax forest, and subalpine fir (*Abies lasiocarpa*)/ Engelmann spruce (*Picea engelmanni*) climax forest at least 5,000 ft (1,520 m) in elevation; these habitats typically have 26–50 percent or greater canopy closure.
- (ii) Ridge tops and high elevation basins that are generally 6,000 ft (1,830 m) in elevation or higher, associated with mature to old stands of subalpine fir (*Abies lasiocarpa*)/Engelmann spruce (*Picea engelmanni*) climax forest, with relatively open canopy.
- (iii) Presence of arboreal hair lichens.
- (iv) High-elevation benches and shallow slopes, secondary stream bottoms, riparian areas, and seeps, and subalpine meadows with succulent forbs and grasses, flowering plants, horsetails, willow, huckleberry, dwarf birch, sedges and lichens. The southern Selkirk Mountains population of woodland caribou, including pregnant females, uses these areas for feeding during the spring and summer seasons.
- (v) Corridors/Transition zones that connect the habitats described above. If human activities occur, they are such that they do not impair the ability of caribou to use these areas.

The PBFs for the southern Selkirk Mountains population of woodland caribou are, therefore, the arrangement of the above habitat types and their components and transition zones on the landscape in a manner that supports seasonal movement, feeding, breeding, and sheltering needs. Each of the seasonal use areas creates space on the landscape that allows caribou to spread out and avoid predators. These areas also have little or no disturbance from forest practices, roads, or recreational activities.

**Special Management Considerations or Protections**

Critical habitat does not include manmade structures (such as buildings, roads, and other paved areas) and the land on which they are located existing within the legal boundaries on December 28, 2012.

The primary land uses are forest management activities and recreational activities, which occur throughout the year. Recreational activities include, but are not limited to, snowmobiling, offhighway vehicle (OHV) use, backcountry skiing, and hunting. Special management considerations or protection needed within the unit are required to address habitat fragmentation of contiguous old growth forests due to forest practices and activities, wildfire, and disturbances such as roads and recreation.

Special management considerations or protection are required within critical habitat areas to address these threats. Management activities that could ameliorate these threats include, but

are not limited to, conservation measures and actions to minimize the effects of forest management practices on the PBFs, actions to minimize the potential for wildfire and the implementation of rapid-response measures, as appropriate, when wildfire occurs, road and recreational area closures as appropriate to avoid or minimize the potential for disturbance-related impacts, and reducing opportunities for predator-caribou interactions.

The United States-Canada border in the Selkirk Mountains is remote, rugged, and permeable to the southern Selkirk Mountains population of woodland caribou. Illegal border-related activities and resultant law enforcement response (such as increased human presence, and vehicles including trucks, motorcycles, and all-terrain-vehicles), has the potential to cause adverse effects in these remote areas. While current levels of law enforcement activity do not pose a threat, a substantial increase in activity levels could be of concern. We note that some level of law enforcement activity can be beneficial, as it decreases illegal traffic. Significant increases in illegal cross-border activities in the designated critical habitat areas could pose a threat to the southern Selkirk Mountains population of woodland caribou, and therefore, to a degree, border security actions provide a beneficial decrease in cross-border violations and their impacts. There are no known plans to construct security fences in the designated critical habitat. We do not anticipate impermeable fencing being built in areas with rugged terrain. Technological solutions and other tactics for Homeland Security purposes would be more likely to be applied in these areas.

### ***Life History***

#### **Feeding Narrative**

Adult: In many areas, relies heavily on lichens. In the southern Selkirk Mountains of British Columbia, caribou diet shifted from primarily vascular taxa during snow-free months to an arboreal lichen-conifer diet during late winter; in winters with slower snow accumulation, caribou foraged extensively on myrtle boxwood (*Pachistima*) and other vascular plants (Rominger and Oldemeyer 1990). Late fall and early winter diet consists of low evergreen shrubs, mushrooms, grasses, and sedges (Matthews and Moseley 1990) (USFWS, 2016).

#### **Reproduction Narrative**

Adult: Mating occurs primarily in last half of October. Male defends group of several females. Calving occurs in late May or early June, though the timing of the rut and calving varies somewhat among different herds. Females begin breeding at 2-3 years. Most adult females bear a calf each year. End of metabolic weaning occurs at about 6 weeks, though nursing may continue for at least 160 days (Can. J. Zool. 70:1753).; Tends to congregate in family groups of 3-10; adult males solitary except in breeding season (Matthews and Moseley 1990). Annual calf mortality, due to predation, severe weather, or malnutrition, 40-70% (Matthews and Moseley 1990) (NatureServe, 2015). Newborn calves weigh an average of 13 pounds (6 kg) and grow very quickly. They may double their weight in 10-15 days. Weights of adult bulls average 350-400 pounds (159-182 kg). However, weights of 700 pounds (318 kg) have been recorded. Mature females average 175-225 pounds (80-120 kg). Caribou in northern and southwestern Alaska are generally smaller than caribou in the Interior and in southern parts of the state (USFWS, 2016).

#### **Habitat Narrative**

Adult: In most areas, woodland caribou favor large tracts of mature to old forests and forested peatlands containing lichens, which provide the primary winter food source (Alberta Sustainable Resource Development and Alberta Conservation Association 2010). Forest-tundra migratory

ecotypes of subspecies caribou exist in some areas, such as the George and Leaf River area in Quebec-Labrador and on the Hudson Plains in Manitoba and Ontario (COSEWIC 2002) (USFWS, 2016).

### ***Dispersal/Migration***

#### **Motility/Mobility**

Adult: High (USFWS, 2016)

#### **Migratory vs Non-migratory vs Seasonal Movements**

Adult: Migratory (NatureServe, 2015)

#### **Dispersal/Migration Narrative**

Adult: Makes seasonal elevational migrations; early winter habitat lower in elevation than late winter habitat (Rominger and Oldemeyer 1989).; Nonmigrant: N; Local migrant: Y; Distant migrant: N; (NatureServe, 2015). They are local migrants, not long-distance migrants, making season elevational migrations. Preferred early winter habitat is lower in elevation than late winter habitat (Rominger and Oldemeyer 1989) (USFWS, 2016).

#### **Additional Life History Information**

Adult: Makes seasonal elevational migrations; early winter habitat lower in elevation than late winter habitat (Rominger and Oldemeyer 1989).; Nonmigrant: N; Local migrant: Y; Distant migrant: N; (NatureServe, 2015)

### ***Population Information and Trends***

#### **Population Narrative:**

As of late 1980s, southern Selkirk population was relatively stable at 25-30 individuals (Rominger and Oldemeyer 1989); population in 1990 was 60-70 due to an infusion of 60 caribou translocated from larger herds in British Columbia; both native and transplanted populations were reproducing (USFWS 1990). A 1993 winter census yielded a total of 51 caribou, including 7 calves, in the Selkirk ecosystem (1993 End. Sp. Tech. Bull. 18(3):21). In 1997, the population was up to 59; in addition, 13 were translocated from western Canada to the eastern Washington part of the Selkirk Mountain Recovery Area in 1997 (End. Sp. Bull. 22(3):20). Larger populations occur elsewhere. (NatureServe, 2015)

### ***Threats and Stressors***

#### **Stressor:**

#### **Exposure:**

#### **Response:**

#### **Consequence:**

**Narrative:** Decline has been attributed to habitat alteration caused by logging, mining, road construction, severe winter weather, and fire, which reduced availability of lichens (primary winter food), and/or to wolf predation and overharvesting by hunters. Low reproductive potential, traditional habits, gregariousness, and curiosity make woodland caribou vulnerable to hunting and to changes or destruction of habitat (Kelsall, 1984 COSEWIC report). It has been suggested that colonization of range by moose resulted in increased wolf numbers and increased

predation on caribou (see Seip 1992, Matthews and Moseley 1990). Predation by an expanding coyote population threatened a remnant caribou herd in southeastern Quebec (Crete and Desrosiers 1995). Black bear predation may be a significant cause of poor calf survival in Saskatchewan (Rettie and Messier 1998). Potential threats include poaching, collisions with motor vehicles, and genetic problems from inbreeding (Rominger 1990). (NatureServe, 2015)

## **Recovery**

### **Recovery Actions:**

- Recovery Component for Populations and Habitat: • Promote the recovery of the population and areas disturbed and that are located within the priority areas for the pronghorn conservation, with emphasis in the Natural Areas and areas of historical distribution.
- Activities: • Identification of “critical” (critical) zones within the current pronghorn distribution areas that are key for the continuation of the genetic flow of this species and to promote the fixing or removal of the fences built to contain the livestock. • Determine the possibility and mechanisms necessary for the recovery of the populations and the identification of critical areas. • Coordinate, across and within institutions, actions to implement the recovery of populations and improvement of disturbed areas identified as “critical”. • Implement actions for the restoration of critical areas identified as distribution areas for the pronghorn along with the ANP.
- Impact Mitigation and Prevention Components: • Reduce the impact generated by the property fencing, changes in the use of the land, and other factors, in the pronghorn populations and their habitats.
- Activities: • Establish preventive and corrective actions, in coordination with the local authorities and property owners, to prevent the fencing needed for the cattle, thus allowing the free flow of pronghorn between different areas. • Monitor the effect that the main risk factors identified may have in the pronghorn populations. • Periodically evaluate the impact of the main risk factors in the pronghorn populations. • Establish mechanisms, within the institutions, that will guarantee the prevention of impacts in the pronghorn population and its habitat.
- Management Strategies: XVII. 1 Habitat Management Component • Develop and implement actions and activities that will guarantee the existence of enough habitats to be able to maintain viable pronghorn populations in the areas of distribution of this specie.
- Activities: • Promote and manage payment programs for environmental services with the Comisión Nacional Forestal (CONAFOR) for the areas with pristine habitat for the pronghorn. • Promote the creation of new federal, state, or governmental natural areas, the certification of the properties for the conservation and establishment of the Unidades de Manejo para la Conservación de la Vida Silvestre, in distribution areas for the pronghorn, as a tool for the conservation and restoration of the habitat for the species. • Accomplish the implementation of the properties where conservation efforts for the pronghorn and its habitat are taking place, with the benefits from the Pago por Servicios Ambientales (PSA), Programas de Empleo Temporal (PET), and Programas de Conservación para el Desarrollo Rural Sustentable (PROCOCODES), in priority areas that may be under any protection status or that may have been identified as important for this species, as well as, limit and/or regulate the productive activities and the infrastructure that can threaten such areas. • Promote the review and follow up of the management programs of the ANP and

- UMA located in the distribution areas with the objective of proposing adaptations and improvements, in an agreeable way with the property and landowners in these areas. • Promote and follow up the Programas de Ordenamiento Territorial in the elected municipality and communities settled in regions with conservation priority for the pronghorn, with the objective of promoting the continuity of the habitat that will allow the genetic flow of the species. • Promote the productive diversification in areas located within the pronghorn distribution areas, with low impact activities that will benefit the conservation of the wildlife and their habitats. • Establish, organize and coordinate agricultural and livestock activities in, or around, the important habitat for the pronghorn. • Promote the recovery of the habitat throughout the implementation of sustainable tourism programs that will increase the interest of the pronghorn in the society (showing live individuals, guided tours, camping, nature tourism through the protected areas, etc).
- Species Management Component: • Determine and standardize the procedures for the management of individuals and populations.
  - Activities: • Elaborate a standard manual of procedures for the management of individuals, focused on reproduction, and of populations, focused on recovery and sustainable benefit. • Continue with the reproduction, breeding, and translocation for the creation of new populations. • Develop regional diagnosis with the objective of promoting intersectional meetings according to the priority to be addressed. • Coordinate the Programa de Fomento Ganadero (PROGAN) de la SAGARPA, mainly in the natural areas located in the distribution areas for the pronghorn, with the objective of organizing the livestock activity. • Subscribe the production organizations to the Sistema- Producto Ganadería Diversificada SAGARPA, with the objective of financing the recovery, repopulation, and reproduction projects for the pronghorn. • Promote an agreement between SEMARNAT and SAGARPA, for the implementation of an improvement program for cattle management in the distribution areas for the pronghorn. • Promulgate a directory of specialists and working groups that will conduct studies or actions for the management, recovery, conservation, and protection of the pronghorn at the regional, national, and international level. 249 XVII. Strategies to Deve

***Conservation Measures and Best Management Practices:***

- Reinforce, coordinate, and implement the mechanisms to protect the distribution areas for the pronghorn.
- Promote the conservation and protection of the priority areas for the conservation of the pronghorn and its habitat. Such areas include Áreas Naturales Protegidas (Protected Natural Areas), Predios Certificados para la Conservación (Certified Properties for the Conservation), Reservas comunales y/o privadas (Common and/or private reserves), as well as, Unidades de Manejo para la Conservación de la Vida Silvestre (UMA).
- Achieve the incorporation of properties where there currently exist activities related to the conservation of the pronghorn and its habitat, for the benefit of the Pago por Servicios Ambientales (PSA - Captura de carbono, Hidrológicos y para Conservación de Biodiversidad), Programas de Conservación para el Desarrollo Rural Sustentable (PROCOCODES) and all of those who help with the productive activities
- Promote the productive activities within the zones classified as priority.
- Promote the steps that will help reduce the risks and threats for the pronghorn populations, such as, exclusion of free-range cattle that may compete for the same habitat as the pronghorn's, stimulation of the habitat, and management or even control the predators in those areas.

- Promote through an institutional coordination, the Ordenamientos Territoriales Municipales (Municipal territory laws) in the areas with conservation priorities for the pronghorn.
- Consolidate, along with the authorities, the outline for the participation of different sectors to avoid the destruction of the pronghorn habitat, due to changes in the use of the land.
- Activities:
  - Create an efficient system for the uptake and processing of complaints to the pertinent authorities that will require an immediate set of actions with the objective of stopping and discouraging any illegal attempts that may be taking place in those areas designated for pronghorn.
  - Promote social participation strategies for the environmental surveillance, with different approaches that will target several sectors, for the conservation of areas designated for pronghorn.
  - Promote, closely with the Procuraduría Federal de Protección al Ambiente (PROFEPA), the timely processing of any complaints that are related with affecting, directly or indirectly, the pronghorn and its habitat.
  - Recognize and involve the legal hunting departments, to request their assistance in spreading the regulations and conservation efforts for the species, with the objective of reducing any pronghorn hunting by designing actions for each kind of identified hunting.
  - Promote inspection and surveillance rounds in the areas where pronghorn are distributed, during the seasons when hunting is allowed for other species that share the habitat with the pronghorn.
  - Collaborate with the Procuraduría Federal de Protección al Ambiente (PROFEPA), in training federal inspectors and the community surveillance group, whose main objective is to help prevent and detect pronghorn illegal hunting and any activities related to the destruction and fragmentation of its habitat.
  - Reinforce inspection and surveillance activities with state and municipal governments.

## References

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## SPECIES ACCOUNT: *Reithrodontomys raviventris* (Salt marsh harvest mouse)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered (USFWS 1970)

### **Physical Description**

The salt marsh harvest mouse is a small, rodent in the family Muridae. It is a buff to brown, colored mouse with a body 2.75–3 inch (7-7.6 centimeters) long. From nose to tip of tail the mouse can be up to 6.3 inches (16 centimeters) long. It has dark ears and a long, indistinctly bicolored tail; it may have a dark stripe down its back. It is one of two subspecies harvest mouse. Both subspecies of salt marsh harvest mouse have grooved upper front teeth; the scientific name *Reithrodontomys raviventris* means “grooved-toothed mouse with a red belly.” They are primarily, but may be active during the day. (EPA 2010, USFWS 2013, NatureServe 2014)

### **Historical Range**

Historically, the salt marsh harvest mouse probably occurred in salt marshes along the central coast of California, with population concentrations in the salt marshes of the San Francisco Bay area. (EPA 2010, USFWS 2010, USFWS 2013, NatureServe 2014).

### **Current Range**

: Currently, the salt marsh harvest mouse’s populations inhabit less than 40-100 square miles (100-250 square kilometers). The range encompasses salt marshes of the San Francisco Bay system (San Francisco, San Pablo, and Suisun bays), in central California. This includes lands in the San Pablo Bay National Wildlife Refuge (NWR), in the San Francisco Bay NWR, and lands protected and managed under the Suisun Marsh Preservation Agreement. (EPA 2010, USFWS 2010, USFWS 2013, NatureServe 2014)

### **Distinct Population Segments Defined**

No

### **Critical Habitat Designated**

No;

### ***Life History***

#### **Feeding Narrative**

Adult: The salt marsh harvest mouse uses the pickleweed marsh for year-round cover from predators, an escape from, and as a food source. They eat green vegetation including salt grass and pickleweed, as well as some seeds. The salt marsh harvest mouse can tolerate high salinity in both food (grasses, forbs, seeds, and insects) and water

#### **Reproduction Narrative**

Adult: The salt marsh harvest mouse typically lives about 8 to 12 months. The mouse is sexually active from March to November and females often bear one to three litters of four offspring. Nests may only be a loose ball of grasses on the ground.

**Spatial Arrangements of the Population**

Adult: clumped according to suitable resources

**Environmental Specificity**

Adult: generalist with special habitat requirements

**Tolerance Ranges/Thresholds**

Adult: tolerate salt

**Site Fidelity**

Adult: high

**Dependency on Other Individuals or Species for Habitat**

Adult: pickleweed (*Sarcocornia pacifica*)

**Habitat Narrative**

Adult: Salt marsh harvest mice are usually found in tall, dense, continuous stands of pickleweed (*Sarcocornia pacifica*). They can also be found in mixed stands of pickleweed that include alkali heath (*Frankenia salina*), spearscale (*Atriplex prostrata*), and possibly saltgrass (*Distichlis spicata*). The tall pickleweed stands or mixed pickleweed stands remain mostly above water during periods of flooding. A pickleweed canopy height of approximately 6 inches (15 centimeters) appears to be the lowest commonly used by salt marsh harvest mice. These stands are the optimal habitat for this mouse because they are adjacent to upland, salt-tolerant vegetation, for escape during high tides. The salt marsh harvest mouse uses the pickleweed marsh for year-round cover from predators, an escape from, and as a food source. They eat green vegetation including salt grass and pickleweed, as well as some seeds. The salt marsh harvest mouse can tolerate high salinity in both food (grasses, forbs, seeds, and insects) and water; its ability to swim and climb enables this mouse to take advantage of its unique habitat. The once extensive salt marsh habitat is now extremely fragmented. Only about 30,100 acres (12,180 hectares) remain of the estimated 193,800 acres (78,500 hectares) of tidal marsh existing in the 1850s. Some of the remaining marshes have no high-ground area where the salt marsh harvest mouse can escape flooding. Sea level rise may contribute to the loss of more salt marsh habitat. (EPA 2010, USFWS 2013, NatureServe 2014)

***Dispersal/Migration*****Motility/Mobility**

Adult: mobile

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: not migratory

**Dispersal**

Adult: yes

**Immigration/Emigration**

Adult: unlikely because of fragmented habitat

**Dependency on Other Individuals or Species for Dispersal**

Adult: pickleweed (*Sarcocornia pacifica*)

**Dispersal/Migration Narrative**

Adult: Salt marsh harvest mice float and swim well, but they do not swim as well as other small tidal marsh mammals and they do not dive. Salt marsh harvest mice remain in their home ranges during high tide and swim or cling to taller vegetation or floating debris. (EPA 2010, USFWS 2013, NatureServe 2014) Young mice disperse a considerable distance through the habitat, but not outside their habitat or across bare areas. The home range averages 0.52 acre (0.21 hectare) in size. (EPA 2010, USFWS 2013, NatureServe 2014)

***Population Information and Trends*****Population Trends:**

Declining

**Species Trends:**

Declining

**Resiliency:**

low

**Representation:**

low

**Redundancy:**

moderate

**Population Growth Rate:**

unknown

**Number of Populations:**

21 to 80

**Population Size:**

estimated between 1000 and 10000

**Minimum Viable Population Size:**

unknown

**Resistance to Disease:**

unknown

**Adaptability:**

low

**Population Narrative:**

The current number of individuals is unknown. Populations are limited to remaining salt marsh habitat in San Francisco, San Pablo and Suisun Bays. Populations adjacent to developed areas may be subject to more threats, such as pollution, predation, and human disturbance.

Significant habitat disturbance and degradation continues in some portions of the range due to routine human activities, including maintenance of dikes, levees, flood control, vegetation control, recreational uses, human and domestic and feral animal incursion from adjoining developments. Without habitat protection, restoration and management, the existing populations cannot adequately absorb losses from environmental events and human disturbance. Extirpated populations may fail to re-establish despite regeneration of suitable habitat conditions because of lack of dispersal opportunities from source populations. The loss of 92% of historic suitable habitat is a significant barrier to dispersal and re-colonization.

**Threats and Stressors**

**Stressor:** Destruction of habitat

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The greatest threat to the salt marsh harvest mouse is the destruction and alteration of habitat. Habitat loss is due to filling and diking from urban, agricultural, and industrial development, subsidence, changes in water salinity, nonnative species invasions, sea level rise, and pollution. The quality of remaining habitat is further limited by small size, lack of connectivity, and lack of other vital features such as tall emergent vegetation. Habitat fragmentation reduces salt marsh harvest mouse populations to many isolated, tiny populations on habitat fragments of varying size, shape, and condition. Marsh fragments may lack the full range of habitat features needed; for example, they may contain feeding and nesting habitat, but not refuge from flooding. Fragmented marsh areas are more susceptible to edge effects from outside influences such as domestic cat predation, human disturbance, or pollution. (EPA 2010, USFWS 2010, USFWS 2013, NatureServe 2014)

**Stressor:** Flooding

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Flooding that submerges vegetation may occur from very high tides near the summer and winter solstices, storm surges, and extreme river outflows into the estuary. Prolonged flooding exposes salt marsh harvest mice to predators, and increases the risk of mortality due to exposure or drowning. Extreme high tides can endanger whole populations of salt marsh harvest mice. Little is known about predation impacts to the salt marsh harvest mouse; however, predation during flooding may be an important factor. During high winter tides it is common to see birds such as shorebirds and raptors taking small mammals from the upper edges and flooded areas of marshes. Protection from predators depends on the dense vegetation cover; mice that leave this cover are exposed to predation. Potential terrestrial predators include red fox (*Vulpes vulpes*), gray fox (*Urocyon cinereoargenteus*), feral or otherwise free-roaming cats (*Felis domestica*), skunks (*Mephitis mephitis*), and raccoons (*Procyon lotor*). Predation is likely

greatest in habitats with incomplete or sparse or patchy vegetation cover. These habitats are usually closer to urban edges where terrestrial predators, such as cats, occur.

### ***Recovery***

#### **Reclassification Criteria:**

The protection, management and restoration of suitable tidal marsh habitat in each marsh complex is sufficient to support multiple viable habitat areas occupied by salt marsh harvest mice, that are distributed among recovery units as specified below in criteria A/1 through A/5.

Sub-criterion A: Protection of Documented Occurrences Habitat supporting all documented salt marsh harvest mouse occurrences must be protected via habitat management.

Sub- criterion B: VHA Characteristics Each marsh complex must support VHAs, as described above, and these areas shall be connected by suitable habitat corridors with sufficiently deep (from shore to bay) pickleweed plains and/or sufficiently deep high marsh zones (and preferably both). This will allow movement of salt marsh harvest mice through these areas to occur unobstructed.

Sub- criterion C: Marsh Connectivity: Wherever possible, the marsh complexes themselves must be connected to one another by marsh or restored tidal marsh of sufficient depth and complexity to allow for dispersal and recolonization.

Sub- criterion D: Marsh Complex Minimum Acreage: Marsh complexes must be 1,000 acres or more in size, except in Corte Madera marsh where, due to constraints on restorable habitat, the marsh complex must be 400 acres or more in size. All VHAs within each marsh complex must be 150 acres or more in size.

#### **Recovery Actions:**

- The most important data/research need at present is to fill in gaps in understanding of the current distribution, density, and demographics of the salt marsh harvest mouse. Most records are greater than ten years old and no systematic surveys have been carried out in key areas. Expectations of salt marsh harvest mouse population expansion into restored marshes are dependent on the presence of extant populations adjacent to restoration areas that can serve as source populations of the mouse. Resources for salt marsh harvest mouse surveys should be shifted from site-specific presence/absence surveys, to systematic regional surveys with replicated sampling over time. Surveys should give special emphasis to building upon information gained after the 2005 floods by tracking salt marsh harvest mouse (and other small salt marsh mammal) populations before and several years after major flood events, comparing population regeneration and extinction probabilities for a range of habitat types, sizes, and landscape positions (location along sloughs or bays, distances from nearest known populations or habitats). Regional survey programs for both subspecies should be established and funded for a minimum of 10 years or one flood/drought cycle.
- High ground adjoining or near marshes should be acquired and protected. Existing steep-sided outboard dikes that back most of the marshes of the bay should be redesigned such that when they need to be replaced or heightened, in response to flooding threats from sea level rise, they have much more gradual slopes on their bay sides (i.e., slopes of 10 to 1 or

more rather than the 1 to 1 to 2 to 1 slopes that presently exist). High ground should be connected to marshes wherever possible. The hills to the west and northwest of Tolay Creek in the San Pablo Bay, for example, should be connected to the flood plain of that creek through acquisitions and easements such that there will be room for future high marsh growth in the future as the rising sea level swamps the creek. The same is true for the Potrero Hills in the Suisun Marsh area. More acquisitions and easements should be made around them so that the marshes surrounding them can migrate landward as sea levels rise. Such planning and acquisitions will help protect future marshes from losing their high marsh zones altogether.

- Further fill of low-lying wetlands, salt ponds or other presently nontidal areas adjacent to tidal salt marshes or narrow fringing marshes should be either prohibited or severely discouraged. Building on such areas will reduce the areas into which marshes can expand as sea level rises. Commercial or residential areas immediately adjacent to marshes, especially narrow fringing marshes, will take priority for protection to prevent the further fragmentation of already fragmented marshes of the bay.
- The relationship and potential use or avoidance of perennial *Lepidium latifolium* (pepperweed) by salt marsh harvest mice should be investigated. *Lepidium latifolium* is increasing in almost all of the more brackish parts of the San Francisco Bay and while we know that the mice will use mixtures of *Lepidium latifolium* and bulrush, we do not have information on how large monocultures of this species will effect salt marsh harvest mice. In addition, the use of bulrush and other brackish species in the South San Francisco Bay should be investigated. The H.T. Harvey 2006 study for the City of San Jose showed that the mice do use it but, "we do not know how the distribution, densities or the persistence of salt marsh harvest mice may change as the ratio of alkali bulrush to perennial peppergrass changes both seasonally and over longer periods of time. Neither do we know the size of a mouse's home range within stands of alkali bulrush, how far they move within it, or whether they live in it for prolonged periods of time."
- Although the salt marsh harvest mouse is relatively well-known in the bay area, public understanding of its ecological needs should be improved. Age appropriate educational materials should be prepared collaboratively by species experts and public educators, and distributed to public schools, university programs and environmental journalists. Public outreach materials should focus on the principal threats to the species (with emphasis on local conservation issues), recovery strategies and actions, and the results or progress of local recovery actions.

## References

FEMA species account

Nature serve

five year review

## SPECIES ACCOUNT: *Sorex ornatus relictus* (Buena Vista Lake ornate Shrew)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; March 6, 2002 (67 FR 10101).

### **Physical Description**

The Buena Vista Lake shrew (*Sorex ornatus relictus*) has a long snout; tiny bead-like eyes; ears that are concealed, or nearly concealed, by soft fur; and five toes on each foot (65 FR 35033). The upper surface of the Buena Vista Lake shrew is blackish-brown, with a pepper-and-salt pattern of buffy brown and black, the black predominating. The sides are more buffy brown than the upper surface. The lower surface is smoke gray. The tail is not noticeably bicolored and darkens toward the end, both above and below. Ranges of external measurements from the type specimen and two additional specimens are: total length, 98 to 105 millimeters (3.86 to 4.13 inches [in.]); tail length, 35 to 39 mm (1.38 to 1.54 in.); hind foot length, 11.5 to 13 mm (0.45 to 0.51 in.); and ear length from the notch 6.5 to 8.5 mm (0.26 to 0.33 in.). Weights ranged from 4.1 to 7.6 grams (0.14 to 0.27 ounce) (USFWS 1998).

### **Taxonomy**

The Buena Vista Lake shrew is one of nine subspecies of the ornate shrew (*Sorex ornatus*). Seven of the subspecies occur only in California; one occurs in California and Baja California; and one subspecies only occurs in Baja California (USFWS 2011). The Buena Vista Lake shrew differs morphologically from ornate shrew (*S. ornatus ornatus*), whose range surrounds that of Buena Vista Lake shrew. The coloration of the Buena Vista Lake shrew is distinctly darker, grayish-black rather than brown. The body size is slightly larger, but the tail is shorter. The teeth are essentially the same, but the third and fifth unicuspid (teeth behind the incisors that have a single main cusp) are even smaller relative to the other teeth (USFWS 1998).

### **Historical Range**

The Buena Vista Lake shrew formerly occurred in wetlands around Buena Vista Lake, and presumably throughout the Tulare Basin. The Tulare Basin, essentially occupying the southern half to the San Joaquin Valley, had no regular outlet to the ocean and contained Buena Vista, Kern, and Tulare lakes. These lakes were fed by the Kern, Kaweah, Tule and Kings rivers and their tributaries, and were interconnected by hundreds of square miles of tule marshes and other permanent and seasonal lakes, wetlands, and sloughs (USFWS 2011). As early as 1933, the distribution of this species began to be much more restricted due to the disappearance of lakes and sloughs. Buena Vista Lake and the surrounding lakes and valley freshwater marshes have been drained and cultivated. Furthermore, canals in the area are steep-sided and kept free of vegetation (USFWS 1998).

### **Current Range**

For more than 50 years, the shrew was known only from the type locality at Buena Vista Lake, where it was presumed to be extinct because its wetland habitat had been replaced by residential and agricultural lands. The shrew was rediscovered at Kern Lake Preserve (Kern Preserve) in 1986, on private property, and at Kern National Wildlife Refuge (Kern NWR) in 1992. Other remnant patches of wetland and riparian communities within the Tulare Basin have

not been surveyed and may support the Buena Vista Lake shrew. These include Jerry Slough, overflow channels of the Kern River owned and managed by the Semitropic Water District as a groundwater recharge basin 10 miles (mi.) south of Kern NWR; and the privately owned Creighton Ranch, near the eastern shore of historical Tulare Lake in Tulare County (USFWS 2011). The Buena Vista Lake shrew is now known from four isolated locations along an approximately 113-kilometer (70-mi.) stretch on the western side of the Tulare Basin. The four locations are the former Kern Preserve on the old Kern Lake bed, the Kern Fan recharge area, Cole Levee Ecological Preserve (Cole Levee), and the Kern NWR (67 FR 10101).

**Distinct Population Segments Defined**

No

**Critical Habitat Designated**

Yes; 1/24/2005.

**Legal Description**

On July 2, 2013, the Service designated approximately 2,485 acres (ac) (1,006 hectares (ha)), in six units in Kings and Kern Counties, California, as critical habitat for the subspecies.

**Critical Habitat Designation**

The six units designated are: (1) Kern National Wildlife Refuge Unit, (2) Goose Lake Unit, (4) Coles Levee Unit, (5) Kern Lake Unit, (6) Semitropic Ecological Reserve Unit, and (7) Lemoore Wetland Reserve Unit. Note that proposed Unit 3 (the Kern Fan Water Recharge Unit) has been excluded from final designation due to the existing habitat conservation plan.

Unit 1: Kern National Wildlife Refuge Unit Unit 1 consists of a total of approximately 387 ac (157 ha). The Kern NWR Unit is completely comprised of Federal lands, and is located within the Kern NWR in northwestern Kern County. The Kern NWR Critical Habitat Unit consists of three subunits: Subunit 1A is approximately 274 ac (111 ha); subunit 1B is 66 ac (27 ha); and subunit 1C is 47 ac (19 ha). The unit was occupied at the time of listing, is currently occupied, and contains the physical and biological features that are essential to the conservation of the shrew. Shrew habitat in Unit 1 receives water from the California Aqueduct. One of the areas where Buena Vista Lake shrews are present has standing water from September 1 through approximately April 15. After that time, the trees in the area may receive irrigation water so the area may possibly remain damp through May, but the area is dry for approximately 3 months during the summer. Another area of known Buena Vista Lake shrew occurrences has standing water from the second week of August through the winter and into early July, and is only dry for a short time during the summer. Buena Vista Lake shrew have been captured in remnant riparian and slough habitat at the Refuge (Service 2005, pp. 48, 49). Like all the critical habitat units we are designating here (see Criteria Used to Designate Critical Habitat, above), this unit is essential to the conservation of the shrew because it is occupied, and because the subunits include riparian habitat that contain the appropriate physical or biological features and primary constituent elements for the shrew. *Populus fremontii* trees (Fremont cottonwood) and *Salix* spp. (willow) are the dominant woody plants in riparian areas. Additional plants include bulrushes, cattails, *Juncus* spp. (rushes), *Heleocharis palustris* (spike rush), and *Sagittaria longiloba* (arrowhead). Other plant communities on the refuge that support shrews are valley iodine bush scrub, dominated by iodine bush, seepweed, *Frankenia salina* (alkali heath), and salt-cedar scrub, which is dominated by *Tamarix* spp. (salt cedar). Both of these communities

occupy sites with moist, alkaline soils. The Kern NWR completed a Comprehensive Conservation Plan (CCP) for the Kern and Pixley NWRs in February 2005 (Service 2005, pp. 1-103). The CCP provides objectives for maintenance and restoration of Buena Vista Lake shrew habitat on the Kern NWR. Objectives listed in the CCP include: completing baseline censuses and monitoring for the shrew; enhancement and maintenance of the 215-ac (87-ha) riparian habitat through regular watering to provide habitat for riparian species including the shrew; and additional restoration of 15 ac (6 ha) of riparian habitat along canals in a portion of the Refuge to benefit the shrew and riparian bird species (Service 2005, pp. 84, 85). The physical and biological features essential to the conservation of the species in this unit may require special management considerations or protection to address threats from nonnative species such as salt cedar, and from changes in hydrology due to offsite water management.

**Unit 2: Goose Lake Unit** The Goose Lake Unit consists of a total of approximately 1,274 ac (515 ha) of private land, and is located about 10 mi (16 km) south of Kern NWR in northwestern Kern County, in the historical lake bed of Goose Lake. The Goose Lake Unit consists of two subunits: Subunit 2A contains 159 ac (64 ha), and Subunit 2B contains 1,115 ac (451 ha). We consider that the unit was occupied at the time of listing and assume that it was not identified as occupied at that time because it had not yet been surveyed for small mammals. In January 2003, when the area was first surveyed for small mammals, approximately 6.5 ac (2.6 ha) of potential shrew habitat located along the Goose Lake sloughs were surveyed (ESRP 2004, p. 8), resulting in the capture of five Buena Vista Lake shrews. The maximum distance between two shrew captures was 1.6 mi (2.6 km), suggesting that Buena Vista Lake shrews are widely distributed on the site. The unit has been determined to have the necessary physical or biological features present and therefore meets the definition of critical habitat under section 3(5)(A)(i) of the Act. The unit was included in the 2004 proposed critical habitat designation. Although we continue to presume that the unit meets the definition of critical habitat under section 3(5)(A)(i) of the Act (prong 1), we are also designating the unit under section 3(5)(A)(ii) of the Act (prong 2). As discussed above under Criteria Used To Identify Critical Habitat, even if subsequent evidence were to indicate that the unit was not occupied at the time of listing, it would remain critical habitat under the second prong of the Act's definition. The unit is essential for the conservation of the shrew because it is among the very few remaining areas that support both an extant shrew population and the physical and biological features necessary to conserve that population. In the past, Buena Vista Lake shrew habitat in this unit experienced widespread losses due to the diversion of water for agricultural purposes. However, small, degraded examples of freshwater marsh and riparian communities still exist in the area of Goose Lake and Jerry Slough (a portion of historical Goose Slough, an overflow channel of the Kern River), allowing shrews to persist in the area. Dominant vegetation along the slough channels includes frankenia, iodine bush, and seepweed. The northern portion of the unit consists of scattered mature iodine bush shrubs in an area that has relatively moist soils. The southern portion of the unit is characterized by a dense mat of saltgrass and clumps of iodine bush and seepweed. A portion of the unit currently exhibits inundation and saturation during the winter months. Dominant vegetation in these areas has included cattails, bulrushes, and saltgrass. The area consisting of the former bed of Goose Lake is managed by the Semitropic Water Storage District (WSD) as a ground-water recharge basin. Water from the California Aqueduct is transferred to the Goose Lake area in years of abundant water, where it is allowed to recharge the aquifer that is used for irrigated agriculture. At the time that the unit was originally proposed, the landowners, in cooperation with Ducks Unlimited, Inc. and Semitropic WSD, proposed to create and restore habitat for waterfowl in the unit area; wetland restoration that we expected to substantially increase the

quantity and quality of Buena Vista Lake shrew habitat on the site. Restoration activities were completed in the last 6 years. The physical and biological features essential to the conservation of the species in this unit may require special management considerations or protection to address threats from nonnative species such as salt cedar, from recreational use, and from changes in hydrology due to water management and maintenance of water conveyance facilities. No conservation agreements currently cover this land.

**Unit 4: Coles Levee Unit** The Coles Levee Unit is approximately 270 ac (109 ha) in Kern County, of which 217 ac (88 ha) is owned by Aera Energy. An additional 46 ac (19 ha) are State lands within the Tule Elk Reserve, and 6 ac (2 ha) are part of a Kern County park. The unit is located northeast of Tupman Road near the town of Tupman, is directly northeast of the California Aqueduct, and is largely within the Coles Levee Ecosystem Preserve, which was established as a mitigation bank in 1992, in an agreement between Atlantic Richfield Company (ARCO) and CDFW. The preserve serves as a mitigation bank to compensate for the loss of habitat for listed upland species; the Buena Vista Lake shrew is not a covered species. ARCO had been issued an incidental take permit under section 10(a)(1)(B) of the Act for the Coles Levee Ecological Preserve Area (Service 2001, p. 1). However, the take authorization provided by the permit lapsed when ARCO sold the property to the current owner and the permit was not transferred. Habitat on the preserve consists mostly of highly degraded upland saltbush and mesquite scrub, and is interlaced with slough channels for the historical Kern River fan where the river entered Buena Vista Lake from the northeast. Most slough channels are dry except in times of heavy flooding. This site runs parallel to the Kern River bed and contains approximately 2 mi (3.2 km) of much-degraded riparian vegetation along the Kern River. A manmade pond, which was constructed in the late 1990s or early 2000s, is located within the unit. Water from the adjacent oil fields is constantly pumped into the basin. Vegetation includes bulrushes, *Urtica dioica* (stinging nettle), *Baccharis salicifolia* (mulefat), salt grass, *Atriplex lentiformis* (quailbush), and *Conium maculatum* (poison hemlock). A few willows and Fremont cottonwoods are scattered throughout the area. In the 2009 proposed rule (74 FR 53999, October 21, 2009), we repropose 214 ac (87 ha) of critical habitat as the Coles Levee Unit. In this unit, Buena Vista Lake shrews were originally captured along a nature trail that was adjacent to a slough, and were close to the water's edge where there was abundant ground cover but little or no canopy cover. The unit is delineated in a general southeast to northwest direction, along both sides of the Kern River Flood Channel and Outlet Canal, which runs through the Preserve. During a construction project in the summer of 2011, two Buena Vista Lake shrews were found just north of the previous northerly boundary of the unit. We have therefore extended the unit boundary along both sides of the canal to encompass the contiguous riparian habitat to the point where water is no longer retained and riparian vegetation essentially stops, thereby including riparian habitat along the Outlet Canal within the Tule Elk Reserve. This unit is essential to the conservation of the species because it was occupied at the time of listing (67 FR 10102), is considered currently occupied, and includes willow-cottonwood riparian habitat that contains the PCEs. The physical and biological features essential to the conservation of the species in this unit may require special management considerations or protection to address threats from construction activities associated with projects to tie-in water conveyance facilities to the California Aqueduct and oil and gas-related activities, including pipeline projects. The area adjacent to Coles Levee is a site of active gas and oil production, and the Coles Levee Unit is within an area that was recently proposed for additional oil and gas exploration.

**Unit 5: Kern Lake Unit** The Kern Lake Unit is approximately 85 ac (35 ha) in size, and is located at the edge of the historical Kern Lake, approximately 16 miles south of Bakersfield in southwestern Kern County. This unit lies between Hwy 99 and Interstate 5, south of Herring Road near the New Rim Ditch. The Kern Lake Unit consists of two subunits: Subunit 5A contains 34 ac (14 ha), and Subunit 5B contains 51 ac (21 ha). The unit was occupied at the time of listing, is considered currently occupied, and contains the physical and biological features that are essential to the conservation of the Buena Vista Lake shrew. Since the advent of reclamation and development, the surrounding lands have seen intensive cattle and sheep ranching and, more recently, cotton and alfalfa farming. Currently, Kern Lake itself is generally a dry lake bed; however, the unit contains wet alkali meadows and a spring-fed pond known as "Gator Pond," which is located near the shoreline of the lake bed. A portion of the runoff from the surrounding hills travels through underground aquifers, surfacing as artesian springs at the pond. The heavy clay soils support a distinctive assemblage of native species, providing an island of native vegetation situated among agricultural lands. The unit contains three ecologically significant natural communities: freshwater marsh, alkali meadow, and iodine bush scrub. This unit is essential to the conservation of the species because it is currently occupied and includes habitat that contains the PCEs identified for the shrew. The Kern Lake area was formerly managed by the Nature Conservancy for the J.G. Boswell Company, and was once thought to contain the last remaining population of the Buena Vista Lake shrew. The physical and biological features essential to the conservation of the species in this unit may require special management considerations or protection to address threats from reductions in water delivery, from effects of surrounding agricultural use, and from industrial and commercial development. This area does not have a conservation easement and is managed by the landowners. We are unaware of any plans to develop this site; however, it is within a matrix of lands managed for agricultural production.

**Unit 6: Semitropic Ecological Reserve Unit** The Semitropic Ecological Reserve Unit is approximately 372 ac (151 ha) in size and is located about 7 mi (11 km) south of Kern NWR and 7 mi (11 km) north of the Goose Lake Unit along the Main Drain Canal in Kern County. It is bordered on the south by State Route 46, approximately 2 mi (3 km) east of the intersection with Interstate 5. The CDFW holds 345 ac (140 ha) under fee title, and manages the area as part of the Semitropic Ecological Reserve. An additional 27 ac (11 ha) of the unit are private land. We consider that the unit was occupied at the time of listing and assume that it was not identified as occupied at that time because it had not yet been surveyed for small mammals (see Criteria Used To Identify Critical Habitat). Buena Vista Lake shrews were identified in the unit on April 27, 2005, when it was first surveyed for small mammals (ESRP 2005, pp. 10-13). At that time, Buena Vista Lake shrews were found in the southwestern portion of the unit, next to the Main Drain Canal. The unit has been determined to have the necessary PCEs present and therefore meets the definition of critical habitat under section 3(5)(A)(i) of the Act. Although we presume that the unit meets the definition of critical habitat under section 3(5)(A)(i) of the Act, we are also designating the unit under section 3(5)(A)(ii) of the Act. Even if the unit was not occupied at the time of listing, it is essential for the conservation of the Buena Vista Lake shrew due to its location approximately midway between Units 1 and 2, and location near the southern edge of remnant natural wetland and riparian habitat. The unit is also essential for the conservation of the shrew because it is considered to be currently occupied, and contains a matrix of riparian and wetland habitat, including riparian habitat both along the canal and within and adjacent to oxbow and slough features. The major vegetative associations at the site are valley saltbush scrub and valley sink scrub. Valley saltbush scrub is found within the

relatively well-drained soils at slightly higher elevations, and the valley sink scrub is found in the heavier clay soils. Dominant vegetation at the site includes *Bromus diandrus* (ripgut brome), *Bromus madritensis* ssp. *rubens* (red brome), *Carex* spp. (sedges), *Juncus* spp. (rushes), *Polygonum* spp. (knotweed), *Polypogon monspeliensis* (rabbitfoot grass), *Rumex crispus* (curly dock), and *Vulpia myuros* (foxtail fescue). There is a light overstory of cottonwoods at the trapping location where the most Buena Vista Lake shrews have been observed. The physical and biological features essential to the conservation of the species in this unit may require special management considerations or protection to address threats from ongoing oil and gas exploration and development, ongoing conversion of natural lands for agricultural development, changes in water management, weed control activities including use of herbicides, and the occurrence of range trespass in an open range area. Semitropic reserve lands are not fenced and are subject to occasional range trespass by sheep and cattle (CDFW 2012). State lands in the unit were acquired under the provisions of the Metro Bakersfield Habitat Conservation Plan (HCP), and are managed for listed upland species. Location of the Main Drain Canal in the unit, and the presence of wetland features are expected to benefit the shrew, although the shrew is not a covered species under the HCP. The State does not yet have a management plan for the Semitropic Ecological Reserve.

**Unit 7: Lemoore Wetland Reserve Unit** The Lemoore Wetland Reserve Unit, 97 ac (39 ha) in size, is located east of the Lemoore Naval Air Station and is 4 mi (6 km) west of the City of Lemoore in Kings County. The unit is bounded along the southern border by State Route 198, and on the north and west sides by a bare water-conveyance canal. The unit is managed by the Natural Resources Conservation Service for waterfowl enhancement. We consider that the unit was occupied at the time of listing and that it was not identified as occupied at that time because it had not yet been surveyed for small mammals (see Criteria Used To Identify Critical Habitat). Buena Vista Lake shrews were identified in the unit in April 2005, when it was first surveyed for small mammals (ESRP 2005, pp. 10-13). The unit has been determined to have the necessary PCEs present and, therefore, meets the definition of critical habitat under section 3(5)(A)(i) of the Act. Although we presume that the unit meets the definition of critical habitat under section 3(5)(A)(i) of the Act, we are also designating the unit under section 3(5)(A)(ii) of the Act. The unit is essential for the conservation of the shrew due to its location at the northernmost extent of the subspecies' range and its geographic isolation from other units, due to occupancy, and due to remnant natural wetland and riparian habitat that contains the PCEs. The site is part of an area that was created to provide a place for city storm water to percolate and drop potential contaminants to shield the Kings River during years of flood runoff. Portions of the area are flooded periodically, forming fragmented wetland communities throughout the area. The plant communities of the Lemoore Wetland Reserve Unit include a mixture of vegetation communities: nonnative grassland, vernal marsh, and elements of valley sink scrub. Commonly occurring plants include *Brassica nigra* (black mustard), red brome, *B. hordeaceus* (soft chess), saltgrass, alkali heath, rushes, *Lactuca serriola* (prickly lettuce), rabbitfoot grass, cottonwood, *Rumex crispus* (curly dock), *Salix* spp. (willow), *Scirpus* spp. (bulrush), *Sonchus oleraceus* (common sowthistle), cattails, foxtail fescue and *Xanthium strumarium* (cocklebur). This unit is essential to the conservation of the species because it is currently occupied and contains the PCEs identified for the shrew.

#### **Primary Constituent Elements/Physical or Biological Features**

Critical habitat units are designated for Kings and Kern Counties, California. Within these areas, the primary constituent elements of the physical or biological features essential to the

conservation of the Buena Vista Lake shrew consist of permanent and intermittent riparian or wetland communities that contain:

- (i) A complex vegetative structure with a thick cover of leaf litter or dense mats of low-lying vegetation. Associated plant species can include, but are not limited to, Fremont cottonwoods, willows, glasswort, wild-rye grass, and rush grass. Although moist soil in areas with an overstory of willows or cottonwoods appears to be favored, such overstory may not be essential.
- (ii) Suitable moisture supplied by a shallow water table, irrigation, or proximity to permanent or semipermanent water.
- (iii) A consistent and diverse supply of prey. Although the specific prey species used by the Buena Vista Lake shrew have not been identified, ornate shrews are known to eat a variety of terrestrial and aquatic invertebrates, including amphipods, slugs, and insects.

### **Special Management Considerations or Protections**

Critical habitat does not include manmade structures (such as buildings, aqueducts, runways, roads, and other paved areas) and the land on which they are located existing within the legal boundaries on the effective date of this rule.

All designated critical habitat units will require some level of management to address the current and future threats to the physical and biological features essential to the conservation of the Buena Vista Lake shrew. Special management considerations or protection may be required to minimize habitat destruction, degradation, or fragmentation associated with such threats as the following: Changes in the water supply allocations, water diversions, flooding, oil and gas extraction, nonnative vegetation, and agriculture. For example, the Coles Levee area is within the boundaries of a proposed oil and gas exploration proposal. Agricultural pressures to convert land to agriculture remain in the southern San Joaquin Valley, with agricultural conversion to orchards noted to have occurred recently in the general area.

The Buena Vista Lake shrew also faces high risks from random catastrophic events (such as floods or drought) (Service 1998, p. 163). The low numbers of Buena Vista Lake shrews located in small isolated areas increases the risk of a random catastrophic event eliminating entire populations or severely diminishing Buena Vista Lake shrew numbers to the point that recovery is precluded. These threats and others mentioned above could render the habitat less suitable for the Buena Vista Lake shrew by washing away leaf litter and complex vegetation structure (floods) or drying wetland habitat so that vegetative and prey communities die (drought), and special management may be needed to address these threats.

### ***Life History***

#### **Feeding Narrative**

Adult: The specific feeding and foraging habits of the Buena Vista Lake shrew are unknown (USFWS 1998). The Buena Vista Lake shrew is an invertivore that feeds indiscriminately on the available larvae and on adults of several species of aquatic and terrestrial insects, some of which are detrimental to agricultural crops. They are also known to consume spiders, centipedes, slugs, snails, and earthworms on a seasonally available basis. Shrews have a high rate of metabolism because of their small size, forcing them to be constantly searching for food to

maintain their body temperatures, especially in cold conditions. Food probably is not cached and stored, so the shrew must forage periodically day and night to maintain its high metabolic rate. The Buena Vista Lake shrew prefers moist habitat that has a diversity of terrestrial and aquatic insect prey. Imported water to the Tulare Basin has resulted in an upward movement of selenium, which has become concentrated in the shrew's invertebrate prey; the potential dietary selenium levels are within the range that is known to be toxic to small mammals (NatureServe 2015; 67 FR 10101; USFWS 1998; USFWS 2011).

**Reproduction Narrative**

Adult: Nothing is known specifically about the reproduction and mating system of the Buena Vista Lake shrew. Most ornate shrews have a gestation period of 21 days. They can have one to two litters per year, with four to six young per litter (Bolster 1998). The life expectancy of most ornate shrews is 12 to 16 months (Bolster 1998). The breeding season begins in February or March and ends with the onset of the dry season in May or June; or may extend later in the year, based on habitat quality and availability of water. The Buena Vista Lake shrew prefers moist habitat that has a diversity of terrestrial and aquatic insect prey (67 FR 10101).

**Geographic or Habitat Restraints or Barriers**

Adult: Habitat fragmentation from the impoundment and diversion of streams, draining of marshes and lakes, and widespread land-leveling (USFWS 2011).

**Spatial Arrangements of the Population**

Adult: Clumped

**Environmental Specificity**

Adult: Narrow/specialist.

**Tolerance Ranges/Thresholds**

Adult: Moderate

**Site Fidelity**

Adult: High

**Dependency on Other Individuals or Species for Habitat**

Adult: No

**Habitat Narrative**

Adult: Ornate shrews in general tend to be associated with the structure of vegetation rather than with species composition of the community. Historically, Buena Vista Lake shrews occupied valley freshwater marshes on the perimeter of Buena Vista Lake and probably occurred throughout the Tulare Basin (USFWS 1998). Currently, the Buena Vista Lake shrew occupies a very small, reduced range in the southern San Joaquin Valley, where there are only a few extant occurrences known. The species may have lost more than 95 percent of historical habitat, because most of its former wetland habitat has been drained, converted to agriculture, or dried up because of water diversion (NatureServe 2015). The Buena Vista Lake shrew requires a complex vegetative structure with a thick cover of leaf litter or dense mats of low-lying vegetation (78 FR 39835). Associated plant species can include, but are not limited to sedges (*Carex* sp.), foxtail barley (*Hordeum murinum*), spikerushes (*Eleocharis* sp.), black mustard

(*Brassica nigra*), rushes (*Juncus* spp.), bromes (*Bromus* sp.), stinging nettle (*Urtica dioica*), mulefat (*Baccharis salicifolia*), bush lupine (*Lupinus albifrons*), and wild rose (*Rosa californica*), along with cattails (*Typha* sp.), tules (*Schoenoplectus acutus*), and other aquatic plants. Areas with an overstory of willows (*Salix* sp.) or cottonwoods (*Populus* sp.) appear to be favored, but may not be an essential habitat feature (USFWS 2011). Habitat fragmentation from the impoundment and diversion of streams, draining of marshes and lakes, and widespread land-leveling serves as a geographic/habitat barrier or restraint (USFWS 2011).

***Dispersal/Migration*****Motility/Mobility**

Adult: Low

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Low

**Immigration/Emigration**

Adult: Unlikely

**Dependency on Other Individuals or Species for Dispersal**

Adult: No

**Dispersal/Migration Narrative**

Adult: Little is known about home range, or territoriality of the Buena Vista Lake shrew or ornate shrews in general. Buena Vista Lake shrews have low motility and are nonmigratory. The limited available habitat of the Buena Vista Lake shrew is fragmented, and therefore limits dispersal. Due to lack of study, information about the home range size of the shrew is lacking. In other species of ornate shrews, juveniles establish their home range—a small area in which they nest, forage, and explore—and remain in this area for most of their life. The distribution and size of a shrew's territory varies, and is primarily influenced by the availability of food (USFWS 2011).

**Additional Life History Information**

Adult: Due to lack of study, information about the home range size of the shrew is lacking. In other species of ornate shrews, juveniles establish their home range—a small area in which they nest, forage, and explore—and remain in this area for most of their life. The distribution and size of a shrew's territory varies, and is primarily influenced by the availability of food (USFWS 2011).

***Population Information and Trends*****Population Trends:**

Unknown (NatureServe 2015)

**Species Trends:**

Unknown (NatureServe 2015)

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Number of Populations:**

Unknown; to date, surveys for the shrew have been conducted at 21 sites, and shrews were found to be present in eight of them (USFWS 2011).

**Population Size:**

Unknown (USFWS 2011)

**Resistance to Disease:**

Unknown (USFWS 2011)

**Adaptability:**

Low

**Additional Population-level Information:**

At the time of the proposed listing for the Buena Vista Lake shrew, there was one known extant population on the Kern Preserve, a private property totaling about 34 hectares (ha) (83 acres). Since that time, two Buena Vista Lake shrew were trapped on the southern side of the Kern NWR in 1998. In 1999, the California State University Stanislaus Foundation's Endangered Species Recovery Program found nine more shrews along the banks of an artificial pond adjacent to the nature center at the Cole Levee, and five more at the Kern County's water recharge area along the Kern Fan. Over the last 20 years, a number of surveys have taken place, all of which were unsuccessful in capturing any Buena Vista Lake shrews. Other remnant patches of wetland and riparian communities in the Tulare Basin have not been surveyed, and may support the Buena Vista Lake shrew (67 FR 10101).

**Population Narrative:**

When the species was listed in 2002, the shrew was only known to occur in four locations along an approximately 70-mi. stretch on the western side of the Tulare Basin. The four locations were the former Kern Preserve in the old Kern Lake bed, the Kern Fan recharge area, Cole Levee, and the Kern NWR. To date, surveys for the shrew have been conducted at 21 sites, and shrews were found to be present in eight of them (USFWS 2011). These eight sites are Goose Lake, Atwell Island, Main Drain Canal/Chicca & Sons Twin Farms South Field Ranch, Lemoore Wetlands preserve, Cole Levee, Kern fan water recharge area, the Kern NWR, and the Kern Preserve (USFWS 2011). Other remnant patches of wetland and riparian communities in the Tulare Basin have not been surveyed, and may support the Buena Vista Lake shrew (67 FR 10101). The abundance of the shrew is unknown due to the lack of regular surveys in areas of past occurrences and in areas possessing suitable habitat. Surveys were only conducted in places containing very high-quality shrew habitat, and in places where access was allowed by the land owner. Based on these surveys, the shrew has been documented as far south as the

Kern Preserve and as far north as Atwell Island. Population size and health cannot be estimated with the available data, but based on the scarcity of suitable habitat in the San Joaquin Valley and the low number of specimens collected in areas with high-quality habitat, the species is expected to be extremely rare. The small sample sizes obtained from each locality is a reflection of the rarity and difficulty of capturing shrews in these areas (USFWS 2011).

### ***Threats and Stressors***

**Stressor:** Habitat loss

**Exposure:** Agricultural and urban development.

**Response:** Habitat degradation.

**Consequence:** Population decline.

**Narrative:** All of the natural plant communities in the Tulare Basin have been affected by the alteration of the area for urban and agricultural development. As more canals are built, and more water is diverted for agricultural irrigation of the historic floodplains of the major rivers of the southern San Joaquin Valley, less water is available to sustain the riparian and wetland areas on which the shrew relies for all aspects of its life. This will continue to cause the already very rare subspecies to decline (USFWS 2011).

**Stressor:** Predation

**Exposure:** Avian predators.

**Response:**

**Consequence:** Unknown

**Narrative:** There are several avian predators, such as barn owls (*Tyto alba*), short-eared owls (*Asio flammeus*), long-eared owls (*Asio otus*), and great horned owls (*Bubo virginianus*), that are known to prey on shrews (USFWS 2011). The overall impact that predation may have on the number of individuals and densities of the shrew remains unknown (USFWS 2011).

**Stressor:** Hybridization

**Exposure:** Overlapping shrew population ranges.

**Response:** Hybridization

**Consequence:** Genetic alteration.

**Narrative:** If shrew population ranges overlap or come in contact through expansion, hybridization may occur in closely related species and certain subspecies. Over time, a population of a subspecies could become genetically indistinguishable from a larger population of an intruding subspecies, so that the true genotypes of the invaded subspecies no longer exist (USFWS 2011).

**Stressor:** Selenium toxicity

**Exposure:** Irrigation

**Response:** Accumulation in plants and animals.

**Consequence:** Adverse effects to growth, reproduction, and survival.

**Narrative:** Selenium toxicity of soils in the Tulare Basin has resulted from the concentration of already naturally elevated levels of selenium on the western side of the San Joaquin Valley. Due to extensive agricultural irrigation, selenium has been leached from the soils and concentrated in the shallow groundwater. In areas where this groundwater reaches the surface or subsurface, selenium can accumulate in both plants and animals. Elevated concentrations of selenium may cause adverse effects to growth, reproduction, and survival of the shrew (USFWS 2011).

**Stressor:** Exposure to pesticides

**Exposure:** Spraying crops, canals, and ditch banks.

**Response:** Lethal and sub-lethal pesticide concentration.

**Consequence:** Possible reduced reproduction, and death by starvation.

**Narrative:** The Buena Vista Lake shrew is exposed to the wide-scale use of pesticides throughout its range and may be directly exposed to lethal and sub-lethal concentrations of pesticides from drift or direct spraying of crops, canals, and ditch banks. Reduced reproduction in shrews could be directly caused by ingested pesticides. Additionally, shrews could die from starvation by the loss of their prey base (USFWS 2011).

**Stressor:** Small population size

**Exposure:** Extreme weather, introduction of nonnative species, pollution, and development.

**Response:** Limited gene flow, genetic variation, and ability to adapt to drastic environmental events.

**Consequence:** Lower fitness and survivability.

**Narrative:** Due to low population numbers and a high degree of habitat fragmentation, the Buena Vista Lake shrew is particularly vulnerable to sudden changes in its environment, be they natural events such as extreme weather or epidemic diseases, or anthropogenic changes such as the introduction of a nonnative species, chemical runoff or spill, or human development of an ecologically important area. Limited gene flow and genetic variation in a population has the capacity to limit the species' ability to adapt to drastic environmental events, and can result in lower breeding success or inbreeding; this can result in decreased fitness and survivability of the shrew (USFWS 2011).

**Stressor:** Climate change

**Exposure:** Predictions indicate warmer temperatures, more intense precipitation events, and increased summer continental drying.

**Response:** Alteration of available habitat.

**Consequence:** Immense stress on the species.

**Narrative:** Current climate change predictions for terrestrial areas indicate warmer air temperatures, more intense precipitation events, and increased summer continental drying. Due to the shrew's reliance on dense riparian vegetation and to the continuing diversion of water from wetland areas for agricultural use, and because the shrew's decline was greatly attributed to the loss of wetland habitat required for its survival, it can be assumed that increased drying could place immense stress on the species (USFWS 2011).

## ***Recovery***

### **Reclassification Criteria:**

Reclassification criteria have not yet been developed for the Buena Vista Lake shrew.

### **Delisting Criteria:**

The species was listed as a species of concern at the time the recovery plan (USFWS 1998) was written and published. Although recovery criteria were not detailed in the plan, long-term conservation-recovery criteria were provided (USFWS 2011).

Secure and protect recovery areas totaling at least 2,000 ha (4,940 ac.) of occupied habitat in three or more disjunct sites (USFWS 1998).

Approve management plans for those sites that feature survival of the species as an objective, and implement those plans (USFWS 1998).

Implement a periodic monitoring plan that demonstrates continuing presence of Buena Vista Lake shrews at occupied sites (USFWS 1998).

**Recovery Actions:**

- Develop and implement a regional cooperative program and participation plan (USFWS 1998).
- Protect and secure core habitat areas (USFWS 1998).
- Determine distributions and population statuses of featured species (USFWS 1998).
- Conduct important research and population monitoring (USFWS 1998).
- Maintain and establish linkages in existing natural lands, and between islands of habitat on the valley floor and natural lands around the fringe of the valley (USFWS 1998).
- Apply adaptive management to protected areas (USFWS 1998).
- If necessary, reintroduce Buena Vista Lake shrew to appropriate habitat within their historic range (USFWS 1998).
- Periodically review the status to determine whether listing as endangered or threatened is necessary (USFWS 1998).
- Conserve habitat and restore riparian and wetland vegetation communities (USFWS 2011).
- Estimate population sizes at existing and potentially inhabited sites (USFWS 2011).
- Establish habitat connectivity between populations (USFWS 2011).
- Develop agreements with private entities to assess and protect areas with potential habitat (USFWS 2011).
- Develop management plans and agreements with land owners and managers (USFWS 2011).

***Conservation Measures and Best Management Practices:***

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***Additional Threshold Information:***

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## SPECIES ACCOUNT: *Sylvilagus bachmani riparius* (Riparian brush rabbit)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; February 23, 2000 (65 FR 8881).

### **Physical Description**

Brush rabbits (*Sylvilagus bachmani*) are small, brownish rabbits that can be distinguished from their relative, the desert cottontail (*S. audubonii*), by a smaller, inconspicuous tail and uniformly colored ears with no black tip. The adult riparian brush rabbit is about 30 to 37.5 centimeters (10.58 to 13.23 inches) long (USFWS 1998).

### **Taxonomy**

There are 13 subspecies of brush rabbit that are recognized; the riparian brush rabbit (*Sylvilagus bachmani riparius*), is one of eight subspecies found in California. The riparian brush rabbit can be distinguished from other subspecies by its relatively pale color, gray sides, darker back, and, when viewed from above, by its cheeks—which protrude outward rather than being straight or concave (USFWS 1998).

### **Historical Range**

The riparian brush rabbit is believed, based on the presence of suitable habitat, to have been historically found in riparian forests along portions of the Stanislaus River and its tributaries on the Valley floor, from at least Stanislaus County to the delta (USFWS 1998).

### **Current Range**

By the mid-1980s, the riparian forest within the former range of the riparian brush rabbit had been reduced to a few small and widely scattered fragments, totaling about 2,100 hectares (5,189 acres). Caswell Memorial State Park, on the Stanislaus River in southern San Joaquin County, is the largest remaining fragment of suitable riparian forest and home to the only extant population of riparian brush rabbit (USFWS 1998).

### **Distinct Population Segments Defined**

No

### **Critical Habitat Designated**

No;

### ***Life History***

### **Feeding Narrative**

Adult: The riparian brush rabbit is a herbivore, feeding on grasses, sedges, clover, forbs, shoots, and leaves in small clearings adjacent to their riparian habitat. Grasses and other herbs are the most important food for brush rabbits, but shrubs such as California wild rose (*Rosa californica*), marsh baccharis (*Baccharis douglasii*), and California blackberry (*Rubus ursinus*) also are eaten. When available, green clover (*Trifolium wormskioldii*) is preferred over all other foods. Food resource distribution is limited due to the need for brush rabbits to remain within 1 meter (m) (3

feet [ft.]) of their riparian habitat to escape to the cover of a dense understory. Competition exists from the more fecund and mobile desert cottontail (*Sylvilagus audubonii*). Riparian brush rabbits are crepuscular (most active during the twilight hours around dawn and dusk). Depending on season, the main activity periods last 2 to 4 hours. The least activity is from about 10:30 a.m. to 4:00 p.m. Growth rates are fast; young rabbits reach adult size in 4 to 5 months (USFWS 1998; 65 FR 8881).

**Reproduction Narrative**

Adult: Riparian brush rabbits reach sexual maturity the winter following their birth. The species requires riparian forests with a dense understory shrub layer for breeding. Brush rabbits live in tunnels that run through the vines and shrubs of California wild rose (*Rosa californica*) and Pacific blackberry (*Rubus vitifolius*), and require areas of higher ground that are not flooded regularly or heavily (65 FR 8881). The percentage of females active during the breeding season is unknown, but in one study, 9 of 25 female adults examined showed no signs of reproductive activity (65 FR 8881). Breeding of riparian brush rabbits is restricted to approximately January to May, putting this species at a competitive disadvantage to the desert cottontails outside the park, which breed all year. The period of gestation is about 26 to 30 days (average 27 days), the usual litter size is three or four. Females typically produce three to four (up to five) litters during the season and give birth to between two and six young per litter. On average, a female may produce 9 to 16 young each year. Following birth, the young rabbits remain in the nest about 2 weeks before venturing out, and the female will continue to suckle her young 2 to 3 weeks after their birth (65 FR 8881). Although this is a relatively high reproductive rate, it is lower than many other cottontail species, and five out of six rabbits do not survive to the next breeding season (USFWS 1998).

**Geographic or Habitat Restraints or Barriers**

Adult: Remaining riparian habitat for the riparian brush rabbit is confined between the Stanislaus River and a levee (NatureServe 2015).

**Spatial Arrangements of the Population**

Adult: Clumped

**Environmental Specificity**

Adult: Narrow/specialist

**Tolerance Ranges/Thresholds**

Adult: Low

**Site Fidelity**

Adult: High

**Dependency on Other Individuals or Species for Habitat**

Adult: No

**Habitat Narrative**

Adult: Riparian brush rabbits occupy riparian forest with a dense shrub layer and dense thickets—including wild rose (*Rosa* sp.), willows (*Salix* sp.), and blackberries (*Rubus* sp.)—close to the Stanislaus River. Where mats of low-growing wild roses, wild grape (*Vitis californica*), and

blackberries are found in savanna-like settings, brush rabbits live in tunnels through the vines and shrubs. The presence of more surface litter and lack of willows in the understory signifies areas of higher ground that are not flooded regularly or heavily (USFWS 1998). Brush rabbits frequent small clearings, where they bask in the sun and feed on a variety of herbaceous vegetation (65 FR 8881). They will not cross large, open areas, and therefore are unable to disperse beyond the dense brush of the riparian forest at Caswell Memorial State Park. Individuals are intolerant of each other when they come too close, but there is no well-defined territoriality. Young are more tolerant of approach by another rabbit than are adults (USFWS 1998). Much of the remaining riparian habitat within the range of the riparian brush rabbit is confined between the Stanislaus River and a levee (NatureServe 2015). The riparian brush rabbit can climb into bushes and trees, though its climbing is awkward and its abilities limited. This trait probably has significant survival value, given that the riparian forests that are its preferred habitat are subject to inundation by periodic flooding (USFWS 1998).

***Dispersal/Migration*****Motility/Mobility**

Adult: Low

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Low

**Immigration/Emigration**

Adult: No

**Dependency on Other Individuals or Species for Dispersal**

Adult: No

**Dispersal/Migration Narrative**

Adult: Riparian brush rabbits require nearly continuous shrub cover and seldom move more than 1 m (3 ft.) from cover. They will not cross large, open areas, and therefore are unable to disperse beyond the dense brush of the riparian forest at Caswell Memorial State Park. Due to these circumstances, natural dispersal is not possible (USFWS 1998).

**Additional Life History Information**

Adult: Given the requirement of nearly continuous shrub cover and the smallness and highly fragmented distribution of the remnant of their habitat, natural dispersal cannot be expected (USFWS 1998).

***Population Information and Trends*****Population Trends:**

The short-term population trend is relatively stable (NatureServe 2015).

**Species Trends:**

Unknown; few captures or sightings have occurred since flooding inundated 80 percent of Caswell Memorial State Park in 1997 (NatureServe 2015).

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Number of Populations:**

One

**Population Size:**

Population estimates from 1988 to 1997 have varied from 88 to more than 600 individuals. Flooding in 1997 and 1998 reduced numbers severely. In 1997, no riparian brush rabbits were live-trapped, one was sighted, and pellets from two others were seen; in 1998, one rabbit was live-trapped (65 FR 8881).

**Minimum Viable Population Size:**

Studies suggest that, to ensure the medium- to long-term persistence of birds or mammals, the geometric mean of population size should be about 1,000 for species with normally varying numbers and about 10,000 for species exhibiting a high variability in population size (USFWS 1998).

**Resistance to Disease:**

Low

**Adaptability:**

Low

**Additional Population-level Information:**

Two phenomena jointly have been the primary cause of the decline of the riparian brush rabbit. Both had their origin in the completion, beginning in the 1940s, of large dams for irrigation and flood control on the major rivers of the Central Valley. The first was the destruction and fragmentation of the San Joaquin Valley riparian forest by conversion to various urban and agricultural uses, and its degradation through a variety of other human activities. The second, more specific phenomenon was the conversion of land in the floodplains from shrub- dotted pasture land to vineyards, orchards, and row crops, with attendant land clearing and leveling, and the building and maintenance of levees. The land along rivers no longer exhibits the small patches of shrub- covered upland that once provided rabbits with refuge from flooding and predation (USFWS 1998).

**Population Narrative:**

The short-term population trend is relatively stable. The species level trend is unknown; few captures or sightings have occurred since flooding inundated 80 percent of Caswell Memorial

State Park in 1997 (NatureServe 2015). The population at Caswell Memorial State Park may have reached its lowest numbers after a flood in 1976, when survivors were removed from trees and shrubs and transported in boats by Park personnel. After flooding in 1986, the population was estimated at between 10 and 20 individuals. In 1993, the population was estimated at 213 to 312 individuals, and considered to be at carrying capacity under prevailing environmental conditions. Population estimates from 1988 to 1997 have varied from 88 to more than 600 individuals. Flooding in 1997 and 1998 reduced numbers severely. In 1997, no riparian brush rabbits were live-trapped, one was sighted, and pellets from two others were seen; in 1998, one rabbit was live-trapped (65 FR 8881).

### ***Threats and Stressors***

**Stressor:** Habitat destruction

**Exposure:** Diminished available habitat (USFWS 1998).

**Response:** No ability for natural dispersal (USFWS 1998).

**Consequence:** Little chance for individuals to escape drowning or predation, meet mates, or reproduce (USFWS 1998).

**Narrative:** The destruction and fragmentation of the San Joaquin Valley riparian forest by conversion to various urban and agricultural uses, as well as its degradation through a variety of other human activities, has diminished available habitat to about 5.8 percent of its original extent. Riparian brush rabbits are confined to a narrow habitat range with no ability for natural dispersal. With behavioral restrictions on the species' freedom of movement and extensive habitat fragmentation, there is little chance that those individuals who escape drowning or predation will meet mates or reproduce (USFWS 1998).

**Stressor:** Predation

**Exposure:** Exposure to predators (USFWS 1998).

**Response:** Take refuge on cleared levees (USFWS 1998).

**Consequence:** Population decline and an elevated risk of extinction (USFWS 1998).

**Narrative:** To escape periodic flooding, riparian brush rabbits take refuge on cleared levees. The cleared levees do not provide the same protection as their typical riparian habitat, and they are more exposed to predators. This contributes directly to population decline and an elevated risk of extinction (USFWS 1998).

**Stressor:** Risk of high severity wildfire

**Exposure:** Long-term suppression of fire in Caswell Memorial State Park (USFWS 1998).

**Response:** Buildup of high fuel loads in the dense, brushy habitat to which the rabbits are restricted (USFWS 1998).

**Consequence:** Riparian brush rabbit habitat is highly susceptible to catastrophic wildfire that would cause high mortality and severe destruction of habitat (USFWS 1998).

**Narrative:** Long-term suppression of fire in Caswell Memorial State Park has caused a buildup of high fuel loads in the dense, brushy habitat to which the rabbits are restricted. Riparian brush rabbit habitat is highly susceptible to catastrophic wildfire that would cause high mortality and severe destruction of habitat (USFWS 1998).

**Stressor:** Disease

**Exposure:** Like most rabbits, the riparian brush rabbit is subject to a variety of common diseases (USFWS 1998).

**Response:** Contagious diseases could be easily transmitted from neighboring populations of desert cottontails (USFWS 1998).

**Consequence:** In the small, remnant brush rabbit population, this kind of epidemic could quickly destroy the entire population (USFWS 1998).

**Narrative:** Like most rabbits, the riparian brush rabbit is subject to a variety of common diseases. Contagious diseases could be easily transmitted from neighboring populations of desert cottontails. In the small, remnant brush rabbit population, this kind of epidemic could quickly destroy the entire population (USFWS 1998).

### ***Recovery***

#### **Reclassification Criteria:**

The Recovery Plan for Upland Species of the San Joaquin Valley was published prior to the species listing under the Endangered Species Act. Reclassification criteria have not been identified.

#### **Delisting Criteria:**

The Recovery Plan for Upland Species of the San Joaquin Valley was published prior to the species listing under the Endangered Species Act. Delisting criteria have not been identified.

#### **Recovery Actions:**

- At the time of the publication of the Recovery Plan for Upland Species of the San Joaquin Valley, the riparian brush rabbit was not listed under the Endangered Species Act. The recovery plan considers the riparian brush rabbit a species of concern, and identified a number of generalized criteria for long-term conservation. Range-wide population monitoring should be provided for in all management plans. Specifically, the plan identifies the following actions:
- Secure and protect specified recovery areas from incompatible uses. Three or more sites, each with no fewer than 300 adults during average years (USFWS 1998).
- Management Plan approved and implemented for recovery areas that include survival of the species as an objective for all protected sites (USFWS 1998).
- Population monitoring in specified recovery areas shows populations sizes of 300 or more adults during average years during a precipitation cycle at each of three or more sites (USFWS 1998).

#### ***Conservation Measures and Best Management Practices:***

- Because of the small size of remaining blocks of potential habitat, and the severely limited dispersal capability of the riparian brush rabbit, it is likely to require continuing special protection of its habitat and population. Conservation efforts should focus on the establishment of as many populations in remnant habitat as possible, and sustaining those populations by reintroduction should any one become extinct. In furtherance of these effects, the identified conservation actions are:
- 1. Establish an emergency plan and monitoring system to provide swift action to save individuals and habitat at Caswell Memorial State Park in the event of flooding, wildfire, or a disease epidemic.
- 2. Develop and implement a cooperative riparian brush rabbit conservation program that will include, at a minimum:

- a. Identifying and obtaining biological information needed in management decisions; researching captive breeding methodology using surrogate species; and conducting genetic composition analysis on the riparian brush rabbit population prior to any captive breeding or introduction/reintroduction, and implementing the captive breeding program.
- b. Developing a riparian brush rabbit management plan for Caswell Memorial State Park that will incorporate elements related to predator and pest control; fire lines and access roads; campground, picnic, and recreation areas; brush and fuel control; mosquito abatement; habitat enhancement and expansion of the park.
- c. Establish at least three additional wild populations in the San Joaquin Valley, in restored and expanded suitable habitat within the rabbit's historical range.
- d. Develop a monitoring program of all riparian rabbit populations to assess population trends and status.
- e. Develop a long-term reintroduction preplan for the prompt re-establishment of eliminated populations.
- f. Establish a cooperative program, to take effect once the minimum of four protected populations is established, to place excess young (or other animals as appropriate) from populations at carrying capacity onto private parcels with suitable habitat where owners are willing to enter into a management agreement.

***Additional Threshold Information:***

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## SPECIES ACCOUNT: *Sylvilagus palustris hefneri* (Lower Keys marsh rabbit)

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### *Species Taxonomic and Listing Information*

**Commonly-used Acronym:** LKMR

**Listing Status:** Endangered; Southeast Region (R4) (USFWS, 2015)

### **Physical Description**

LKMR are small rabbits with short, dark brown fur, a gray-white belly, and inconspicuous gray and brown tails. Male and female rabbits are similar in size or color. Adults average 339 millimeters in length and 1,224 g in weight (Forys 1995). The LKMR differs from mainland marsh rabbits (*S. p. palustris*) and Upper Keys marsh rabbits (*S. p. paludicola*) in several cranial characteristics. The LKMR has a shorter molariform tooth row, higher and more convex frontonasal profile, broader cranium, and elongated dentary symphysis (Lazell 1984).

### **Taxonomy**

The LKMR is a subspecies of the marsh rabbit, *S. palustris*. Conspecific marsh rabbits are found throughout southeastern North America. *S. p. palustris* is found from southeastern Virginia south to the Georgia/Florida border. *S. p. paludicola* is found from the Georgia/Florida border south to the Upper Keys. The LKMR is endemic and restricted in range, occurring only in the Lower Keys below the Seven-mile Bridge. It presently inhabits specific areas of salt marsh, transitional buttonwood habitat, freshwater wetlands, and mangrove habitats from Big Pine Key to Boca Chica Key.

### **Historical Range**

See Current

### **Current Range**

The rabbit's original range extended from Big Pine Key to Key West, encompassing a linear distance of about 48 km. Historically, Lower Keys rabbits probably occurred on all of the Lower Keys that supported suitable habitat, but did not occur east of the Seven-mile Bridge where it is replaced by *S. p. paludicola*. Lower Keys rabbits currently occur on only a few of the larger Lower Keys including Boca Chica, Saddlebunch, Sugarloaf, and Big Pine Keys and small islands near these keys (Forys et al. 1996, Faulhaber 2003). Due to urbanization between 1970 and 1996, rabbit habitat has been lost and fragmented. Currently, the habitat consists of a mosaic of small native and disturbed habitat patches. In the 2 years between the 1988-1990 study for the Lower Keys rabbit's listing and the actual listing, four of the 15 original sites used in the listing were destroyed. Approximately 23 percent of the total suitable habitat (both occupied and unoccupied by rabbits) is owned by the military, 38 percent is federally, State, or county owned, and the remaining 39 percent is privately owned. Many of the sites that remain are isolated from each other by urbanized areas, and population interchange seems unlikely. Rangewide surveys of habitat occupancy were initiated in 2001. From 2001 to 2003, Faulhaber (2003) identified, delineated, and mapped suitable (potentially occupied) habitat patches throughout the species' range. He identified 228 suitable habitat patches and all but 8 were surveyed for occupancy. Faulhaber (2003) also tabulated data from historical surveys (1988-1995), and

determined that, of the 228 patches he delineated, 142 had been investigated in the earlier period. Overall, 71 (50 percent) had been occupied during 1988-1995. Faulhaber's (2003) comprehensive inventory indicated that 102 (45 percent) patches were occupied during 2001-2003. Of the 142 patches that were surveyed in both periods, only 62 were still occupied between 2001 and 2003. This represented a 13 percent decrease or an average decline in patch occupancy of about 1.4 percent annually. Two of the patches no longer provided suitable habitat (Faulhaber 2003). By 2005, only 52 of the 71 patches remained occupied (Faulhaber et al. 2007), indicating a 27 percent decrease since the 1988-1995 period. Only a portion of the potential or historical range has been known to be occupied since detailed investigations were initiated in 1988. Of the 228 patches with suitable habitat as of 2001-2003, only 152 are encompassed within the current range (i.e., 152 patches occur on currently occupied keys). Overall, 131 of those 152 patches had been occupied one or more years since 1988. During 1988-1995, 2001-2003, 2004, 2005, 2006, and 2007, approximately 73, 70, 66, 61, 44, and 44 percent of surveyed patches within the current range were occupied (Service 2007a). Again, the results indicate a steep decline in patch occupancy. The greatest decline between any two years occurred between 2005 and 2006. During the 2001-2003 surveys, only six out of the 10 keys occupied by rabbits reported only one or two occupied patches. Evidence of reproduction in the form of juvenile pellets was detected at 43 of the 102 (42.2 percent) occupied patches. Potential habitat was also assessed for rabbit occupancy in various distribution surveys between 1988 and 1995 (Howe 1988, Forsys 1995, Forsys et al. 1996). Of 19 consistently occupied patches observed by Forsys et al. (1996), 4 appeared to be abandoned as of the 2001-2003 survey (Faulhaber 2003). Of the patches observed by Forsys et al. (1996) to be consistently unoccupied, none were subsequently found to be colonized by Faulhaber (2003). During the 2001-2003 survey, Faulhaber (2003) re-surveyed previously surveyed patches and additional patches that he observed to possess potential habitat. Considering only those that were surveyed during 1988-1995 and resurveyed during 2001-2003 (n=134), there was a net decrease in the number of occupied patches (n=9), and further loss of potential habitat patches (n=2). After the 1988-1995 period, observed trends in patch occupancy indicate that patch extinctions accumulated steadily in BCHI. Initially, extinctions appear to have been less frequent in BPK. However, a major decline occurred in BPK after 2005. That decline is temporally correlated with Hurricane Wilma in October 2005. The influence of Wilma may have been mostly due to secondary effects on habitat as opposed to direct mortality. LKMR were frequently observed from roads in at least two areas on Big Pine Key in the winter and spring months of 2005-2006. Yet the survey data indicates the occurrence of a widespread crash in subpopulation persistence. These observations may indicate that LKMR were on the move more than usual, engaged in habitat seeking behavior. Such behavior would likely result in stress to LKMR, and would expose them to increased risks. On BCHI, individual LKMR drowned, but in contrast to BPK the survey data indicate that resilience or resistance was exhibited at the scale of patches. Subpopulations on the southeastern portion of BPK were extirpated earlier, between the 1988-1995 and 2001-2003 survey periods. All six patches documented to contain LKMR in the early period were extinct by 2001-2003. Faulhaber et al. (2007) speculated that this might have resulted from the storm surge associated with Hurricane Georges in 1998. The evidence indicates that impacts from Georges were not widespread in BPK (Service 2007a). However, the characteristics of the storm's surge may have resulted in disproportionate impacts in southeastern Big Pine Key. Ultimately, however, BPK declined more precipitously than the two other metapopulations due to the post-Wilma (2005) losses. Patches throughout the range were inundated by the storm surge associated with Wilma. The available data suggest that LKMR in BPK were impacted more severely by Wilma than were the other metapopulations.

**Distinct Population Segments Defined**

No. (USFWS, 2015)

**Critical Habitat Designated**

Yes;

***Life History*****Feeding Narrative**

Adult: The species feeds on grasses, sedges, succulent plants, and herbaceous shrubs, including at least 19 different plant species representing 14 families (Forys 1995). Important food plants include bushy seaside oxeye, seashore dropseed, Virginia glasswort, cordgrass, beeftree, red mangrove, and white mangrove. Bushy seaside oxeye appears to be the most important food species for the rabbit. Based on their distribution, Lower Keys rabbits appear to need little fresh water to survive. In a study of several mammals from the Lower Florida Keys, the rabbit was found to have one of the highest capacities to concentrate urine. The rabbit may be able to survive solely on dew and brackish water, but probably cannot use seawater.

**Reproduction Narrative**

Adult: The species breeds throughout the year (Holler and Conway 1979). However, higher anestrus (infertile) periods are evident from mid-October through mid-March, although anestrus females are present in every month. During a breeding season, male rabbits become sexually active just prior to females, whose breeding may be induced by male behavior. Both sexes begin to mature at about 9 months of age. During this time, the majority of the males disperse. Sexually maturing females tend to be philopatric to the natal area. Young remain in the nest for about 2 weeks (Forys 1995). Holler and Conway (1979) reported that the subspecies averaged 3.7 litters per year with 1 to 3 young per litter, less than the marsh rabbit (*S. p. paludicola*) in southern Florida (5.7 litters per year with 2 to 4 young per litter). Usually, 75 percent of female marsh rabbits (*S. p. paludicola*) in south Florida are pregnant during the height of the breeding season. The average gestation period for that subspecies in mainland Florida ranges from 30 to 37 days. The species can exhibit total litter resorption that can affect their reproductive output, but it is not yet known if such litter resorption occurs in Lower Keys rabbits. The species is believed to have a maximum longevity of 3 to 4 years in the wild (Whitaker and Hamilton 1998).

**Habitat Narrative**

Adult: In general, marsh rabbits (*S. palustris*) inhabit a variety of wet areas with dense cover (Layne 1974). The Lower Keys rabbit inhabits a narrower range of cover types, occurring primarily in grassy marshes and transition zones. These include freshwater marshes and saltmarsh-buttonwood transition zones (Forys et al. 1996) and historically, coastal beach berm communities. Coastal beach berm, a relatively rare habitat, consists of a vegetated high ridge of storm-deposited sand and shell. Significant tracts of coastal berm habitat in the Lower Keys are found on Big Pine, Newfound Harbor, Lower Sugarloaf, Saddlebunch, Boca Chica, and Geiger Keys. Important plants include grasses and shrubs (shoregrass [*Monanthochloe littoralis*], saltwort [*Batis maritima*], Virginia glasswort [*Salicornia virginica*], marsh fimbry [*Fimbristylis spadiacea*]); succulent herbs (bushy seaside oxeye [*Borrchia frutescens*]); sedges (*Cyperus* spp.); and sparse tree cover (buttonwood [*Conocarpus erectus*], catclaw blackbeard [*Pithecellobium*

guadalupense]). It also inhabits the transitional areas of freshwater wetlands and uplands. These sites are dominated by sawgrass (*Cladium jamaicense*) and cordgrass (*Spartina* spp) (Forys and Humphrey 1999a), and include succulent herbs such as seashore dropseed (*Sporobolus virginicus*). Freshwater marshes are found in depressions in the interior of only a few islands, primarily Sugarloaf, Cudjoe, and Big Pine Keys. During the wet season these areas can accumulate standing water. Rabbits use freshwater marshes extensively on Big Pine Key. Lower Keys rabbits select areas close to large bodies of water, with relatively high amounts of clump grass, ground cover, and bushy seaside oxeye present (Forys 1995). They spend most of their time in the mid-marsh (bushy seaside oxeye) and high-marsh areas (*Spartina* spp. and marsh fimbry), both of which are used for cover and foraging, while most nesting occurs in the high-marsh area (Forys 1995). Lower Keys rabbits occasionally use low shrub marshes and mangrove communities (red mangrove [*Rhizophora mangle*], black mangrove [*Avicennia germinans*], white mangrove [*Laguncularia racemosa*], and buttonwood) for feeding and as a corridor between patches of transitional habitats. Plant species that are most important to the Lower Keys rabbit for cover and nesting include Gulf cordgrass, marsh fimbry, and sawgrass, all of which can form thick cover.

***Dispersal/Migration*****Motility/Mobility**

Adult: High

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory

**Dispersal**

Adult: Dispersing rabbits travel up to 2 km from their nests.

**Dispersal/Migration Narrative**

Adult: Adult rabbits of the same sex tend to have permanent, non-overlapping home ranges (Forys and Humphrey 1996). Those authors found home range area to average 0.32 hectares (ha) and vary widely among individuals. This individual variability may be due to differences in habitat quality, population density, and age or social status. Juvenile rabbits appear to establish a home range near their nest site. Faulhaber (2003) estimated a mean home range size of 1.2 ha for nine rabbits over a minimum of 5 months. Juvenile dispersal occurs at about 9 months of age, and is male biased (Forys 1995). When dispersing from their natal to permanent home ranges, rabbits crossed roads and traveled through a variety of habitats, including mangroves, upland hardwood hammocks, and roadside vegetation. However, they tended to use natural habitats that included dense ground cover (Forys 1995). The distance among habitat patches is important because the ability of rabbits to re-colonize vacant habitat patches depends upon the presence of habitat corridors. Forys (1995) reported that suitable habitat patches occur in a highly fragmented mosaic of native and disturbed habitat, with few contiguous areas of native habitat greater than 5 ha. Dispersing rabbits exhibit high mortality rates, particularly when there is a lack of habitat between populations or when there are roads to cross. Dispersing rabbits travel up to 2 km from their nests. Rabbits are good swimmers and swim as a function of routine movement as well as in dispersal and when pursued. The species appears to be chiefly nocturnal, although they can be active on cloudy days and when they are protected by dense cover.

***Population Information and Trends*****Population Size:**

100 - 300

**Population Narrative:**

Population structure: The Lower Keys rabbit exists as a metapopulation in patches of saltmarsh-buttonwood transition zones, freshwater wetlands and coastal beach berm habitats (Forys 1995, Forys and Humphrey 1996, 1999a, Faulhaber 2003). Rabbits exist in patches, populations and islands that result in various levels of social isolation and limits to interaction through dispersal. The rabbit requires recolonization of vacant habitat patches for survival (Forys et al. 1996). Populations in habitat patches are vulnerable to extinction, but vacant habitat patches have the potential to be recolonized by dispersing rabbits. Problems stemming from random demographic effects on small numbers (e.g., Allee effect) are evident in the populations. Several populations were so small and contained so few individuals of the same sex that extirpation ensued (Forys 1995, Forys and Humphrey 1999b). Over two-thirds of the habitat identified in the Lower Keys is currently occupied by rabbits at a density below carrying capacity (Forys et al. 1996). During rangewide surveys conducted by Faulhaber (2003) in 2001 to 2003, 102 out of 228 potential habitat patches were occupied by rabbits, based on fecal pellets or trapping. Faulhaber (2003) aggregated localized clusters of patches into 56 occupied and 88 potential "local populations." Population estimates range between 100 (in 1993) and 300 (in 1991) (Forys and Humphrey 1999b). Forys et al. (1996) reported an estimated population size of 275 rabbits.

***Threats and Stressors*****Stressor:** Habitat loss**Exposure:****Response:****Consequence:**

**Narrative:** A primary cause of the Lower Keys rabbit's decline is habitat loss. In the past 20 years, more than half the area of suitable habitat has been destroyed for construction of residential housing, commercial facilities, utility lines, roads, and other infrastructure in the Lower Keys. As of 2003, 55 percent of habitat patches occupied by rabbits were within 320 meters of developed parcels and may be vulnerable to cat predation (Faulhaber 2003). Much of the remaining suitable habitat has been degraded by exotic invasive plants, repeated mowing, dumping of trash, off-road vehicle use, and free-ranging cats. Development has fragmented the sites occupied by rabbits and has eliminated many of the corridors that allow rabbits to move among sites. In urbanized areas where the vegetation has been mowed, dispersing rabbits have no cover from predation. These threats have resulted in a decrease in the number of populations, a decline in the size of the populations, isolation of populations, and the loss of foraging, shelter, and nesting habitat, as well as secondary effects including an increase in road mortalities and increased free-ranging cat predation. Additional threats to the long-term persistence of the rabbit include contaminants, poaching, the proliferation of exotic vegetation, and predation by domestic animals, feral hogs (*Sus scrofa*), and exotic fire ants (*Solenopsis invicta*).

**Stressor:** Predators

**Exposure:****Response:****Consequence:**

**Narrative:** Mortality from free-ranging cats may be the greatest current threat to the persistence of the rabbit (Forys and Humphrey, 1999b). A detailed study of cat diets in the Keys has not been conducted, but rabbits were the largest component of feral cat diets in several studies conducted elsewhere. The number of cats present in the Lower Keys has increased over the past 20 years with the increase in the human residential population. Rabbits of all ages and both sexes are susceptible to cat predation. Feral and domestic cats are present on most of the larger islands that contain some degree of development. Faulhaber (2003) compiled a list of known and likely predators that included the northern harrier (*Circus cyaneus*), bald eagles (*Haliaeetus leucocephalus*), other raptors, eastern diamondback rattlesnakes (*Crotalus adamanteus*), feral and domestic cats, and dogs (*Canis familiaris*). Species known or likely to prey upon young rabbits include raccoons, fire ants, and feral hogs. Imported red fire ants threaten neonate small mammals (Forys et al. 2002). Fire ants have been increasing in marsh habitat and pose a threat to newborn rabbits. Additionally, the non-native black rat poses a predatory threat to neonate rabbits (Dunson and Lazell 1982, Mitchell 1996, Forys et al. 1996).

**Stressor:** Vehicle mortality

**Exposure:****Response:****Consequence:**

**Narrative:** Roads can interfere with home range movements and dispersal and may prevent essential interchange between subpopulations (Forys and Humphrey 1999b). Dispersing males are the most vulnerable to road mortality. Dispersal is the mechanism by which sites where rabbits have been extirpated are repopulated. Since only a portion of the males breed during the year, the loss of these males may lower the likelihood of mating and hence decrease the reproductive potential. Off-road vehicular activities also affect the rabbit through habitat degradation and direct mortality.

**Stressor:** Degraded habitat

**Exposure:****Response:****Consequence:**

**Narrative:** Increased nutrients from septic tanks and fertilizers degrade water quality in habitat of the rabbit. Illegal dumping deteriorates habitat and allows the infestation of exotic plants and animals to occur. Feral hogs destroy rabbit habitat while foraging, but the extent of impact has not been analyzed.

**Recovery****Delisting Criteria:**

The Lower Keys marsh rabbit will be considered for delisting when all the following criteria have been met: 1. At least 13 LKMR populations on eight (8) islands connected by U.S. Highway 1 and five (5) "backcountry" islands exhibit a stable or increasing trend, as evidenced by natural recruitment for multiple generations. (Factor A) 2. The LKMR metapopulation is connected to the extent that genetic diversity can be naturally maintained without translocations or captive breeding. (Factor A, D, E) 3. Predation from non-native species (e.g., Burmese pythons and free-

roaming pets) is low enough for LKMR to remain viable for the foreseeable future. (Factor C, D)  
4. When, in addition to the above criteria, it can be demonstrated that habitat loss associated with sea level rise, development, fire suppression, lack of natural disturbance, and buttonwood encroachment are diminished or reversed such that enough suitable habitat remains in the foreseeable future for LKMR to remain viable. (Factor A, E) (USFWS 2019).

**Recovery Actions:**

- Recovery potential will increase if active management of populations and habitats is undertaken (Forys 1995). Recent studies illustrate that population reductions continue to the present, though presumably at a lesser rate than during the 1970s through 1990s, since urban growth has been somewhat curtailed (Monroe County Growth Management Division 1992, Lopez 2001, Service 1997). Extinction and recolonization dynamics are observed in the metapopulation. However, a net decrease of 8 populations was documented in 106 populations that were surveyed as of 1995 and resurveyed during 2001-2003. During the latter period, translocation efforts resulted in the founding of a population on Little Pine Key, offsetting the net loss by one.

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## SPECIES ACCOUNT: *Tamias minimus atristriatus* (Penasco least chipmunk)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Candidate; (USFWS, 2016)

### **Physical Description**

The Peñasco least chipmunk is grayish-brown mixed with cinnamon-buff on the rump and thighs (Sullivan 1993, p. 1). The Peñasco least chipmunk has pale yellowish orange hindfeet, a light beige, yellowish, or orange belly, and dark underfur (Frey 2010, p. 11). The gray-footed chipmunk (*Tamias canipes*) occurs with the similar Peñasco least chipmunk and they are easily confused in the field (New Mexico Department of Game and Fish (NMDGF) 2008, p. 1). Frey reported that these species can be difficult to distinguish without physically comparing specimens (Frey 2007, p. 17). Specimens of the Peñasco least chipmunk from the Sacramento Mountains had a mean body length of 11.4 centimeters (cm) 4.5 (in) a mean tail length of 9.3 cm (3.7 in), a mean ear length of 1.4 cm (0.6 in), and a mean hindfoot length of 3.0 cm (1.2 in) (Frey 2010, p. 7). An identification key for the subspecies is provided in Frey (2007, pp. 1721).

### **Taxonomy**

The Peñasco least chipmunk was originally described as a distinct species (*Eutamias atristriatus*) based on specimens collected in 1902 (Bailey 1913, pp. 129130). In a revision of the North American chipmunks, Howell (1929) reclassified the Peñasco least chipmunk as *Tamias minimus atristriatus*. *Tamias minimus atristriatus* is genetically distinct from other subspecies of least chipmunk (Sullivan and Petersen 1988, p. 21) and is recognized as a valid subspecies (Wilson and Reeder 2005, entire).

### **Historical Range**

New Mexico: Lincoln and Otero counties. The Peñasco least chipmunk (least chipmunk) (*Tamias* (=Neotamias) *minimus atristriatus*) is endemic to the White Mountains, Otero and Lincoln Counties, and the Sacramento Mountains, Otero County, New Mexico (Frey and Boykin 2007). In the Sacramento Mountains, the least chipmunk was abundant and widespread until the early 1930s (Frey and Boykin 2007, pp. 15, 50). The last verification of persistence of the Sacramento Mountains population of the least chipmunk was in 1966 (Conley 1970, p. 699); however, there were unverified reports of the subspecies in the 1990s from the Sacramento Mountains, Otero County (Ward 2001, p. 234; Frey and Boykin 2007, pp. 1617).

### **Current Range**

New Mexico: Lincoln and Otero counties. The least chipmunk population within the Sacramento Mountains has not been verified since 1966. Although it is unknown whether the Sacramento Mountains population persists, most authors believe it has been extirpated (Hope and Frey 2000 p. 10; Frey and Boykin 2007, pp. 12-18, 50; Wampler 2007; Frey 2009, p. 5). The persistence of the White Mountain population of the least chipmunk was last verified in 1998 and 2000 (Ortiz 1999; Hope and Frey 2000).

### **Distinct Population Segments Defined**

No.

**Critical Habitat Designated**

No;

***Life History*****Feeding Narrative**

Adult: Peñasco least chipmunks forage mainly on the ground or in shrubs (Hoffmeister 1986, p. 15). The seeds of shrubs and forbs are their main food source, though they also feed on arthropods, leaves, fruits, flowers, and fungi (Bailey 1931, p. 91; Vaughn 1974, pp. 770772). The least chipmunk does not develop fat deposits in the fall, but relies on brief periods of activity to consume cached food for survival over the winter (Verts and Carraway 2001).

**Reproduction Narrative**

Adult: Least chipmunks dig burrows for nesting, often under large rocks, but may also use tree cavities or other natural structures (Verts and Carraway 2001, pp. 67). In spring, females typically produce one litter of 4-5 pups (Skryja 1974, p. 223). The average life span of the least chipmunk is 0.7 years (Erlien and Tester 1984, p. 2).

**Habitat Narrative**

Adult: The Peñasco least chipmunk has been found in two different and distinctive habitat types in New Mexico: 1) the ponderosa pine forest zone in the Sacramento Mountains; and 2) high elevation talus slopes and glacial cirques surrounded by Englemann spruce (*Picea engelmanni*), quaking aspen (*Populus tremuloides*), corkbark fir (*Abies lasiocarpa*), and Douglas fir (*Pseudotsuga menziesii*) above treeline in the White Mountains (Sullivan 1993; p. 3; Frey and Boykin 2007, pp. 2728). In the Sacramento Mountains, historic mature ponderosa pine forests have been described as lacking lower limbs and providing an open structure with dense grass cover (U.S. Forest Service (Forest Service) 2002, pp. BiiBiii; Frey and Boykin 2007, p. 51). The Sacramento Mountains population appears to have been nearly exclusively associated with large open mature stands of ponderosa pine forest, which have mostly been eliminated and subsequently replaced by dense coniferous stands of young trees that are unsuitable for the least chipmunk (Kaufmann et al. 1998, pp. 4648; Frey and Boykin 2007, pp. 27, 51). In contrast, in the White Mountains, which are about 40 kilometers (km) (30 miles (mi)) north of the Sacramento Mountains, the least chipmunk has only been associated with patches of rock and talus above treeline within close proximity of Sierra Blanca Peak (Frey and Boykin 2007, p. 28). Least chipmunks dig burrows for nesting, often under large rocks, but may also use tree cavities or other natural structures (Verts and Carraway 2001, pp. 67).

***Dispersal/Migration******Population Information and Trends*****Population Trends:**

Decline

**Species Trends:**

Decline

**Number of Populations:**

Unknown, but likely only one

**Population Size:**

Unknown

**Population Narrative:**

The distribution and abundance of the least chipmunk has dramatically declined since the early 20th century (Frey and Boykin 2007, p. 50). Despite field surveys in 1981-1982, 1991-1996, 2000, and 2005-2006 (reviewed in Frey and Boykin 2007, entire), the least chipmunk population within the Sacramento Mountains has not been verified since 1966. Although it is unknown whether the Sacramento Mountains population persists, most authors believe it has been extirpated (Hope and Frey 2000 p. 10; Frey and Boykin 2007, pp. 12-18, 50; Wampler 2007; Frey 2009, p. 5). From at least 1994, the Lincoln National Forest has reported that the subspecies no longer occurs on the Sacramento Ranger District within its historically occupied habitat (Forest Service 1993, p. 26; 2008, p. 31; 2011, p. 62). The persistence of the White Mountain population of least chipmunk was last verified in 1998 and 2000 (Ortiz 1999; Hope and Frey 2000). The core of this population is likely associated with a large area of rocky habitat on Sierra Blanca Peak and probably extended to adjacent areas such as Buck Mountain on Forest Service lands (Frey and Boykin 2007, p. 50). The subalpine areas in the White Mountains likely contain suitable habitat for the least chipmunk because this area has remained relatively unaltered from historic conditions (Frey and Boykin 2007, p. 40).

***Threats and Stressors***

**Stressor:** Timber harvest

**Exposure:** Habitat degradation, destruction, and fragmentation

**Response:**

**Consequence:**

**Narrative:** The Peñasco least chipmunk faces threats from present or threatened destruction, modification, and curtailment of its habitat from the alteration or loss of mature ponderosa pine forests in one of the two historically-occupied areas. The subspecies is further threatened by residential development, fire suppression and exclusion, and high-intensity fire. In the Sacramento Mountains, the subspecies habitat requirements of open ponderosa pine forests have essentially been eliminated due to the historical impact of these activities. Ongoing impacts due to these activities continue to further degrade these habitats and further preclude recovery of these areas. Further, the existing regulatory mechanisms have not been adequate to prevent the continuing decline of the least chipmunk. The documented decline in occupied localities, in conjunction with the small numbers of individuals captured, are linked to widespread habitat alteration (Frey and Boykin 2007). Moreover, the highly-fragmented nature of its current distribution is a significant contributor to the vulnerability of this subspecies and increases the likelihood of very small, isolated populations being extirpated (Factor E). As a result of this fragmentation, even if suitable habitat exists (or is restored) in the Sacramento Mountains, the likelihood of natural recolonization of historic habitat or population expansion from the White Mountains is extremely remote.

**Stressor:** Livestock grazing

**Exposure:**

**Response:****Consequence:**

**Narrative:** Grazing has contributed to the altered composition of ponderosa pine forests in the Sacramento Mountains, particularly in James Canyon, which has been heavily overgrazed (Alexander et al. 1984, p.16; Sullivan et al. undated, p. 2). Overgrazing, drought, and erosion eliminated continuous stretches of grass that would have historically carried surface fires necessary for maintaining the open ponderosa pine habitat utilized by the least chipmunk. As a result, overgrazing contributed to the risk of high-intensity fire in the Sacramento Mountains. In 1964, White Mountain Wilderness status was conferred on the adjacent 19,533 hectare-area (48,266 acres) and all grazing halted (Dyer and Moffett 1999, p. 451); therefore grazing is not an issue in the White Mountains.

**Stressor:** Residential development

**Exposure:****Response:****Consequence:**

**Narrative:** The least chipmunk population in the White Mountains is not currently threatened by residential development. In the Sacramento Mountains, however, the subspecies habitat has been altered by development associated with private lands in James Canyon. The James Canyon area historically supported more stands of pure ponderosa pine than any other area on the Lincoln National Forest (Kaufmann et al. 1998, p. 46). Residential development of private land continues to fragment the small areas of remnant least chipmunk habitat within the Sacramento Mountains. The human population in James Canyon and surrounding areas has increased to several thousand residents over the last several decades (Forest Service 1999, p. 3; 2000 p. 43). This area is now the most heavily developed part of the Sacramento Mountains with campgrounds, a ski area, and numerous subdivisions with summer and year-round homes (Kaufmann et al. 1998, pp. 46, 48). Residential development activities within James Canyon and other surrounding areas in the Sacramento Mountains will continue to destroy or modify areas that potentially could be restored for use by the subspecies. Further, the extensive fragmentation of the historic habitat within the Sacramento Mountains and the isolation of this area from the extant Sierra Blanca locality (separated by 40 km (30 mi)) indicates that natural recolonization of the Sacramento Mountains by the Peñasco least chipmunk is highly unlikely.

**Stressor:** High-intensity crown fires

**Exposure:****Response:****Consequence:**

**Narrative:** Fire exclusion, and the resulting overstocked, dense forests has also significantly increased the potential for high-intensity, destructive crown fires (Covington and Moore 1992, p. 94; Allen et al. 2002, p. 1420). In fact, since 1921, seven large stand-replacing type fires have occurred in the area east of the Village of Cloudcroft that historically contained habitat of the least chipmunk (Forest Service 2002, p. 3.18). Five of these fires have burned since 1993 (Forest Service 2002, p. 3.18). Within the watersheds that historically supported the least chipmunk in the Sacramento Mountains, 94 percent of the area is highly susceptible to stand-replacing fires (Forest Service 2002, p. 3.20). During June 2012, the Little Bear Fire burned 17,806 ha (44,000 ac) within the White Mountains, including Buck Mountain and adjacent areas (Forest Service 2012a, pp. 1, 4, 10). It is unknown whether the subspecies was affected because suitable habitat in the White Mountains is generally associated with patches of rock and talus above treeline that are

unlikely to burn with high severity. Nevertheless, because of continued exclusion and suppression of fire in the Sacramento Mountains, we conclude that further curtailment of the range of the subspecies is likely through the prevention of restoration of ponderosa pine and increasing the risk of high-intensity fire. The continued implementation of the Forest Services strategy of fire suppression and exclusion is likely to remove and effectively eliminate, degrade, or fragment any remaining potential habitat of the least chipmunk in the Sacramento Mountains.

**Stressor:** Inadequate regulatory mechanisms

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Even though the least chipmunk is state-listed (NMDGF) as endangered, this designation only conveys protection from collection or intentional harm; no New Mexico State statutes address habitat protection, indirect effects, or other threats to the subspecies identified by the State as endangered. Because most of the risks to the least chipmunk are from effects to habitat, protecting individuals from direct take will not ensure long-term protection of the subspecies. The least chipmunk has been on the Regional Foresters Sensitive Species List since 1990 (Forest Service 1999a). The Regional Foresters Sensitive Species List policy is applied to projects implemented under the 1982 National Forest Management Act Planning Rule (49 FR 43026, September 30, 1982). All existing plans continue to operate under the 1982 Planning Rule and all of its associated implementing regulations and policies; however, all new plans and plan revisions must conform to the new 2012 planning requirements (68 FR 21162, April 9, 2012). The Lincoln National Forest will begin to revise their Forest Plan in 2015. When this Forest Plan is revised, it is unclear whether the 2012 planning requirements will provide adequate protection to the least chipmunk. In the interim, the Forest Plan will continue to operate under the 1982 planning rule (Forest Service 2012, entire; 2013, entire). However, even if increased protections were afforded to the subspecies due to its Forest Service sensitive-species status and potential updated planning rule, the single extant population in the White Mountains is likely insufficient to conserve the Peñasco least chipmunk.

### ***Recovery***

#### **Recovery Actions:**

- Ponderosa pine habitat restoration (e.g., following the principles outlined in Allen et al. 2002, pp. 14241428) will be necessary before significant risk reduction for the Peñasco least chipmunk is possible.
- Reintroductions of the subspecies in the Sacramento Mountains will be necessary before significant risk reduction for the Peñasco least chipmunk is possible.
- Additional surveys should be conducted for the least chipmunk in the Sacramento and White Mountains, including private lands and the Mescalero Apache Nation.
- A conservation strategy should be developed for the subspecies, to guide coordinated conservation efforts by multiple partners.

#### ***Conservation Measures and Best Management Practices:***

- Ponderosa pine habitat restoration (e.g., following the principles outlined in Allen et al. 2002, pp. 14241428) will be necessary before significant risk reduction for the Peñasco least chipmunk is possible.

- Reintroductions of the subspecies in the Sacramento Mountains will be necessary before significant risk reduction for the Peñasco least chipmunk is possible.

**References**

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Southwest Region

18 p.

USFWS. 2014. U.S. Fish and Wildlife Service Species Assessment and Listing Priority Assignment Form for *Tamias minimus atristriatus* (Peñasco least chipmunk)

## SPECIES ACCOUNT: *Tamiasciurus hudsonicus grahamensis* (Mount Graham red squirrel)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; June 3, 1987 (52 FR 20994).

### **Physical Description**

The Mount Graham red squirrel is a small, arboreal (tree-dwelling) rodent. They are grayish brown in color, with rusty and yellowish markings along the back. In summer, a thin, black lateral line separates the upper parts from the whitish underparts. The tail is fluffy and the ears are slightly tufted in winter. The cheek teeth number 16 (P1/1, M3/3), and are low-crowned and tuberculate (with small knob-like processes); the skull is rounded, with the postorbital process present. The species ranges from 270 to 385 millimeters (mm) (10.8 to 15.4 inches [in.]) in total length, and from 92 to 158 mm (3.7 to 6.3 in.) in tail length. They weigh about 224 grams (8 ounces). Unlike many other squirrels, the Mount Graham species does not have a white-fringed tail (USFWS 2011; USFWS 2015).

### **Taxonomy**

There are 25 recognized subspecies of American red squirrels (*Tamiasciurus hudsonicus*) in North America. In the southern part of its range, the red squirrel is restricted to montane forests. There are two recognized subspecies in Arizona: the Mogollon red squirrel (*T. hudsonicus mogollonensis*), which is found throughout northern Arizona; and the Mount Graham red squirrel (*T. hudsonicus grahamensis*), which is found only in the Pinaleno Mountains in southeastern Arizona. The Mount Graham red squirrel is the southernmost subspecies in North America. First described in 1894 by J.A. Allen, the Mount Graham red squirrel was designated as a separate subspecies, based on differences in pelage (fur) and its apparent isolation from other red squirrel populations for at least 10,000 years. The Mount Graham red squirrel is slightly smaller than the Mogollon red squirrel in body measurements, including total body, hind foot, and skull length. The skull is also narrower postorbitally (behind the eyes) than that of Mogollon red squirrel. Research with both cellular and molecular genetic analysis, in conjunction with morphological and ecological considerations, demonstrates that the Mount Graham red squirrel is a distinct population that warrants subspecific status. There is no migration between these two subspecies (USFWS 2011).

### **Historical Range**

Historically, Mount Graham red squirrel inhabited about 4,478 hectares (ha) (11,733 acres [ac.]) in the spruce-fir, mixed-conifer, and the ecotone zone between these two distinct forest types in the higher elevations (generally above 8,000 feet [ft.]) of the Pinaleno Mountains, in Graham County, Arizona. Historically, suitable habitat once existed (in the 1960s) between the West Peak portion of the mountain range and the greater portion of habitat a little farther east on the mountain (USFWS 2008). Surveys have located red squirrel middens (piles of cone scales in which squirrels cache live, unopened cones as an over-wintering food source) at elevations as high as 3,268 meters (m) (10,722 ft.) and as low as 2,353 m (7,720 ft.) (USFWS 2011).

### **Current Range**

Currently, Mount Graham red squirrel occupy a portion of their previous range. Their current range is entirely within the Safford Ranger District of the Coronado National Forest. Their elevational range is about 2,375 to 3,265 m (7,792 to 10,712 ft.) (NatureServe 2015). Information at the time of their listing indicated that Mount Graham red squirrel preferred the higher-elevation spruce-fir and ecotone zones of the mountain. Since the 1990s, the highest numbers of middens have occurred in the ecotone zone rather than in the spruce-fir or the mixed-conifer alone (USFWS 2008; USFWS 2011).

**Distinct Population Segments Defined**

No

**Critical Habitat Designated**

Yes; 1/5/1990.

**Legal Description**

On January 5, 1990, the U.S. Fish and Wildlife Services designated critical habitat for the Mount Graham red squirrel (*Tamiasciurus hudsonicus grahamensis*) under the Endangered Species Act of 1973, as amended (55 FR 425-429). The critical habitat areas are all within the Coronado National Forest in Graham County, Arizona. The three designated areas contain major concentrations of the Mount Graham red squirrel, and the habitat necessary to its survival, including cover, food sources, nest sites, and midden sites.

**Critical Habitat Designation**

Arizona. Areas of land, water, and airspace in the Coronado National Forest, T. 8 S., R. 24 E., and T. 9 S., R. 24 E. (Gila and Salt River Meridian), Graham County, with the following components:

1. Hawk Peak-Mount Graham Area. The area above the 10,000-foot (3,048-meter) contour surrounding Hawk Peak and Plain View Peak, plus the area above the 9,800-foot (2,987-meter) contour that is south of lines extending from the highest point of Plain View Peak eastward at 90° (from true north) and southwestward at 225° (from true north).
2. Heliograph Peak Area. The area on the north-facing slope of Heliograph Peak that is above the 9,200-foot (2,804-meter) contour surrounding Heliograph Peak and that is between a line extending at 15° (from true north) from a point 160 feet (49 meters) due south of the horizontal control station on Heliograph Peak and a line extending northwestward at 300° (from true north) from that same point.
3. Webb Peak Area. The area on the east-facing slope of Webb Peak that is above the 9,700-foot (2,957-meter) contour surrounding Webb Peak and that is east of a line extending due north and south through a point 160 feet (49 meters) due west of the horizontal control station on Webb Peak.

The Mount Graham Red Squirrel Refugium established by the AICA has the same boundary as the designated critical habitat boundary surrounding Hawk and Plain View peaks (about 688 ha [1,700 acres]), but does not include critical habitat on Heliograph or Webb Peaks. The main attribute of these areas at that time was the existing dense stands of mature (about 300 years old) spruce-fir forest. Unfortunately, due to damage by insects, wildfire, and associated fire

suppression activities, only approximately 112 ha (277 ac) of designated critical habitat currently provide potential habitat for the red squirrel (Hatten, unpub. data).

**Primary Constituent Elements/Physical or Biological Features**

The major constituent element is dense stands of mature spruce-fir forest.

**Special Management Considerations or Protections**

Not available

***Life History*****Feeding Narrative**

Adult: The Mount Graham red squirrel diet consists of conifer seeds from closed cones, fungi and rusts, pollen (pistillate cones) and cone buds, cambium of conifer twigs, bones, and berries and seeds from broadleaf trees and shrubs. It occasionally feeds on invertebrates and small vertebrates (NatureServe 2015). Different foods are used seasonally: pollen and buds in the spring, bones by females during lactation, fungi in the spring and late summer, and closed cones low in lipids in the early summer. Closed cones high in lipids are stored for winter-time use. In the Pinaleño Mountains, Mount Graham red squirrel eat seeds and store cones from Englemann spruce (*Picea engelmannii*), white fir (*Abies concolor*), Douglas-fir (*Pseudotsuga menziesii*), corkbark fir (*Abies lasiocarpa*), and southwestern white pine (*Pinus strobiformis*). Mount Graham red squirrel also eat false truffles and other fungi, which appear during spring snowmelt and after summer rains begin (USFWS 2011).

**Reproduction Narrative**

Adult: Mount Graham red squirrel can reproduce after their first winter. After the second winter, all squirrels are considered adults. Mount Graham red squirrel breed from February through early April. The onset of breeding is not well understood, but has been related to the quality and quantity of the spring bud crop on conifers. The proportion of yearling and adult squirrels that breed varies widely from year to year and also appears to be crudely related to seed crop availability. Two breeding seasons per year have been reported in a few populations of red squirrels, including the populations in central Arizona and one female Mount Graham red squirrel. However, the percentage of females that produce two litters per year is likely rare, and only occurs during periods of high food availability. On average, gestation period is 35 to 40 days, and female Mount Graham red squirrel give birth to fewer young than other red squirrels (reported means are 2.35 and 2.15 for Mount Graham red squirrel; 3.69 and 3.72 for other red squirrels). Nests are constructed in natural hollows or abandoned cavities made by other animals, such as woodpeckers. Red squirrel nests can be in a tree hollow, hollow snag, or downed log, or among understory branches of a sheltered canopy. Mount Graham red squirrel appear to favor cavity nests over nests built in the tops of trees (also called dreys), whereas the nearest population of red squirrels, the Mogollon red squirrel, predominantly uses dreys. Once occupied, nests are often enlarged by squirrels and can be anywhere from 0 m to more than 610 m (0 to 2,000 ft.) away from the midden (USFWS 2008; USFWS 2011).

**Geographic or Habitat Restraints or Barriers**

Adult: The Mount Graham red squirrel is restricted to the cooler and wetter habitat that is now isolated atop the highest mountain in southern Arizona, Mount Graham, at 3,267 m (10,720 ft.) above sea level (USFWS 2008; USFWS 2011).

**Spatial Arrangements of the Population**

Adult: Clumped

**Environmental Specificity**

Adult: Narrow/specialist.

**Tolerance Ranges/Thresholds**

Adult: Low

**Site Fidelity**

Adult: High

**Dependency on Other Individuals or Species for Habitat**

Adult: None

**Habitat Narrative**

Adult: The Mount Graham red squirrel is primarily habitat-limited. Suitable habitat for Mount Graham red squirrel consists of high elevation (2,375 to 3,265 m [7,792 to 10,712 ft.]) spruce-fir forest, mixed-conifer forests, and ecotone zones of the Coronado National Forest (NatureServe 2015; USFWS 2008; USFWS 2011). Mount Graham red squirrels require reliable and adequate conifer cone crops for food as well as microclimatic conditions suitable for storage of closed cones. These conditions exist in mature to old-growth stands that have closed canopies, and their ecotones, which may increase fungal food supplies. Other elements that increase the quality of habitat are downed logs, snags, and interlocking branch networks. These characteristics provide adequate food resources; perching, storage, and nesting sites; runways that allow cone retrieval in the winter; and escape routes for avoidance of predators. Nest site selection is important for thermoregulation, cone and fungal storage, and predator avoidance. Nests are often cavities in snags or downed logs, but may be constructed of leaves (also termed dreys); underground burrows may also be used. Nest sites are typically in stands of trees with large-diameter and significant canopy closure, and interdigitation with adjacent trees. Midden sites (piles of cone scales in which squirrels cache live, unopened cones as an over-wintering food source) require cool temperatures and a moist environment for optimal storage of cones. Midden sites are near nests and have similar habitat characteristics as nests. Territories are usually centered around middens, likely because they contain 1 to 2 years of cone resources and are critical to Mount Graham red squirrel survival. Territory sizes for red squirrels are typically are less than 1 ha (2.47 ac.), and may increase markedly during years of food shortage or in suspected marginal habitat (USFWS 2011).

***Dispersal/Migration*****Motility/Mobility**

Adult: Low; juveniles disperse approximately 2 kilometers (1.2 miles) (USFWS 2008).

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Low

**Immigration/Emigration**

Adult: Unlikely

**Dependency on Other Individuals or Species for Dispersal**

Adult: No

**Dispersal/Migration Narrative**

Adult: The Mount Graham red squirrel has low motility and requires contiguous suitable habitat for survival. Devastating losses of trees have dictated changes in the Mount Graham red squirrel's opportunities for foraging, nesting, and dispersal, and the current habitat of the red squirrel is primarily in the mixed-conifer forest and ecotone rather than the spruce-fir forest (USFWS 2008; USFWS 2011).

**Additional Life History Information**

Adult: Dispersal corridors and habitat have been lost to drought, fire, and roads.

***Population Information and Trends*****Population Trends:**

A statistical trend cannot be determined based on the population estimates to date. However, recent population estimates show two periods of relative stability punctuated by a spike in population during 1998 to 2000 (USFWS 2008).

**Species Trends:**

Relatively stable in the short term (less than or equal to 10 percent change) (NatureServe 2015).

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Population Growth Rate:**

Stable

**Number of Populations:**

1 to 5 (NatureServe 2015). Mount Graham red squirrel is restricted to the suitable spruce-fir and mixed-conifer forests of Mount Graham.

**Population Size:**

The current total population is estimated at 250 to 1,000 individuals (NatureServe 2015). The mean of Mount Graham red squirrel population estimates from 1991 to 1997 was 327 (range

259 to 423). From 1998 through 2000, the mean population was 525 (range 462 to 583). Population estimates from 2002 through 2009 vary from 199 to 346 (USFWS 2011).

**Resistance to Disease:**

A number of parasites and several infectious agents have been reported from red squirrels in various parts of their range. However, parasite and disease infestations are not known to significantly contribute to the mortality of Mount Graham red squirrel (USFWS 2011).

**Adaptability:**

Low

**Additional Population-level Information:**

A combination of drought, poor conifer cone crops, two major catastrophic wildfires, and insect damage (with a resultant loss of habitat) has likely caused recent Mount Graham red squirrel population reductions (USFWS 2008). The population size of the Mount Graham red squirrel throughout its range has been estimated and tracked since 1986 by an interagency team. Assumptions for estimating abundance are: (1) squirrel occupancy can be inferred from signs of recent caching and digging and from the condition of midden material, even when squirrels are not directly observed; and (2) one squirrel occupies only one active midden at a time (USFWS 2011).

**Population Narrative:**

There are between one and five populations of the Mount Graham red squirrel, restricted to spruce-fir and mixed conifer forest of Mount Graham. The population size of Mount Graham red squirrel throughout its range has been estimated and tracked since 1986 by an interagency team. A statistical trend cannot be determined based on the population estimates to date. However, recent population estimates show two periods of relative stability, punctuated by a spike in population from 1998 through 2000. The mean of Mount Graham red squirrel population estimates from 1991 through 1997 was 327 (range 259 to 423). From 1998 through 2000, the mean population was 525 (range 462 to 583); and from 2001 through 2008, the mean population was 272 (range 199 to 362). A combination of drought, poor conifer cone crops, two major catastrophic wildfires, and insect damage (with a resultant loss of habitat) has likely caused recent Mount Graham red squirrel population reductions (USFWS 2008).

**Threats and Stressors**

**Stressor:** Catastrophic wildfire

**Exposure:** Destruction of suitable habitat.

**Response:** Several, including loss of cover, food source, and increased predation.

**Consequence:** Decreased population numbers.

**Narrative:** Between 1996 and 2004, 16,800 ac. of oak, ponderosa pine, mixed-conifer, ecotone, and spruce-fir forest types burned within the range of Mount Graham red squirrel. The threat of catastrophic wildfire to the remaining Mount Graham red squirrel habitat remains high due to the tons of dead and down fuel load; overcrowded tree conditions leading to poor forest health; dense thickets of small-diameter trees; dry winters; lightning strikes; drought conditions; and the likelihood of an increasing number of mountain visitors (which provide ignition sources) (USFWS 2008).

**Stressor:** Climate change

**Exposure:** Permanent loss of remaining habitat.

**Response:** Disappearance of suitable ecosystem to support Mount Graham red squirrel.

**Consequence:** Extinction of population.

**Narrative:** Mount Graham red squirrel depend on the ability of the forest to produce reliable and adequate conifer cone crops for food, as well as microclimatic conditions suitable for storage of closed cones. These habitat characteristics provide red squirrels with adequate food resources; perching, storage, and nesting sites; runways that allow cone retrieval in the winter; and escape routes for avoidance of predators (USFWS 2011). Long-term drying and warming trends associated with global climate change are predicted for the southwest, which may lead to additional drought stress on the trees that support Mount Graham red squirrel (USFWS 2008).

**Stressor:** Insect damage

**Exposure:** Damaged or dead trees.

**Response:** Stressed or dying trees do not produce the amount of cones needed to support the Mount Graham red squirrel.

**Consequence:** Decreased population numbers.

**Narrative:** From 1996 to date, a massive four-insect outbreak destroyed most of the spruce-fir forest (including small trees and saplings) on top of the mountain. This outbreak was likely driven by warm winters that allowed the insects to overwinter, and a 10-plus-year drought that made the trees more susceptible to infestation (USFWS 2008).

**Stressor:** Geographic isolation and low population numbers

**Exposure:** Genetic inbreeding.

**Response:** Increased risk of extinction due to genetic and demographic problems associated with small population sizes.

**Consequence:** Reduced vitality and health of existing population.

**Narrative:** Low genetic variability, often characteristic of small populations, is a concern because deleterious alleles are expressed more frequently, disease resistance might be compromised, and there is little capacity for evolutionary change in response to environmental change (USFWS 2008).

**Stressor:** High avian predation rates (compared to other red squirrel populations)

**Exposure:** Predation

**Response:** Low adult and juvenile survival.

**Consequence:** Decreased population numbers.

**Narrative:** Mount Graham red squirrels have low adult and juvenile survival and high avian predation rates compared to other red squirrel populations. Up to 75 to 80 percent of the mortality experienced by Mount Graham red squirrel appears to be due to predation, most of which is caused by raptors (USFWS 2011).

### ***Recovery***

#### **Reclassification Criteria:**

Restore and maintain a mosaic of at least 70 percent of the range, or 5,00 ha (13,838 ac) that meets the criteria for habitat listed below. This amount of habitat is slightly greater than what was available to the red squirrel at the time of its listing in 1987.

Areas are considered habitat if they meet the following conditions: 1) They are within the mixed conifer, ecotone, and spruce-fir series AND 2) They are above 2,744 m (9,000 ft) OR 3) If they are below 2,744 m (9,000 ft), they meet the following criteria: a) > 2,353 m (7,720 ft) elevation b) north or east aspect c) < 45-degree slope

Management agreements among the U.S. Fish and Wildlife Service (USFWS), Coronado National Forest, and Arizona Game and Fish Department that will protect this habitat indefinitely are in place and being implemented.

Maintain a self-sustaining population of Mount Graham red squirrels sufficient to ensure the species' survival with a statistical confidence (90 percent) that the rate of increase over a time of 10 years (five generations) is 20 percent or greater of the known population, as measured by mountain-wide monitoring.

None

**Delisting Criteria:**

Restore and maintain a mosaic of at least 80 percent of the range, or 6,400 ha (15,815 ac.) that meets the criteria for habitat listed below.

Areas are considered habitat if they meet the following conditions: 1) They are within the mixed conifer, ecotone, and spruce-fir series AND 2) They are above 2,744 m (9,000 ft) OR 3) If they are below 2,744 m (9,000 ft), they meet the following criteria: a) > 2,353 m (7,720 ft) elevation b) north or east aspect c) < 45-degree slope

Management agreements among the USFWS, Coronado National Forest, and Arizona Game and Fish Department that will protect this habitat indefinitely are in place and being implemented.

Once downlisting criteria are achieved, maintain a self-sustaining population of Mount Graham red squirrels sufficient to ensure the species' survival with statistical confidence (90 percent) that the rate of increase over the following 20 years (10 generations) is increasing or stable, as measured by mountain-wide monitoring.

None

**Recovery Actions:**

- Protect and manage existing habitat and population.
- Restore and create habitat to allow for the existence of a viable and robust population.
- Research the conservation biology of the Mount Graham red squirrel, with the objective of facilitating efficient recovery.
- Develop support and build partnerships to facilitate recovery.
- Monitor progress and practice adaptive management in which the recovery plan and management actions are revised to reflect new information developed through research and monitoring.

***Conservation Measures and Best Management Practices:***

- Employing best management practices to conduct projects, while protecting and restoring habitat.

- Conducting vegetation monitoring.
- Employing best management practices for recreational activities (a recreation plan is being implemented to protect the red squirrel).
- Coordinating with Arizona Game and Fish Department for monitoring of red squirrel population.
- Conducting fire suppression and wildfire risk abatement activities.

***Additional Threshold Information:***

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## SPECIES ACCOUNT: *Thomomys mazama glacialis* (Roy Prairie pocket gopher)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Threatened; Pacific Region (R1) (USFWS, 2016)

### **Physical Description**

Adult Mazama pocket gophers are reddish brown to black above, and the underparts are lead-colored with buff-colored tips. The lips, nose, and patches behind the ears are black; the wrists are white. Adults range from 7 to 9 inches (189 to 220 millimeters (mm)) in total length, with tails that range from 2 to 3 inches (45 to 85 mm)(Verts and Carraway 2000, p.2). Mazama pocket gophers are morphologically similar to other species of pocket gophers that exploit a subterranean existence. They are stocky and tubular in shape, with short necks, powerful limbs, long claws, and tiny ears and eyes. Their short, nearly hairless tails are highly sensitive and probably assist when navigating tunnels. The “pockets” are external, fur-lined cheek pouches on either side of the mouth that are used to transport nesting material and plant cuttings. Mazama pocket gophers reach reproductive age in the spring of the year after their birth and produce litters between spring and early summer. Litter size ranges from one to nine (Wight 1918, p. 14), with an average of five (Scheffer 1938, p. 222). They do not hibernate in winter; they remain active throughout the year (Case and Jasch 1994, p. B-20) (USFWS, 2016).

### **Taxonomy**

Although the species *Thomomys mazama*, or Mazama pocket gopher, includes numerous subspecies that are found in the States of Washington, Oregon, and California, only the subspecies found in the State of Washington have recently been considered for listing. The Mazama pocket gopher complex consists of 15 subspecies, eight of which occur only in Washington, five of which occur only in Oregon, one that occurs only in California, and one subspecies with a distribution that spans the boundary between Oregon and California (Hall 1981, p. 467). The first pocket gophers collected in western Washington were considered subspecies of the northern pocket gopher (*Thomomys talpoides*)(Goldman 1939), until 1960 when the complex of pocket gophers found in western Washington was determined to be more similar to the western pocket gopher (*T. mazama*)(Johnson and Benson 1960, p. 20). Eight western Washington subspecies of Mazama pocket gopher (*T. mazama*, ssp. *couchi*, *glacialis*, *louiei*, *melanops*, *pugetensis*, *tacomensis*, *tumuli*, and *yelmensis*) have been identified (Hall 1981, p. 467). *Thomomys mazama* is recognized as a valid species by the Integrated Taxonomic Information System (ITIS 2012). Although there have been suggestions that potential changes to the classification of some of these subspecies should be considered, we have no information to suggest that any of the presently recognized subspecies are the subject of serious dispute. We follow the subspecies designations of Verts and Carraway (2000), as this text represents the currently accepted taxonomy for the species *T. mazama*. Verts and Carraway (2000, p.1) recognize *T. m. glacialis*, *pugetensis*, *tumuli*, and *yelmensis* as separate subspecies (the Roy Prairie, Olympia, Tenino, and Yelm pocket gophers, respectively) based on morphological characteristics, distribution, and differences in number of chromosomes. Due to the close proximity of the four subspecies located in Thurston and Pierce Counties, and the fact that at least three of them occur in the same clade, we refer to these four subspecies (*T. m. glacialis*,

pugetensis, tumuli, and yelmensis) as “the four Thurston/Pierce subspecies” of the Mazama pocket gopher (USFWS, 2016).

**Current Range**

In Washington, Mazama pocket gophers are found west of the Cascade Mountain Range, in the Olympic Mountains and in the Puget Sound trough, with an additional single locality known from Wahkiakum County (Verts and Carraway 2000, p.3). Their populations are concentrated in well-drained friable soils often associated with glacial outwash. The Roy Prairie pocket gopher (*Thomomys mazama glacialis*) is found in the vicinity of the Roy Prairie and on JBLM in Pierce County. The subspecies was described as plentiful in 1983 but by 1993 the type locality was described as a “small population” (Steinberg 1996, p. 24). Due to proximity to the subspecies’ type locality, it is likely that the 91st Division Prairie and Marion Prairie in Pierce County support this subspecies. Soil series and soil series complexes in and around this area that may support pocket gophers include Alderwood, Everett, Everett-Spanaway complex, Everett-Spanaway-Spana complex, Nisqually, Spana-Spanaway-Nisqually complex, and Spanaway (USFWS, 2016).

**Critical Habitat Designated**

No;

***Life History*****Feeding Narrative**

Adult: Pocket gophers are generalist herbivores and their diet includes a wide variety of plant material, including leafy vegetation, succulent roots, shoots, and tubers. In natural settings pocket gophers play a key ecological role by aerating soils, activating the seed bank, and stimulating plant growth, though they can be considered pests in agricultural systems. In prairie and meadow ecosystems, pocket gopher activity plays an important role in maintaining species richness and diversity. Foraging primarily takes place below the surface of the soil, where pocket gophers snip off roots of plants before occasionally pulling the whole plant below ground to eat or store in caches. If above-ground foraging occurs, it’s usually within a few feet of an opening and forage plants are quickly cut into small pieces and carried back to the nest or cache (Wight 1918, p. 12). Any water they need is obtained from their food (Gettinger 1984, pp. 749-750; Wight 1918, p. 13). The probability of pocket gopher occupancy is much higher in areas with less than 10 percent woody vegetation cover (Olson 2011a, p. 16), presumably because such vegetation will shade out the forbs, bulbs, and grasses that pocket gophers prefer to eat, and high densities of woody plants make travel both below and above the ground difficult (USFWS, 2016).

**Reproduction Narrative**

Adult: Pocket gophers reach sexual maturity during the spring of the year following their birth, and generally produce one litter per year (Case and Jasch 1994, p. B-20), though timing of sexual maturity has been shown to vary with habitat quality (Patton and Brylski 1987, p. 502; Patton and Smith 1990, p. 76). Gestation lasts approximately 18 days (Andersen 1978, p. 421; Schramm 1961, p. 169). Young are born in the spring to early summer (Wight 1918, p. 13), and are reared by the female. Aside from the breeding season, males and females remain segregated in their own tunnel systems. There are 1-9 pups per litter (averaging 5), born without hair, pockets, or teeth, and they must be kept warm by the mother or “packed” in dried vegetation (Case and Jasch 1994, p. B-20; Wight 1918, p. 14). Juvenile pelage starts growing in at just over a week

(Andersen 1978, p. 420). The young eat vegetation in the nest within three weeks of birth, with eyes and ears opening and pockets developing at about a month (Andersen 1978, p. 420; Wight 1918, p. 14). At six weeks they are weaned, fighting with siblings, and nearly ready to disperse (Andersen 1978, p. 420; Wight 1918, p. 15), which usually occurs at about two months of age (Stinson 2005, p. 26). They attain their adult weight between four and five months of age (Andersen 1978, pp. 419, 421). Most pocket gophers live only a year or two, with few living to three or four years of age (Hansen 1962, pp. 152, 153; Livezey and Verts 1979, p. 39) (USFWS, 2016).

### **Spatial Arrangements of the Population**

Adult: Clumped (USFWS, 2016)

### **Habitat Narrative**

Adult: The Mazama pocket gopher (pocket gopher) is associated with glacial outwash prairies in western Washington, an ecosystem of conservation concern (Hartway and Steinberg 1997, p. 1), as well as alpine and subalpine meadows and other meadow-like openings at lower elevations. Steinberg and Heller (1997, p. 46) found that pocket gophers are even more patchily distributed than are prairies, as there are some seemingly high quality prairies within the species' range that lack pocket gophers; e.g., Mima Mounds Natural Area Preserve (NAP), and 13th Division Prairie on Joint Base Lewis-McChord (JBLM). Pocket gopher distribution is affected by the rock content of soils, drainage, forage availability, and climate (Case and Jasch 1994, p. B-21; Hafner et al. 1998, p. 279; Reichman 2007, pp. 273-274; Steinberg and Heller 1997, p. 45; Stinson 2005, p. 31; WDFW 2009). Prairie and meadow habitats used by pocket gophers have a naturally patchy distribution. In their prairie habitats, there is an even patchier distribution of soil rockiness which may further restrict the total area that pocket gophers can utilize (Steinberg and Heller 1997, p. 45; WDFW 2009). We assume that meadow soils have a similarly patchy distribution of rockiness, though the soil surveys to support this are, at this time, incomplete. In western Washington, pocket gophers currently occupy the following soils series: Alderwood, Cagey, Carstairs, Everett, Everett-Spanaway complex, Everett-Spanaway-Spana complex, Godfrey, Grove, Indianola, Kapowsin, McKenna, Murnen, Nisqually, Norma, Shelton, Spana, Spana-Spanaway-Nisqually complex, Spanaway, Spanaway-Nisqually complex, and Yelm. No soil survey information is currently available for occupied sites in the Olympic National Park, so the soils occupied there are unknown. We purposely avoid using specific map unit names, because we know that there are imperfections in soil mapping. Maps are based on the technology, standards, and tools available at the time soil surveys were conducted, sometimes up to 50 years ago. We recognize that soil survey boundaries may be adjusted in the future, and that soil series names may be added or removed to soil survey maps and databases. As a result, the overlap of pocket gopher locations with soil series names may be different in the future. The soils information presented here is based on best scientific data available at the time of listing. We also recognize that some of these soil series or soil series complexes are not typically either deep or well-drained. For a variety of reasons, mapped soil types may or may not have all of the characteristics described by the U.S. Department of Agriculture, Natural Resources Conservation Service, and the actual soils that occur on sites may have characteristics that make them more or less habitable by pocket gophers. These reasons may include: map boundary or transcription errors, map projection errors or differences, map identification or typing errors, soil or hydrological manipulations that have occurred since mapping took place, and small-scale inclusions that are different from the mapped soil. Because soils are mapped at large scales, mapped soils may not identify smaller inclusions. Any of the soil series or soil series complexes

listed above could potentially be suitable for the four Thurston/Pierce subspecies of the Mazama pocket gopher. And, the four Thurston/Pierce subspecies of the Mazama pocket gopher may also inhabit soil series not included in the above list. Although some soils are sandier, more gravelly, or may have more or less silt than described, most all soils used by pocket gophers are friable (easily pulverized or crumbled), loamy, and deep, and generally have slopes less than 15 percent. There have been reports of pocket gophers (subspecies unknown) occurring on other types of soils, on managed forest lands in Capitol State Forest (owned by the Washington State Department of Natural Resources, WDNR) and Vail Forest (owned by Weyerhaeuser) in Thurston County. These were subsequently determined to be moles (*Scapanus* spp.), based on trapping conducted in these areas by the Washington State Department of Fish and Wildlife (WDFW) during 2012 (Thompson, pers. comm. 2012b). A study of the relationship between soil rockiness and pocket gopher distribution revealed a strong negative correlation between the proportion of medium-sized rocks in the soil, and the presence of pocket gophers (eight of nine prairies sampled); medium sized rocks were considered greater than 0.5 inch (12.7 mm), but less than 2 inches (50.8 mm) in diameter (Steinberg 1996, p. 32). In observations of pocket gopher distribution on JBLM, pocket gophers did not occur in areas with a high percentage of Scot's broom cover (*Cytisus scoparius*), or where mole populations were particularly dense (Steinberg 1995, p. 26). A more recent study on JBLM also found that pocket gopher presence was negatively associated with Scot's broom; however, the researcher found no relationship between pocket gopher presence and mole density (Olson 2011a, pp. 12, 13). Pocket gopher burrows consist of a series of main runways, off which lateral tunnels lead to the surface of the ground (Wight 1918, p. 7). Pocket gophers dig their burrows using their sharp teeth and claws and then push the soil out through the lateral tunnels (Case and Jasch 1994, p. B-20; Wight 1918, p. 8). Nests containing dried vegetation are generally located near the center of each pocket gopher's home tunnel system (Wight 1918, p. 10). Food caches and store piles are usually placed near the nest, and excrement is piled into blind tunnels or loop tunnels, and then covered with dirt, leaving the nest and main runways clean (Wight 1918, p. 11). (USFWS, 2016).

***Dispersal/Migration*****Motility/Mobility**

Adult: High (USFWS, 2016)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory (USFWS, 2016)

**Dispersal**

Adult: Low (USFWS, 2016)

**Immigration/Emigration**

Adult: Unlikely (USFWS, 2016)

**Dispersal/Migration Narrative**

Adult: Pocket gophers have limited dispersal capabilities (Williams and Baker 1976, p. 303). Mazama pocket gophers are smaller in size than other sympatric or peripatric *Thomomys* species (Verts and Carraway 2000, p. 1). Both dispersal distance and home range size are therefore likely to be smaller than for other *Thomomys* species. Dispersal distances may vary

based on surface or soil conditions and size of the animal. For other, larger, *Thomomys* species, dispersal distances average about 131 feet (40 meters) (Barnes Jr. 1973, pp. 168, 169; Daly and Patton 1990, pp. 1286, 1288; Williams and Baker 1976, p. 306). Initial results from research being conducted on JBLM indicate that juvenile pocket gophers usually make movements from 13.1 to 32.8 feet (4-10 meters), though these may not be dispersal movements. One juvenile made a distinct dispersal movement of 525 feet (160 meters) in a single day (Olson 2012, p. 5). Suitable dispersal habitat is free of barriers to movement, and may need to contain foraging habitat if an animal is required to make a long-distance dispersal movement. Potential barriers include, but are not limited to, forest edges, roads (paved and unpaved), abrupt elevation changes, Scot's broom thickets (Olson 2012, p. 3), highly cultivated lawns, inhospitable soil types or substrates (Olson 2008, p. 4), development and buildings, slopes greater than 35 percent, and open water. Barriers may be permeable, meaning that they impede movement from place to place without completely blocking it, or they may be impermeable, meaning they cannot be crossed. Permeable barriers, as well as lower quality dispersal habitats, may present a risk of mortality for animals that use them (e.g., open areas where predation risk is increased, or a paved area where vehicular mortality is high). The WDFW conducted a study to determine dispersal distances of juvenile pocket gophers on JBLM. Twenty-eight juveniles were radio-collared and tracked for 17 to 56 days, with all but three animals tracked for more than 30 days. Of these, only nine gophers moved more than 32.8 feet (10 meters), and 10 gophers were never found more than 13.1 feet (4 meters) from any previous location (Olson 2012, p. 5). Only one animal dispersed what would be considered a larger distance, moving 525 feet (160 meters) in a single day.

### ***Population Information and Trends***

#### **Population Trends:**

Decreasing (USFWS, 2016)

#### **Population Narrative:**

There are few data on historical or current population sizes of *Mazama* pocket gopher (pocket gopher) populations in Washington, although several local populations and one subspecies are believed to be extinct. Knowledge of the past status of the pocket gopher is limited to distributional information. Recent surveys have focused on determining current distribution, primarily in response to development applications. In addition, in 2012, WDFW initiated a five county-wide distribution survey. Because the object of all of these surveys has mainly been presence/absence only, total population numbers for each subspecies are unknown. And, the precise boundaries of each subspecies' range are not currently known. Local population estimates have been reported but are based on using apparent gopher mounds to delineate the number of territories, a method that has not been validated (Stinson 2005, pp. 40, 41). Olson (2011a, p. 2) evaluated this methodology on pocket gopher populations at the Olympia Airport and Wolf Haven International. Although there was a positive relationship between the number of mounds and number of pocket gophers, the relationship varies spatially, temporally, and demographically (Olson 2011a, pp. 2, 39). Based on the results of Olson's 2011 study, we believe past population estimates (Stinson 2005) may have been too high. As there is no generally accepted standard survey protocol to determine population size for pocket gophers, it is not currently possible to obtain an estimate of subspecies population sizes or trends. Overall habitat availability has declined, however, and habitat has a finite ability to support pocket gophers. For these reasons, the Service concludes that the overall population trend of each of

the four Thurston/Pierce subspecies of the *Mazama* pocket gopher is negative. Increased survey effort since 2007 has resulted in the identification of numerous additional occupied sites located on private lands, especially in Thurston County (WDFW 2013a). Some of these new detections are adjacent to other known occupied sites, such as the population at the Olympia Airport. The full extent of these smaller discontinuous sites is currently unknown, and no research has been done to determine whether or not these aggregations are “stepping stone” sites that may facilitate dispersal into nearby unoccupied suitable habitat, or if they are population sinks (sites that do not add to the overall population through recruitment). Others of these additional occupied sites are separate locations, seemingly unassociated (physically) with known populations (Tirhi, in litt. 2008). The largest known expanse of areas occupied by any subspecies in Washington occur on JBLM (Roy Prairie and Yelm pocket gophers), and at the Olympia and Shelton airports (Olympia and Shelton pocket gophers, respectively). A translocated population occurs on Wolf Haven International’s land near Tenino, Washington. Between 2005 and 2008, over 200 gophers from a variety of areas in Thurston County (some from around Olympia Airport (Olympia pocket gopher, *T. m. pugetensis*)) and some from near the intersection of Rich Road and Yelm Highway (assumed to be Olympia pocket gophers) were released into the 38 acres (15 ha) mounded prairie site. Based on the best available information, we do not believe the property previously supported pocket gophers. Today pocket gophers continue to occupy the site (Tirhi, in litt. 2011); however, current population estimates are not available. Another site, West Rocky Prairie Wildlife Area, has received a total of 560 translocated pocket gophers (*T. m. pugetensis*) from the Olympia Airport between 2009 and 2011. Initial translocation efforts were unsuccessful; a majority of the pocket gophers died within three days due to predation (Olson 2009, p. 3). Modified release techniques used in 2010 and 2011 resulted in improved survival rates (Olson 2011b, p. 4). It is too soon to know if the population will become self-sustaining, or if additional translocations of gophers will be necessary.

### ***Threats and Stressors***

**Stressor:** Destruction, Modification, or Curtailment of Habitat and Range (USFWS, 2016)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** The primary long term threats to the pocket gopher are the loss, conversion, and degradation of habitat, particularly to urban development, successional changes to grassland habitat, and the spread of invasive plants. The threats also include increased predation pressure, which is closely linked to habitat degradation. The prairies of south Puget Sound are one of the rarest ecosystems in the United States (Dunn and Ewing 1997b, p. v; Noss et al. 1995, p. I-2). Dramatic changes have occurred on the landscape over the last 150 years, including a 90 to 95 percent reduction in the extent of the prairie ecosystem. In the south Puget Sound region, where most of western Washington’s prairies historically occurred, less than 10 percent of the original prairie persists, and only three percent remains dominated by native vegetation (Crawford and Hall 1997, pp. 13, 14). Development: Native prairies and grasslands have been severely reduced throughout the range of the four Thurston/Pierce subspecies of *Mazama* pocket gopher, especially as a result of conversion to residential and commercial development and agriculture. Prairie habitat continues to be lost, particularly to residential development (Stinson 2005, p. 70), by removal and fragmentation of native vegetation, and the excavation, and/or heavy equipment-caused compaction of surfaces and conversion to non-habitat (e.g., buildings, pavement, other infrastructure), rendering soils unsuitable for burrowing. Residential

development is associated with increased infrastructure, such as new road construction, which is one of the primary causes of landscape fragmentation (Watts et al. 2007, p. 736). Activities that accompany low-density development are correlated with decreased levels of biodiversity, mortality to wildlife, and facilitated introduction of nonnative invasive species (Trombulak and Frissell 2001; Watts et al. 2007, p. 736). In the south Puget Sound lowlands, the glacial outwash soils and gravels underlying the prairies are deep and valued for use in construction and road building, which leads to their degradation and destruction. In the south Puget Sound, Nisqually loamy soils appear to support high densities of pocket gophers (Stinson, in litt. 2010a Olson 2008, p. 6), the vast majority of which occur in developed areas of Thurston County, or within the Urban Growth Areas for the cities of Olympia, Tumwater, and Lacey (WDFW 2009), where future development is most likely to occur. Where pocket gopher populations presumably extended across an undeveloped expanse of open prairie (Dalquest and Scheffer 1942, pp. 95, 96), areas currently occupied by the four Thurston/Pierce subspecies of the Mazama pocket gopher are now isolated to small fragmented patches due to development and conversion of suitable habitat to incompatible uses. The presumed extinction of the Tacoma pocket gopher is likely linked directly to residential and commercial development, which has replaced nearly all pocket gopher habitats in the historical range of the subspecies (Stinson 2005, pp. 18, 34, 46). One of the historical Tacoma pocket gopher sites was converted to a large gravel pit and golf course (Steinberg 1996, pp. 24, 27; Stinson 2005, pp. 47, 120). In addition, two gravel pits are now operating on part of the site recognized as the type locality for the Roy Prairie pocket gopher (Stinson 2005, p. 42), and another is in operation near Tenino (Stinson, in litt. 2010b) in the vicinity of the type locality for the Tenino pocket gopher. Multiple pocket gopher sites in Pierce and Thurston Counties may be, or have been, lost to gravel pit development, golf course development, or residential and commercial development (Stinson, in litt. 2005; Stinson 2005, pp. 26, 42; Stinson, in litt. 2010b). Multiple prairies that used to contain uninterrupted expanses of prairie habitat suitable for pocket gophers within the range of the four Thurston/Pierce subspecies have been developed to cities, neighborhoods, agricultural lands, or military bases, and/or negatively impacted by such development, including Baker Prairie, Bush Prairie, Chambers Prairie, Frost Prairie, Grand Mound Prairie, Little Chambers Prairie, Marion Prairie, Roy Prairie, Ruth Prairie, Woods Prairie, Violet Prairie, and Yelm Prairie. Some of these prairie areas still contain smaller areas that support pocket gophers, and some appear to no longer support pocket gophers at all (WDFW 2012). Where their properties coincide with pocket gopher occupancy, many private lands developers and landowners in Thurston County have agreed to create set-asides or agree to other mitigation activities in order to obtain development permits from the County (Tirhi, in litt. 2008). However, it is unknown if any pocket gophers will remain on these sites due to the small size of the set-asides, extensive grading in some areas adjacent to set-asides, lack of dedicated funding for enforcement or monitoring of set-aside maintenance (Thurston County Long Range Planning and Resource Stewardship, in litt. 2011, p. 2), and lack of control of predation by domestic or feral cats and dogs. In addition, some landowners have received variances from Thurston County that allowed development to occur without a requirement to set aside areas for pocket gophers. A population of Olympia pocket gophers is located at and around the Port of Olympia's Olympia Airport, which is sited on the historical Bush Prairie. Gophers on Bush Prairie are currently vulnerable to negative impacts from proposed future development by the Port of Olympia and ongoing development by adjacent landowners. The Port of Olympia has plans to develop large portions of the existing grassland that likely supports the largest population of the Olympia pocket gopher in Washington (Stinson 2007, in litt.; Port of Olympia and WDFW 2008, p.1; Port of Olympia 2012). The Olympia Airport is realigning the airport runway, which is in known occupied habitat. They continue to work with

the Service and WDFW on mitigating airport expansion activities that may negatively impact gophers (Tirhi, in litt. 2010). The Olympia pocket gopher has a population at the Olympia Airport that spans several hundred acres, and there are two translocated populations: one at West Rocky Prairie Wildlife Area (some individuals from the Olympia Airport) and one at Wolf Haven (individuals from the Olympia Airport and some from near the intersection of Rich Road and Yelm Highway). The population centered on the Olympia Airport could be negatively impacted by plans for development both on and off the airport, while the two translocated populations are currently secure from intense commercial and residential development pressures as they occur on conserved lands. The Roy Prairie pocket gopher is known to occur across a large expanse of prairie on JBLM, which is currently secure from the threat of development. The Tenino pocket gopher has a single known population, which has been detected during surveys on the Rocky Prairie NAP, although the intermittent nature of these detections suggests it must be part of a larger metapopulation that occurs across nearby areas that have not been accessible for surveys. No known development poses a threat to the NAP, but any future conversion of the surrounding area to incompatible land use would likely hinder the recovery of this subspecies. The Yelm pocket gophers on Tenalquot prairie (which is owned in large part by JBLM) and Scatter Creek Wildlife Area are also secure from such residential and commercial development, but the Yelm pocket gopher habitat on Rock Prairie north of Old Highway 99 is in an area that is likely to be developed soon, which may negatively affect any local populations in the vicinity.

**Loss or Curtailment of Natural Disturbance Processes:** The suppression and loss of ecological disturbance regimes across vast portions of the landscape, such as fire, has resulted in altered vegetation structure in the prairies and meadows and has facilitated invasion by native and nonnative woody vegetation, rendering habitat unusable for the four Thurston/Pierce subspecies of *Mazama* pocket gopher. The basic ecological processes that maintain prairies and meadows have disappeared from, or have been altered on, all but a few protected and managed sites. Historically, the prairies and meadows of the south Puget Sound region are thought to have been actively maintained by native peoples, who lived here for at least 10,000 years before the arrival of Euro-American settlers (Boyd 1986; Christy and Alverson 2011, p. 93). Frequent burning reduced the encroachment and spread of shrubs and trees (Boyd 1986; Chappell and Kagan 2001, p. 42), favoring open grasslands with a variety of native plants and animals. Following Euro-American settlement of the region in the mid-19th century, fire was actively suppressed on grasslands, allowing encroachment by woody vegetation into the remaining prairie habitat and oak woodlands (Agee 1993, p. 360; Altman et al. 2001, p. 262; Boyd 1986; Franklin and Dyrness 1973, p. 122; Kruckeberg 1991, p. 287). Fires on the prairie create a mosaic of vegetation conditions, which serve to maintain native prairie plant communities. In some prairie patches fires will kill encroaching woody vegetation and reset succession back to bare ground, creating early successional vegetation conditions suitable for many native prairie species. Early succession forbs and grasses are favored by pocket gophers. The historical fire frequency on prairies has been estimated to be 3 to 5 years (Foster 2005, p. 8). On sites where regular fires occur, there is a high complement of native plants and fewer invasive species. These types of fires maintain the native short-statured plant communities favored by pocket gophers. The result of fire suppression has been the invasion of the prairies and oak woodlands by native and nonnative plant species (Dunn and Ewing 1997a, p. v; Tveten and Fonda 1999, p. 146), notably woody plants such as the native Douglas-fir (*Pseudotsuga menziesii*) and the nonnative Scot's broom. On tallgrass prairies in midwestern North America, fire suppression has led to degradation and the loss of native grasslands (Curtis 1959, pp. 296, 298; Panzer 2002, p. 1297). On northwestern prairies, fire suppression has allowed Douglas-fir to encroach on and outcompete native prairie vegetation for light, water, and nutrients (Stinson 2005, p. 7). This

increase in woody vegetation and nonnative plant species has resulted in less available prairie habitat overall and habitat that is unsuitable for and avoided by many native prairie species, including pocket gophers (Olson 2011a, pp. 12, 16; Pearson et al. 2005, pp. 2, 27; Tveten and Fonda 1999, p. 155). Pocket gophers prefer early successional vegetation as forage. Woody plants shade out the forbs and grasses that pocket gophers prefer to eat, and high densities of woody plants make travel both below and above the ground difficult. In locations with poor forage, pocket gophers tend to have larger territories, which may be difficult or impossible to establish in densely forested areas. The probability of pocket gopher occupancy is much higher in areas with less than 10 percent woody vegetation cover (Olson 2011a, p. 16). On JBLM alone, over 16,000 acres (6,477 ha) of prairie has converted to Douglas-fir forest since the mid-19th century (Foster and Shaff 2003, p. 284). Where controlled burns or direct tree removal are not used as a management tool, this encroachment will continue to cause the loss of open grassland habitats for pocket gophers and is an ongoing threat to the species. Restoration in some of the south Puget Sound grasslands has resulted in temporary control of Scot's broom and other invasive plants through the careful and judicious use of herbicides, mowing, grazing, and fire. Fire has been used as a management tool to maintain native prairie composition and structure and is generally acknowledged to improve the health and composition of grassland habitat by providing a short-term nitrogen addition, which results in a fertilizer effect to vegetation, thus aiding grasses and forbs to sprout. Unintentional fires ignited by military training burn patches of prairie grasses and forbs on JBLM on an annual basis. These light ground fires create a mosaic of conditions within the grassland, maintaining a low vegetative structure of native and nonnative plant composition, and patches of bare soil. Because of the topography of the landscape, fires create a patchy mosaic of areas that burn completely, some areas that do not burn, and areas where consumption of the vegetation is mixed in its effects to the habitat. One of the benefits of fire in grasslands is that it tends to kill regenerating conifers, and reduces the cover of nonnative shrubs such as Scot's broom, although Scot's broom seed stored in the soil can be stimulated by fire (Agee 1993, p. 367). Fire also improves conditions for many native bulb-forming plants, such as *Camassia* spp. (Agee and Dunwiddie 1984). On sites where regular fires occur, such as on JBLM, there is a high complement of native plants and fewer invasive species. These types of fires maintain the native, short-statured plant communities favored by pocket gophers. Management practices such as intentional burning and mowing require expertise in timing and technique to achieve desired results. If applied at the wrong season, frequency, or scale, fire and mowing can be detrimental to the restoration of native prairie species. Excessive and high-intensity burning can result in a lack of vegetation or encourage regrowth of nonnative grasses. Where such burning has occurred over a period of more than 50 years on the artillery ranges of JBLM, prairies are covered by nonnative forbs and grasses instead of native perennial bunchgrasses (Tveten and Fonda 1999, pp. 154, 155). Pocket gophers are not commonly found in areas colonized by Douglas-fir trees because pocket gophers require forbs and grasses of an early successional stage for food (Witmer et al. 1996a, p. 96). Pocket gophers observed on JBLM did not occur in areas with high cover of Scot's broom (Steinberg 1995, p. 26). A more recent study on JBLM also found that pocket gopher presence was negatively associated with Scot's broom (Olson 2011a, pp. 12, 13, 16). Some subspecies may disperse through forested areas or may temporarily establish territories on forest edges, but there is currently not enough data available to determine how common this behavior may be or which subspecies employ it. The four Thurston/Pierce subspecies of the *Mazama* pocket gopher occur on prairie-type habitats, many of which, if not actively managed to maintain vegetation in an early-successional state, have been invaded by shrubs and trees that either preclude pocket gophers or limit their ability to fully occupy the landscape. Typical management at civilian airports prevents woody

vegetation from encroaching onto surrounding areas for flight safety reasons. Woody vegetation encroachment is therefore not a threat at civilian airports. Military Training: Pocket gopher populations occurring on JBLM are exposed to differing levels of training activities on the base. The Department of Defense's (DOD) proposed actions under their "Grow the Army" initiative include stationing 5,700 new soldiers, new combat service support units, a combat aviation brigade, facility demolition and construction to support the increased troop levels, and additional aviation, maneuver, and live fire training (75 FR 55313, September 10, 2010). The increased training activities will affect nearly all training areas at JBLM, resulting in an increased risk of accidental fires, and habitat destruction and degradation attributable to vehicle use in occupied areas, mounted and dismounted training, bivouac activities, and digging. Even though the training areas on the base are degraded, with implementation of agreed-upon conservation measures, these areas still provide habitat for the Roy Prairie and Yelm pocket gopher. JBLM's recently signed Endangered Species Management Plan (ESMP) for the Mazama pocket gopher will serve to minimize threats across the base by redirecting some training activities to areas outside of occupied habitat, designating areas where no vehicles are permitted, designating areas where vehicles will remain on roads only, and designating areas where no digging is allowed, among other conservation measures. JBLM has further committed to enhancing and expanding suitable habitat for the Roy Prairie and Yelm pocket gophers in "priority habitat" areas on base (areas that were proposed as critical habitat); enforcing restrictions on recreational use of occupied habitat by dog owners and horseback riders; and continuing to support the off-base recovery of the four Thurston/Pierce subspecies of the Mazama pocket gopher. Several moderate- to large-sized areas supporting pocket gophers have been identified on JBLM. These areas are within the historical ranges of the Roy Prairie (Pierce County) and Yelm (Thurston County) pocket gophers. Their absence from some sites of what is presumed to have been formerly suitable habitat may be related to compaction of the soil due to years of mechanized vehicle training (Steinberg 1995, p. 36). Training infrastructure (e.g., roads, firing ranges, bunkers) also degrades pocket gopher habitat and may lead to reduced use of these areas by pocket gophers. For example, JBLM has plans to add a third rifle range on the south impact area where it overlaps with a densely occupied pocket gopher site. The area may be usable by pocket gophers when the project is completed; however, construction of the rifle range may result in removal of forage and direct mortality of pocket gophers through crushing of burrows (Stinson, in litt. 2011). Recent survey access to the center of the artillery impact area on 91st Division Prairie, where bombardment is presumably of the highest intensity, did detect some unspecified level of occupancy by the Roy Prairie pocket gopher (WDFW 2013b, enclosure 1, p. 6). This apparently suitable central portion of the 91st Division Prairie is subject to repeated and ongoing bombardment, which may create an ecological trap for dispersing juveniles. JBLM training areas have varying levels of use; some allow excavation and off-road vehicle use, while other areas have restrictions that limit off-road vehicle use. The ESMP specifically requires coordination between the JBLM Fish and Wildlife personnel and the JBLM entities responsible for training activities (e.g., Range Support, battalion commanders, and/or first field grade officers) to ensure all parties are aware of where occupied areas occur in relation to training activities, the effects of training, and the potential ramifications of habitat destruction or animal mortality. Since military training has the potential to directly or indirectly harm or harass pocket gophers, we conclude that these activities will negatively impact the Roy Prairie and Yelm pocket gophers. JBLM has committed to operational restrictions on portions of the base in order to avoid and minimize potential impacts to Roy Prairie and Yelm pocket gophers. Currently-occupied areas will be buffered from training activities, with an emphasis on occupied habitat in "priority habitat" areas. Regular surveys will be conducted with the goals of determining distribution, protecting

pocket gophers and their habitat from disturbance or destruction, and determining population status. Where possible, JBLM will alleviate training pressure by transferring activities to unoccupied areas where encroaching forest has been removed. This strategy has the effect of both releasing large areas of land that were historically prairie and providing unoccupied areas where training is free of the risk of negatively impacting Roy Prairie or Yelm pocket gophers. While the Service fully supports the implementation of these impact minimization efforts and will continue to collaborate with DOD to address all aspects of training impacts on the species, not all adverse impacts on pocket gophers can be fully avoided. Military training continues to pose a threat to the Roy Prairie and Yelm subspecies at this time. No military training occurs in the ranges of the Olympia or Tenino subspecies of the Mazama pocket gopher (USFWS, 2016).

**Stressor:** Poor Connectivity Between Small and Isolated Populations (USFWS, 2016)

**Exposure:**

**Response:**

**Consequence:** Isolated genetics

**Narrative:** Most species' populations fluctuate naturally, responding to various factors such as weather events, disease, and predation. Populations that are small, fragmented, or isolated by habitat loss or modification of naturally patchy habitat, and other human-related factors, are more vulnerable to extirpation by natural randomly occurring events, cumulative effects, and to genetic effect (collectively known as small population effects). These effects can include genetic drift (loss of recessive alleles), founder effects (over time, an increasing percentage of the population inheriting a narrow range of traits), and genetic bottlenecks leading to increasingly lower genetic diversity, with consequent negative effects on evolutionary potential. To date, of the eight subspecies of Mazama pocket gopher in Washington, only the Olympic pocket gopher has been documented as having low genetic diversity (Welch and Kenagy 2008, p. 7), although the six other extant subspecies have local populations that are small, fragmented, and physically isolated from one another. The four Thurston/Pierce subspecies of the Mazama pocket gopher face threats from loss or fragmentation of habitat. Historically, pocket gophers probably persisted by continually recolonizing habitat patches after local extinctions. However, widespread development and conversion of habitat has resulted in widely separated populations, and intervening habitat corridors are now gone, with the effect of impeding or stopping much of the natural recolonization that historically occurred (Stinson 2005, p. 46). Although pocket gophers are not known to have low genetic diversity, small population sizes at most sites, coupled with disjunct and fragmented habitat, may contribute to further population declines. Little is known about the local or rangewide reproductive success of pocket gophers found in Washington State (USFWS, 2016).

**Stressor:** Predation and Pest Control (USFWS, 2016)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** Predation: Predation influences the distribution, abundance, and diversity of species in ecological communities. Generally, predation leads to changes in both the population size of the predator and that of the prey. In unfavorable environments, prey species are stressed or living at low population densities such that predation is likely to have negative effects on all prey species, thus lowering species richness. In addition, when a nonnative predator is introduced to the ecosystem, negative effects on the prey population may be higher than those from co-evolved native predators. The effect of predation may be magnified when populations are small,

and the disproportionate effect of predation on declining populations has been shown to drive rare species even further towards extinction (Woodworth 1999, pp. 74, 75). Predation has an impact on populations of the four Thurston/Pierce subspecies of *Mazama* pocket gopher. Urbanization, particularly in the south Puget Sound region, has resulted in not only habitat loss, but also increased exposure to feral and domestic cats and dogs. Domestic cats are known to have serious impacts on small mammals and birds and have been implicated in the decline of several endangered and threatened mammals, including marsh rabbits (*Sylvilagus palustris*) in Florida and the salt-marsh harvest mouse (*Reithrodontomys raviventris*) in California (Ogan and Jurek 1997, p. 89). Domestic cats and dogs have been specifically identified as common predators of pocket gophers (Case and Jasch 1994, p. B-21; Henderson 1981, p. 233; Wight 1918, p. 21) and at least two pocket gopher locations were found as a result of house cats bringing home pocket gopher carcasses (WDFW 2001). Informal interviews with area biologists document multiple incidents of domestic pet predation on pocket gophers (Chan, in litt. 2013; Clouse, in litt. 2012 Skriletz 2013 in litt., Wood 2013 in litt.). There is also one recorded instance of a WDFW biologist being presented with a dead *Mazama* pocket gopher by a dog during an east Olympia, Washington, site visit in 2006 (Burke Museum 2012 McAllister 2013 in litt.). Some local populations of the pocket gopher occur in areas where people recreate with their dogs, bringing these potential predators into environments that may otherwise be relatively free of them, consequently increasing the risks to individual pocket gophers and populations that may be small and isolated. The four Thurston/Pierce subspecies of *Mazama* pocket gopher occur in rapidly developing areas. Local populations that survive commercial and residential development (adjacent to and within habitat) are potentially vulnerable to extirpation by domestic and feral cats and dogs (Case and Jasch 1994, p. B-21; Henderson 1981, p. 233). As stated previously, predation is a natural part of the pocket gopher's life history; however, the effect of predation may be magnified when populations are small and habitat is fragmented. The disproportionate effect of additional predation on declining populations has been shown to drive rare species even further towards extinction (Woodworth 1999, pp. 74, 75). Predation, particularly from nonnative species, will likely continue to be a threat to the four Thurston/Pierce subspecies of the *Mazama* pocket gopher now and in the future. This is particularly likely where development abuts gopher habitat, resulting in increased numbers of cats and dogs in the vicinity, and in areas where people recreate with their dogs – particularly if dogs are off-leash and not prevented from harassing wildlife. In such areas, where local populations of pocket gophers are already small, this additional predation pressure (above natural levels of predation) is expected to further negatively impact population numbers. Pest Control: Pocket gophers are often considered a pest because they sometimes damage crops and seedling trees, and their mounds can create a nuisance. Several site locations were found as a result of trapping conducted on Christmas tree farms, a nursery, and in a livestock pasture (WDFW 2001). The type locality for the Cathlamet pocket gopher is on a commercial tree farm. Pocket gophers from Thurston County were used in a rodenticide experiment as recently as 1995 (Witmer et al. 1996a, p. 97). In Washington State it is currently illegal to trap or poison *Mazama* pocket gophers, or to trap or poison moles where they overlap with *Mazama* pocket gopher populations, but not all property owners are cognizant of these laws, nor are most citizens capable of differentiating between moles, pocket gophers, or the signs of their habitation (e.g., soil disturbance). In light of this, it is reasonable to believe that mole trapping or poisoning still has the potential to adversely affect pocket gopher populations. Local populations that survive commercial and residential development (adjacent to and within habitat) may be subsequently extirpated by trapping or poisoning. Lethal control by trapping or poisoning is most likely to be a threat to the four Thurston/Pierce subspecies where their ranges overlap residential properties (USFWS, 2016).

***Recovery*****References**

USFWS 2016. Status of the Species: (Thomomys mazama ssp.) Mazama Pocket Gopher. U.S. Fish and Wildlife Service 2600 SE 98TH Ave., Suite 100. Portland, OR 97266. Provided to FESTF from Chris Mullens 9/30/2016

NatureServe. 2015. NatureServe Central Databases. Arlington, Virginia, U.S.A.

USFWS. 2016. Status of the Species: (Thomomys mazama ssp.) Mazama Pocket Gopher. U.S. Fish and Wildlife Service 2600 SE 98TH Ave., Suite 100. Portland, OR 97266. Provided to FESTF from Chris Mullens 9/30/2016

USFWS 2016. Status of the Species: (Thomomys mazama ssp.) Mazama Pocket Gopher. U.S. Fish and Wildlife Service 2600 SE 98TH Ave., Suite 100. Portland, OR 97266. Provided to FESTF from Chris Mullens 9/30/2016.

USFWS. 2016. Status of the Species: (Thomomys mazama ssp.) Mazama Pocket Gopher. U.S. Fish and Wildlife Service 2600 SE 98TH Ave., Suite 100. Portland, OR 97266. Provided to FESTF from Chris Mullens 9/30/2016.

## SPECIES ACCOUNT: *Thomomys mazama pugetensis* (Olympia pocket gopher)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Threatened; Pacific Region (R1) (USFWS, 2016)

### **Physical Description**

Adult Mazama pocket gophers are reddish brown to black above, and the underparts are lead-colored with buff-colored tips. The lips, nose, and patches behind the ears are black; the wrists are white. Adults range from 7 to 9 inches (189 to 220 millimeters (mm)) in total length, with tails that range from 2 to 3 inches (45 to 85 mm)(Verts and Carraway 2000, p.2). Mazama pocket gophers are morphologically similar to other species of pocket gophers that exploit a subterranean existence. They are stocky and tubular in shape, with short necks, powerful limbs, long claws, and tiny ears and eyes. Their short, nearly hairless tails are highly sensitive and probably assist when navigating tunnels. The “pockets” are external, fur-lined cheek pouches on either side of the mouth that are used to transport nesting material and plant cuttings. Mazama pocket gophers reach reproductive age in the spring of the year after their birth and produce litters between spring and early summer. Litter size ranges from one to nine (Wight 1918, p. 14), with an average of five (Scheffer 1938, p. 222). They do not hibernate in winter; they remain active throughout the year (Case and Jasch 1994, p. B-20) (USFWS, 2016).

### **Taxonomy**

Although the species *Thomomys mazama*, or Mazama pocket gopher, includes numerous subspecies that are found in the States of Washington, Oregon, and California, only the subspecies found in the State of Washington have recently been considered for listing. The Mazama pocket gopher complex consists of 15 subspecies, eight of which occur only in Washington, five of which occur only in Oregon, one that occurs only in California, and one subspecies with a distribution that spans the boundary between Oregon and California (Hall 1981, p. 467). The first pocket gophers collected in western Washington were considered subspecies of the northern pocket gopher (*Thomomys talpoides*)(Goldman 1939), until 1960 when the complex of pocket gophers found in western Washington was determined to be more similar to the western pocket gopher (*T. mazama*)(Johnson and Benson 1960, p. 20). Eight western Washington subspecies of Mazama pocket gopher (*T. mazama*, ssp. *couchi*, *glacialis*, *louiei*, *melanops*, *pugetensis*, *tacomensis*, *tumuli*, and *yelmensis*) have been identified (Hall 1981, p. 467). *Thomomys mazama* is recognized as a valid species by the Integrated Taxonomic Information System (ITIS 2012). Although there have been suggestions that potential changes to the classification of some of these subspecies should be considered, we have no information to suggest that any of the presently recognized subspecies are the subject of serious dispute. We follow the subspecies designations of Verts and Carraway (2000), as this text represents the currently accepted taxonomy for the species *T. mazama*. Verts and Carraway (2000, p.1) recognize *T. m. glacialis*, *pugetensis*, *tumuli*, and *yelmensis* as separate subspecies (the Roy Prairie, Olympia, Tenino, and Yelm pocket gophers, respectively) based on morphological characteristics, distribution, and differences in number of chromosomes. Due to the close proximity of the four subspecies located in Thurston and Pierce Counties, and the fact that at least three of them occur in the same clade, we refer to these four subspecies (*T. m. glacialis*,

pugetensis, tumuli, and yelmensis) as “the four Thurston/Pierce subspecies” of the Mazama pocket gopher (USFWS, 2016).

**Current Range**

In Washington, Mazama pocket gophers are found west of the Cascade Mountain Range, in the Olympic Mountains and in the Puget Sound trough, with an additional single locality known from Wahkiakum County (Verts and Carraway 2000, p.3). Their populations are concentrated in well-drained friable soils often associated with glacial outwash. The type locality for the Olympia pocket gopher (*Thomomys mazama pugetensis*) was the prairie on and around the Olympia Airport (Dalquest and Scheffer 1944, p. 445). Gophers continue to occupy this area. Soil series and soil series complexes in and around this area that may support pocket gophers include Alderwood, Cagey, Everett, Indianola, McKenna, Nisqually, Norma, Spana, Spanaway- Nisqually complex, and Yelm (USFWS, 2016).

**Critical Habitat Designated**

Yes; 4/9/2014.

**Legal Description**

On April 9, 2014, the U.S. Fish and Wildlife Service (Service) designated critical habitat for three subspecies of the Mazama pocket gopher (the Olympia pocket gopher, *Thomomys mazama pugetensis*; the Tenino pocket gopher, *T. m. tumuli*; and the Yelm pocket gopher, *T. m. yelmensis*) under the Endangered Species Act of 1973, as amended (Act). In total, approximately 1,607 acres (650 hectares) in Thurston County, Washington, fall within the boundaries of the critical habitat designation for the Olympia, Tenino, and Yelm pocket gophers. The effect of this regulation was to designate critical habitat for the Olympia, Tenino, and Yelm subspecies of the Mazama pocket gopher found in Thurston County, Washington, under the Act.

**Critical Habitat Designation**

Olympia Pocket Gopher Critical Habitat—Olympia Airport Unit. This unit consists of 676 ac (274 ha) and is made up of land owned by the Port of Olympia, a municipal corporation. The Olympia Airport Unit is located south of the cities of Olympia and Tumwater, in Thurston County, Washington. This unit is occupied by the Olympia pocket gopher and contains the physical or biological features essential to the conservation of the subspecies due to the underlying soil series (Cagey, Everett, Indianola, and Nisqually), suitable forb and grass vegetation present onsite, and its large size

**Primary Constituent Elements/Physical or Biological Features**

Critical habitat for the Olympia pocket gopher is designated in Thurston County, Washington. Within this area, the primary constituent elements of the physical or biological features essential to the conservation of the Olympia pocket gopher consist of two components:

(i) Friable, loamy, and deep soils, some with relatively greater content of sand, gravel, or silt, all generally on slopes less than 15 percent in the following soil series or soil series complex: (A) Alderwood; (B) Cagey; (C) Everett; (D) Godfrey; (E) Indianola; (F) Kapowsin; (G) McKenna; (H) Nisqually; (I) Norma; (J) Spana; (K) Spanaway; (L) Spanaway-Nisqually complex; and (M) Yelm.

(ii) Areas equal to or larger than 50 ac (20 ha) in size that provide for breeding, foraging, and dispersal activities, found in the soil series listed in paragraph (2)(i) of this entry that have: (A)

Less than 10 percent woody vegetation cover; (B) Vegetative cover suitable for foraging by gophers. Pocket gophers' diets include a wide variety of plant material, including leafy vegetation, succulent roots, shoots, tubers, and grasses. Forbs and grasses that Mazama pocket gophers eat are known to include, but are not limited to: *Achillea millefolium* (common yarrow), *Agoseris* spp. (agoseris), *Cirsium* spp. (thistle), *Bromus* spp. (brome), *Camassia* spp. (camas), *Collomia linearis* (tiny trumpet), *Epilobium* spp. (several willowherb spp.), *Eriophyllum lanatum* (woolly sunflower), *Gayophytum diffusum* (groundsmoke), *Hypochaeris radicata* (hairy cat's ear), *Lathyrus* spp. (peavine), *Lupinus* spp. (lupine), *Microsteris gracilis* (slender phlox), *Penstemon* spp. (penstemon), *Perideridia gairdneri* (Gairdner's yampah), *Phacelia heterophylla* (varileaf phacelia), *Polygonum douglasii* (knotweed), *Potentilla* spp. (cinquefoil), *Pteridium aquilinum* (bracken fern), *Taraxacum officinale* (common dandelion), *Trifolium* spp. (clover), and *Viola* spp. (violet); and (C) Few, if any, barriers to dispersal. Barriers to dispersal may include, but are not limited to, forest edges, roads (paved and unpaved), abrupt elevation changes, Scot's broom thickets, highly cultivated lawns, inhospitable soil types or substrates, development and buildings, slopes greater than 35 percent, and open water.

### **Special Management Considerations or Protections**

Critical habitat does not include manmade structures (such as buildings, aqueducts, runways, roads, railroad tracks, and other paved areas) and the land on which they are located existing within the legal boundaries on May 9, 2014.

The physical or biological features essential to the conservation of each subspecies may require special management considerations or protection to restore, protect, and maintain the essential features found there. The physical or biological features in this subunit are threatened by: Loss of habitat through conversion to incompatible uses, such as development; predation; and the habitat degradation or destruction due to the inadequacy of existing regulatory mechanisms.

The physical or biological features essential to the conservation of the four Thurston/Pierce subspecies of the Mazama pocket gopher may require special management considerations or protection to control or prevent the establishment of invasive woody plants, which create shade and compete for light, food and nutrients otherwise utilized by the forb, bulb, and grass species that the gophers require for forage. Management may be implemented using hand tools or mechanical methods, prescribed fire, and the judicious use of herbicides. Although several management techniques are being implemented on public lands, we may need to improve our outreach to educate private landowners on controlling their pets and appropriately managing grazing on their properties, as well as to developing incentives for landowners who agree to conserve habitat. Incentives would create protected areas, through agreements or acquisitions. These would include corridors between existing protected habitat areas that may require management, enhancement actions, and long-term maintenance.

### ***Life History***

#### **Feeding Narrative**

Adult: Pocket gophers are generalist herbivores and their diet includes a wide variety of plant material, including leafy vegetation, succulent roots, shoots, and tubers. In natural settings pocket gophers play a key ecological role by aerating soils, activating the seed bank, and stimulating plant growth, though they can be considered pests in agricultural systems. In prairie and meadow ecosystems, pocket gopher activity plays an important role in maintaining species

richness and diversity. Foraging primarily takes place below the surface of the soil, where pocket gophers snip off roots of plants before occasionally pulling the whole plant below ground to eat or store in caches. If above-ground foraging occurs, it's usually within a few feet of an opening and forage plants are quickly cut into small pieces and carried back to the nest or cache (Wight 1918, p. 12). Any water they need is obtained from their food (Gettinger 1984, pp. 749-750; Wight 1918, p. 13). The probability of pocket gopher occupancy is much higher in areas with less than 10 percent woody vegetation cover (Olson 2011a, p. 16), presumably because such vegetation will shade out the forbs, bulbs, and grasses that pocket gophers prefer to eat, and high densities of woody plants make travel both below and above the ground difficult (USFWS, 2016).

### **Reproduction Narrative**

Adult: Pocket gophers reach sexual maturity during the spring of the year following their birth, and generally produce one litter per year (Case and Jasch 1994, p. B-20), though timing of sexual maturity has been shown to vary with habitat quality (Patton and Brylski 1987, p. 502; Patton and Smith 1990, p. 76). Gestation lasts approximately 18 days (Andersen 1978, p. 421; Schramm 1961, p. 169). Young are born in the spring to early summer (Wight 1918, p. 13), and are reared by the female. Aside from the breeding season, males and females remain segregated in their own tunnel systems. There are 1-9 pups per litter (averaging 5), born without hair, pockets, or teeth, and they must be kept warm by the mother or "packed" in dried vegetation (Case and Jasch 1994, p. B-20; Wight 1918, p. 14). Juvenile pelage starts growing in at just over a week (Andersen 1978, p. 420). The young eat vegetation in the nest within three weeks of birth, with eyes and ears opening and pockets developing at about a month (Andersen 1978, p. 420; Wight 1918, p. 14). At six weeks they are weaned, fighting with siblings, and nearly ready to disperse (Andersen 1978, p. 420; Wight 1918, p. 15), which usually occurs at about two months of age (Stinson 2005, p. 26). They attain their adult weight between four and five months of age (Andersen 1978, pp. 419, 421). Most pocket gophers live only a year or two, with few living to three or four years of age (Hansen 1962, pp. 152, 153; Livezey and Verts 1979, p. 39) (USFWS, 2016).

### **Spatial Arrangements of the Population**

Adult: Clumped (USFWS, 2016)

### **Habitat Narrative**

Adult: The Mazama pocket gopher (pocket gopher) is associated with glacial outwash prairies in western Washington, an ecosystem of conservation concern (Hartway and Steinberg 1997, p. 1), as well as alpine and subalpine meadows and other meadow-like openings at lower elevations. Steinberg and Heller (1997, p. 46) found that pocket gophers are even more patchily distributed than are prairies, as there are some seemingly high quality prairies within the species' range that lack pocket gophers; e.g., Mima Mounds Natural Area Preserve (NAP), and 13th Division Prairie on Joint Base Lewis-McChord (JBLM). Pocket gopher distribution is affected by the rock content of soils, drainage, forage availability, and climate (Case and Jasch 1994, p. B-21; Hafner et al. 1998, p. 279; Reichman 2007, pp. 273-274; Steinberg and Heller 1997, p. 45; Stinson 2005, p. 31; WDFW 2009). Prairie and meadow habitats used by pocket gophers have a naturally patchy distribution. In their prairie habitats, there is an even patchier distribution of soil rockiness which may further restrict the total area that pocket gophers can utilize (Steinberg and Heller 1997, p. 45; WDFW 2009). We assume that meadow soils have a similarly patchy distribution of rockiness, though the soil surveys to support this are, at this time, incomplete. In

western Washington, pocket gophers currently occupy the following soils series: Alderwood, Cagey, Carstairs, Everett, Everett-Spanaway complex, Everett-Spanaway-Spana complex, Godfrey, Grove, Indianola, Kapowsin, McKenna, Murnen, Nisqually, Norma, Shelton, Spana, Spana-Spanaway-Nisqually complex, Spanaway, Spanaway-Nisqually complex, and Yelm. No soil survey information is currently available for occupied sites in the Olympic National Park, so the soils occupied there are unknown. We purposely avoid using specific map unit names, because we know that there are imperfections in soil mapping. Maps are based on the technology, standards, and tools available at the time soil surveys were conducted, sometimes up to 50 years ago. We recognize that soil survey boundaries may be adjusted in the future, and that soil series names may be added or removed to soil survey maps and databases. As a result, the overlap of pocket gopher locations with soil series names may be different in the future. The soils information presented here is based on best scientific data available at the time of listing. We also recognize that some of these soil series or soil series complexes are not typically either deep or well-drained. For a variety of reasons, mapped soil types may or may not have all of the characteristics described by the U.S. Department of Agriculture, Natural Resources Conservation Service, and the actual soils that occur on sites may have characteristics that make them more or less habitable by pocket gophers. These reasons may include: map boundary or transcription errors, map projection errors or differences, map identification or typing errors, soil or hydrological manipulations that have occurred since mapping took place, and small-scale inclusions that are different from the mapped soil. Because soils are mapped at large scales, mapped soils may not identify smaller inclusions. Any of the soil series or soil series complexes listed above could potentially be suitable for the four Thurston/Pierce subspecies of the Mazama pocket gopher. And, the four Thurston/Pierce subspecies of the Mazama pocket gopher may also inhabit soil series not included in the above list. Although some soils are sandier, more gravelly, or may have more or less silt than described, most all soils used by pocket gophers are friable (easily pulverized or crumbled), loamy, and deep, and generally have slopes less than 15 percent. There have been reports of pocket gophers (subspecies unknown) occurring on other types of soils, on managed forest lands in Capitol State Forest (owned by the Washington State Department of Natural Resources, WDNR) and Vail Forest (owned by Weyerhaeuser) in Thurston County. These were subsequently determined to be moles (*Scapanus* spp.), based on trapping conducted in these areas by the Washington State Department of Fish and Wildlife (WDFW) during 2012 (Thompson, pers. comm. 2012b). A study of the relationship between soil rockiness and pocket gopher distribution revealed a strong negative correlation between the proportion of medium-sized rocks in the soil, and the presence of pocket gophers (eight of nine prairies sampled); medium sized rocks were considered greater than 0.5 inch (12.7 mm), but less than 2 inches (50.8 mm) in diameter (Steinberg 1996, p. 32). In observations of pocket gopher distribution on JBLM, pocket gophers did not occur in areas with a high percentage of Scot's broom cover (*Cytisus scoparius*), or where mole populations were particularly dense (Steinberg 1995, p. 26). A more recent study on JBLM also found that pocket gopher presence was negatively associated with Scot's broom; however, the researcher found no relationship between pocket gopher presence and mole density (Olson 2011a, pp. 12, 13). Pocket gopher burrows consist of a series of main runways, off which lateral tunnels lead to the surface of the ground (Wight 1918, p. 7). Pocket gophers dig their burrows using their sharp teeth and claws and then push the soil out through the lateral tunnels (Case and Jasch 1994, p. B-20; Wight 1918, p. 8). Nests containing dried vegetation are generally located near the center of each pocket gopher's home tunnel system (Wight 1918, p. 10). Food caches and store piles are usually placed near the nest, and

excrement is piled into blind tunnels or loop tunnels, and then covered with dirt, leaving the nest and main runways clean (Wight 1918, p. 11). (USFWS, 2016).

### ***Dispersal/Migration***

#### **Motility/Mobility**

Adult: High (USFWS, 2016)

#### **Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory (USFWS, 2016)

#### **Dispersal**

Adult: Low (USFWS, 2016)

#### **Immigration/Emigration**

Adult: Unlikely (USFWS, 2016)

### **Dispersal/Migration Narrative**

Adult: Pocket gophers have limited dispersal capabilities (Williams and Baker 1976, p. 303). Mazama pocket gophers are smaller in size than other sympatric or peripatric *Thomomys* species (Verts and Carraway 2000, p. 1). Both dispersal distance and home range size are therefore likely to be smaller than for other *Thomomys* species. Dispersal distances may vary based on surface or soil conditions and size of the animal. For other, larger, *Thomomys* species, dispersal distances average about 131 feet (40 meters) (Barnes Jr. 1973, pp. 168, 169; Daly and Patton 1990, pp. 1286, 1288; Williams and Baker 1976, p. 306). Initial results from research being conducted on JBLM indicate that juvenile pocket gophers usually make movements from 13.1 to 32.8 feet (4-10 meters), though these may not be dispersal movements. One juvenile made a distinct dispersal movement of 525 feet (160 meters) in a single day (Olson 2012, p. 5). Suitable dispersal habitat is free of barriers to movement, and may need to contain foraging habitat if an animal is required to make a long-distance dispersal movement. Potential barriers include, but are not limited to, forest edges, roads (paved and unpaved), abrupt elevation changes, Scot's broom thickets (Olson 2012, p. 3), highly cultivated lawns, inhospitable soil types or substrates (Olson 2008, p. 4), development and buildings, slopes greater than 35 percent, and open water. Barriers may be permeable, meaning that they impede movement from place to place without completely blocking it, or they may be impermeable, meaning they cannot be crossed. Permeable barriers, as well as lower quality dispersal habitats, may present a risk of mortality for animals that use them (e.g., open areas where predation risk is increased, or a paved area where vehicular mortality is high). The WDFW conducted a study to determine dispersal distances of juvenile pocket gophers on JBLM. Twenty-eight juveniles were radio-collared and tracked for 17 to 56 days, with all but three animals tracked for more than 30 days. Of these, only nine gophers moved more than 32.8 feet (10 meters), and 10 gophers were never found more than 13.1 feet (4 meters) from any previous location (Olson 2012, p. 5). Only one animal dispersed what would be considered a larger distance, moving 525 feet (160 meters) in a single day.

### ***Population Information and Trends***

#### **Population Trends:**

Decreasing (USFWS, 2016)

**Population Narrative:**

There are few data on historical or current population sizes of Mazama pocket gopher (pocket gopher) populations in Washington, although several local populations and one subspecies are believed to be extinct. Knowledge of the past status of the pocket gopher is limited to distributional information. Recent surveys have focused on determining current distribution, primarily in response to development applications. In addition, in 2012, WDFW initiated a five county-wide distribution survey. Because the object of all of these surveys has mainly been presence/absence only, total population numbers for each subspecies are unknown. And, the precise boundaries of each subspecies' range are not currently known. Local population estimates have been reported but are based on using apparent gopher mounds to delineate the number of territories, a method that has not been validated (Stinson 2005, pp. 40, 41). Olson (2011a, p. 2) evaluated this methodology on pocket gopher populations at the Olympia Airport and Wolf Haven International. Although there was a positive relationship between the number of mounds and number of pocket gophers, the relationship varies spatially, temporally, and demographically (Olson 2011a, pp. 2, 39). Based on the results of Olson's 2011 study, we believe past population estimates (Stinson 2005) may have been too high. As there is no generally accepted standard survey protocol to determine population size for pocket gophers, it is not currently possible to obtain an estimate of subspecies population sizes or trends. Overall habitat availability has declined, however, and habitat has a finite ability to support pocket gophers. For these reasons, the Service concludes that the overall population trend of each of the four Thurston/Pierce subspecies of the Mazama pocket gopher is negative. Increased survey effort since 2007 has resulted in the identification of numerous additional occupied sites located on private lands, especially in Thurston County (WDFW 2013a). Some of these new detections are adjacent to other known occupied sites, such as the population at the Olympia Airport. The full extent of these smaller discontinuous sites is currently unknown, and no research has been done to determine whether or not these aggregations are "stepping stone" sites that may facilitate dispersal into nearby unoccupied suitable habitat, or if they are population sinks (sites that do not add to the overall population through recruitment). Others of these additional occupied sites are separate locations, seemingly unassociated (physically) with known populations (Tirhi, in litt. 2008). The largest known expanse of areas occupied by any subspecies in Washington occur on JBLM (Roy Prairie and Yelm pocket gophers), and at the Olympia and Shelton airports (Olympia and Shelton pocket gophers, respectively). A translocated population occurs on Wolf Haven International's land near Tenino, Washington. Between 2005 and 2008, over 200 gophers from a variety of areas in Thurston County (some from around Olympia Airport (Olympia pocket gopher, *T. m. pugetensis*)) and some from near the intersection of Rich Road and Yelm Highway (assumed to be Olympia pocket gophers) were released into the 38 acres (15 ha) mounded prairie site. Based on the best available information, we do not believe the property previously supported pocket gophers. Today pocket gophers continue to occupy the site (Tirhi, in litt. 2011); however, current population estimates are not available. Another site, West Rocky Prairie Wildlife Area, has received a total of 560 translocated pocket gophers (*T. m. pugetensis*) from the Olympia Airport between 2009 and 2011. Initial translocation efforts were unsuccessful; a majority of the pocket gophers died within three days due to predation (Olson 2009, p. 3). Modified release techniques used in 2010 and 2011 resulted in improved survival rates (Olson 2011b, p. 4). It is too soon to know if the population will become self-sustaining, or if additional translocations of gophers will be necessary.

**Threats and Stressors**

**Stressor:** Destruction, Modification, or Curtailment of Habitat and Range (USFWS, 2016)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** The primary long term threats to the pocket gopher are the loss, conversion, and degradation of habitat, particularly to urban development, successional changes to grassland habitat, and the spread of invasive plants. The threats also include increased predation pressure, which is closely linked to habitat degradation. The prairies of south Puget Sound are one of the rarest ecosystems in the United States (Dunn and Ewing 1997b, p. v; Noss et al. 1995, p. 1-2). Dramatic changes have occurred on the landscape over the last 150 years, including a 90 to 95 percent reduction in the extent of the prairie ecosystem. In the south Puget Sound region, where most of western Washington's prairies historically occurred, less than 10 percent of the original prairie persists, and only three percent remains dominated by native vegetation (Crawford and Hall 1997, pp. 13, 14). Development: Native prairies and grasslands have been severely reduced throughout the range of the four Thurston/Pierce subspecies of Mazama pocket gopher, especially as a result of conversion to residential and commercial development and agriculture. Prairie habitat continues to be lost, particularly to residential development (Stinson 2005, p. 70), by removal and fragmentation of native vegetation, and the excavation, and/or heavy equipment-caused compaction of surfaces and conversion to non-habitat (e.g., buildings, pavement, other infrastructure), rendering soils unsuitable for burrowing. Residential development is associated with increased infrastructure, such as new road construction, which is one of the primary causes of landscape fragmentation (Watts et al. 2007, p. 736). Activities that accompany low-density development are correlated with decreased levels of biodiversity, mortality to wildlife, and facilitated introduction of nonnative invasive species (Trombulak and Frissell 2001; Watts et al. 2007, p. 736). In the south Puget Sound lowlands, the glacial outwash soils and gravels underlying the prairies are deep and valued for use in construction and road building, which leads to their degradation and destruction. In the south Puget Sound, Nisqually loamy soils appear to support high densities of pocket gophers (Stinson, in litt. 2010a Olson 2008, p. 6), the vast majority of which occur in developed areas of Thurston County, or within the Urban Growth Areas for the cities of Olympia, Tumwater, and Lacey (WDFW 2009), where future development is most likely to occur. Where pocket gopher populations presumably extended across an undeveloped expanse of open prairie (Dalquest and Scheffer 1942, pp. 95, 96), areas currently occupied by the four Thurston/Pierce subspecies of the Mazama pocket gopher are now isolated to small fragmented patches due to development and conversion of suitable habitat to incompatible uses. The presumed extinction of the Tacoma pocket gopher is likely linked directly to residential and commercial development, which has replaced nearly all pocket gopher habitats in the historical range of the subspecies (Stinson 2005, pp. 18, 34, 46). One of the historical Tacoma pocket gopher sites was converted to a large gravel pit and golf course (Steinberg 1996, pp. 24, 27; Stinson 2005, pp. 47, 120). In addition, two gravel pits are now operating on part of the site recognized as the type locality for the Roy Prairie pocket gopher (Stinson 2005, p. 42), and another is in operation near Tenino (Stinson, in litt. 2010b) in the vicinity of the type locality for the Tenino pocket gopher. Multiple pocket gopher sites in Pierce and Thurston Counties may be, or have been, lost to gravel pit development, golf course development, or residential and commercial development (Stinson, in litt. 2005; Stinson 2005, pp. 26, 42; Stinson, in litt. 2010b). Multiple prairies that used to contain uninterrupted expanses of prairie habitat suitable for pocket gophers within the range of the four Thurston/Pierce

subspecies have been developed to cities, neighborhoods, agricultural lands, or military bases, and/or negatively impacted by such development, including Baker Prairie, Bush Prairie, Chambers Prairie, Frost Prairie, Grand Mound Prairie, Little Chambers Prairie, Marion Prairie, Roy Prairie, Ruth Prairie, Woods Prairie, Violet Prairie, and Yelm Prairie. Some of these prairie areas still contain smaller areas that support pocket gophers, and some appear to no longer support pocket gophers at all (WDFW 2012). Where their properties coincide with pocket gopher occupancy, many private lands developers and landowners in Thurston County have agreed to create set-asides or agree to other mitigation activities in order to obtain development permits from the County (Tirhi, in litt. 2008). However, it is unknown if any pocket gophers will remain on these sites due to the small size of the set-asides, extensive grading in some areas adjacent to set-asides, lack of dedicated funding for enforcement or monitoring of set-aside maintenance (Thurston County Long Range Planning and Resource Stewardship, in litt. 2011, p. 2), and lack of control of predation by domestic or feral cats and dogs. In addition, some landowners have received variances from Thurston County that allowed development to occur without a requirement to set aside areas for pocket gophers. A population of Olympia pocket gophers is located at and around the Port of Olympia's Olympia Airport, which is sited on the historical Bush Prairie. Gophers on Bush Prairie are currently vulnerable to negative impacts from proposed future development by the Port of Olympia and ongoing development by adjacent landowners. The Port of Olympia has plans to develop large portions of the existing grassland that likely supports the largest population of the Olympia pocket gopher in Washington (Stinson 2007, in litt.; Port of Olympia and WDFW 2008, p.1; Port of Olympia 2012). The Olympia Airport is realigning the airport runway, which is in known occupied habitat. They continue to work with the Service and WDFW on mitigating airport expansion activities that may negatively impact gophers (Tirhi, in litt. 2010). The Olympia pocket gopher has a population at the Olympia Airport that spans several hundred acres, and there are two translocated populations: one at West Rocky Prairie Wildlife Area (some individuals from the Olympia Airport) and one at Wolf Haven (individuals from the Olympia Airport and some from near the intersection of Rich Road and Yelm Highway). The population centered on the Olympia Airport could be negatively impacted by plans for development both on and off the airport, while the two translocated populations are currently secure from intense commercial and residential development pressures as they occur on conserved lands. The Roy Prairie pocket gopher is known to occur across a large expanse of prairie on JBLM, which is currently secure from the threat of development. The Tenino pocket gopher has a single known population, which has been detected during surveys on the Rocky Prairie NAP, although the intermittent nature of these detections suggests it must be part of a larger metapopulation that occurs across nearby areas that have not been accessible for surveys. No known development poses a threat to the NAP, but any future conversion of the surrounding area to incompatible land use would likely hinder the recovery of this subspecies. The Yelm pocket gophers on Tenalquot prairie (which is owned in large part by JBLM) and Scatter Creek Wildlife Area are also secure from such residential and commercial development, but the Yelm pocket gopher habitat on Rock Prairie north of Old Highway 99 is in an area that is likely to be developed soon, which may negatively affect any local populations in the vicinity. Loss or Curtailment of Natural Disturbance Processes: The suppression and loss of ecological disturbance regimes across vast portions of the landscape, such as fire, has resulted in altered vegetation structure in the prairies and meadows and has facilitated invasion by native and nonnative woody vegetation, rendering habitat unusable for the four Thurston/Pierce subspecies of *Mazama* pocket gopher. The basic ecological processes that maintain prairies and meadows have disappeared from, or have been altered on, all but a few protected and managed sites. Historically, the prairies and meadows of the south Puget Sound region are thought to have been

actively maintained by native peoples, who lived here for at least 10,000 years before the arrival of Euro-American settlers (Boyd 1986; Christy and Alverson 2011, p. 93). Frequent burning reduced the encroachment and spread of shrubs and trees (Boyd 1986; Chappell and Kagan 2001, p. 42), favoring open grasslands with a variety of native plants and animals. Following Euro-American settlement of the region in the mid-19th century, fire was actively suppressed on grasslands, allowing encroachment by woody vegetation into the remaining prairie habitat and oak woodlands (Agee 1993, p. 360; Altman et al. 2001, p. 262; Boyd 1986; Franklin and Dyrness 1973, p. 122; Kruckeberg 1991, p. 287). Fires on the prairie create a mosaic of vegetation conditions, which serve to maintain native prairie plant communities. In some prairie patches fires will kill encroaching woody vegetation and reset succession back to bare ground, creating early successional vegetation conditions suitable for many native prairie species. Early succession forbs and grasses are favored by pocket gophers. The historical fire frequency on prairies has been estimated to be 3 to 5 years (Foster 2005, p. 8). On sites where regular fires occur, there is a high complement of native plants and fewer invasive species. These types of fires maintain the native short-statured plant communities favored by pocket gophers. The result of fire suppression has been the invasion of the prairies and oak woodlands by native and nonnative plant species (Dunn and Ewing 1997a, p. v; Tveten and Fonda 1999, p. 146), notably woody plants such as the native Douglas-fir (*Pseudotsuga menziesii*) and the nonnative Scot's broom. On tallgrass prairies in midwestern North America, fire suppression has led to degradation and the loss of native grasslands (Curtis 1959, pp. 296, 298; Panzer 2002, p. 1297). On northwestern prairies, fire suppression has allowed Douglas-fir to encroach on and outcompete native prairie vegetation for light, water, and nutrients (Stinson 2005, p. 7). This increase in woody vegetation and nonnative plant species has resulted in less available prairie habitat overall and habitat that is unsuitable for and avoided by many native prairie species, including pocket gophers (Olson 2011a, pp. 12, 16; Pearson et al. 2005, pp. 2, 27; Tveten and Fonda 1999, p. 155). Pocket gophers prefer early successional vegetation as forage. Woody plants shade out the forbs and grasses that pocket gophers prefer to eat, and high densities of woody plants make travel both below and above the ground difficult. In locations with poor forage, pocket gophers tend to have larger territories, which may be difficult or impossible to establish in densely forested areas. The probability of pocket gopher occupancy is much higher in areas with less than 10 percent woody vegetation cover (Olson 2011a, p. 16). On JBLM alone, over 16,000 acres (6,477 ha) of prairie has converted to Douglas-fir forest since the mid-19th century (Foster and Shaff 2003, p. 284). Where controlled burns or direct tree removal are not used as a management tool, this encroachment will continue to cause the loss of open grassland habitats for pocket gophers and is an ongoing threat to the species. Restoration in some of the south Puget Sound grasslands has resulted in temporary control of Scot's broom and other invasive plants through the careful and judicious use of herbicides, mowing, grazing, and fire. Fire has been used as a management tool to maintain native prairie composition and structure and is generally acknowledged to improve the health and composition of grassland habitat by providing a short-term nitrogen addition, which results in a fertilizer effect to vegetation, thus aiding grasses and forbs to sprout. Unintentional fires ignited by military training burn patches of prairie grasses and forbs on JBLM on an annual basis. These light ground fires create a mosaic of conditions within the grassland, maintaining a low vegetative structure of native and nonnative plant composition, and patches of bare soil. Because of the topography of the landscape, fires create a patchy mosaic of areas that burn completely, some areas that do not burn, and areas where consumption of the vegetation is mixed in its effects to the habitat. One of the benefits of fire in grasslands is that it tends to kill regenerating conifers, and reduces the cover of nonnative shrubs such as Scot's broom, although Scot's broom seed stored in the soil can be stimulated by

fire (Agee 1993, p. 367). Fire also improves conditions for many native bulb-forming plants, such as *Camassia* spp. (Agee and Dunwiddie 1984). On sites where regular fires occur, such as on JBLM, there is a high complement of native plants and fewer invasive species. These types of fires maintain the native, short-statured plant communities favored by pocket gophers. Management practices such as intentional burning and mowing require expertise in timing and technique to achieve desired results. If applied at the wrong season, frequency, or scale, fire and mowing can be detrimental to the restoration of native prairie species. Excessive and high-intensity burning can result in a lack of vegetation or encourage regrowth of nonnative grasses. Where such burning has occurred over a period of more than 50 years on the artillery ranges of JBLM, prairies are covered by nonnative forbs and grasses instead of native perennial bunchgrasses (Tveten and Fonda 1999, pp. 154, 155). Pocket gophers are not commonly found in areas colonized by Douglas-fir trees because pocket gophers require forbs and grasses of an early successional stage for food (Witmer et al. 1996a, p. 96). Pocket gophers observed on JBLM did not occur in areas with high cover of Scot's broom (Steinberg 1995, p. 26). A more recent study on JBLM also found that pocket gopher presence was negatively associated with Scot's broom (Olson 2011a, pp. 12, 13, 16). Some subspecies may disperse through forested areas or may temporarily establish territories on forest edges, but there is currently not enough data available to determine how common this behavior may be or which subspecies employ it. The four Thurston/Pierce subspecies of the *Mazama* pocket gopher occur on prairie-type habitats, many of which, if not actively managed to maintain vegetation in an early-successional state, have been invaded by shrubs and trees that either preclude pocket gophers or limit their ability to fully occupy the landscape. Typical management at civilian airports prevents woody vegetation from encroaching onto surrounding areas for flight safety reasons. Woody vegetation encroachment is therefore not a threat at civilian airports. Military Training: Pocket gopher populations occurring on JBLM are exposed to differing levels of training activities on the base. The Department of Defense's (DOD) proposed actions under their "Grow the Army" initiative include stationing 5,700 new soldiers, new combat service support units, a combat aviation brigade, facility demolition and construction to support the increased troop levels, and additional aviation, maneuver, and live fire training (75 FR 55313, September 10, 2010). The increased training activities will affect nearly all training areas at JBLM, resulting in an increased risk of accidental fires, and habitat destruction and degradation attributable to vehicle use in occupied areas, mounted and dismounted training, bivouac activities, and digging. Even though the training areas on the base are degraded, with implementation of agreed-upon conservation measures, these areas still provide habitat for the Roy Prairie and Yelm pocket gopher. JBLM's recently signed Endangered Species Management Plan (ESMP) for the *Mazama* pocket gopher will serve to minimize threats across the base by redirecting some training activities to areas outside of occupied habitat, designating areas where no vehicles are permitted, designating areas where vehicles will remain on roads only, and designating areas where no digging is allowed, among other conservation measures. JBLM has further committed to enhancing and expanding suitable habitat for the Roy Prairie and Yelm pocket gophers in "priority habitat" areas on base (areas that were proposed as critical habitat); enforcing restrictions on recreational use of occupied habitat by dog owners and horseback riders; and continuing to support the off-base recovery of the four Thurston/Pierce subspecies of the *Mazama* pocket gopher. Several moderate- to large-sized areas supporting pocket gophers have been identified on JBLM. These areas are within the historical ranges of the Roy Prairie (Pierce County) and Yelm (Thurston County) pocket gophers. Their absence from some sites of what is presumed to have been formerly suitable habitat may be related to compaction of the soil due to years of mechanized vehicle training (Steinberg 1995, p. 36). Training infrastructure (e.g., roads, firing ranges,

bunkers) also degrades pocket gopher habitat and may lead to reduced use of these areas by pocket gophers. For example, JBLM has plans to add a third rifle range on the south impact area where it overlaps with a densely occupied pocket gopher site. The area may be usable by pocket gophers when the project is completed; however, construction of the rifle range may result in removal of forage and direct mortality of pocket gophers through crushing of burrows (Stinson, in litt. 2011). Recent survey access to the center of the artillery impact area on 91st Division Prairie, where bombardment is presumably of the highest intensity, did detect some unspecified level of occupancy by the Roy Prairie pocket gopher (WDFW 2013b, enclosure 1, p. 6). This apparently suitable central portion of the 91st Division Prairie is subject to repeated and ongoing bombardment, which may create an ecological trap for dispersing juveniles. JBLM training areas have varying levels of use; some allow excavation and off-road vehicle use, while other areas have restrictions that limit off-road vehicle use. The ESMP specifically requires coordination between the JBLM Fish and Wildlife personnel and the JBLM entities responsible for training activities (e.g., Range Support, battalion commanders, and/or first field grade officers) to ensure all parties are aware of where occupied areas occur in relation to training activities, the effects of training, and the potential ramifications of habitat destruction or animal mortality. Since military training has the potential to directly or indirectly harm or harass pocket gophers, we conclude that these activities will negatively impact the Roy Prairie and Yelm pocket gophers. JBLM has committed to operational restrictions on portions of the base in order to avoid and minimize potential impacts to Roy Prairie and Yelm pocket gophers. Currently-occupied areas will be buffered from training activities, with an emphasis on occupied habitat in "priority habitat" areas. Regular surveys will be conducted with the goals of determining distribution, protecting pocket gophers and their habitat from disturbance or destruction, and determining population status. Where possible, JBLM will alleviate training pressure by transferring activities to unoccupied areas where encroaching forest has been removed. This strategy has the effect of both releasing large areas of land that were historically prairie and providing unoccupied areas where training is free of the risk of negatively impacting Roy Prairie or Yelm pocket gophers. While the Service fully supports the implementation of these impact minimization efforts and will continue to collaborate with DOD to address all aspects of training impacts on the species, not all adverse impacts on pocket gophers can be fully avoided. Military training continues to pose a threat to the Roy Prairie and Yelm subspecies at this time. No military training occurs in the ranges of the Olympia or Tenino subspecies of the Mazama pocket gopher (USFWS, 2016).

**Stressor:** Poor Connectivity Between Small and Isolated Populations (USFWS, 2016)

**Exposure:**

**Response:**

**Consequence:** Isolated genetics

**Narrative:** Most species' populations fluctuate naturally, responding to various factors such as weather events, disease, and predation. Populations that are small, fragmented, or isolated by habitat loss or modification of naturally patchy habitat, and other human-related factors, are more vulnerable to extirpation by natural randomly occurring events, cumulative effects, and to genetic effect (collectively known as small population effects). These effects can include genetic drift (loss of recessive alleles), founder effects (over time, an increasing percentage of the population inheriting a narrow range of traits), and genetic bottlenecks leading to increasingly lower genetic diversity, with consequent negative effects on evolutionary potential. To date, of the eight subspecies of Mazama pocket gopher in Washington, only the Olympic pocket gopher has been documented as having low genetic diversity (Welch and Kenagy 2008, p. 7), although the six other extant subspecies have local populations that are small, fragmented, and physically

isolated from one another. The four Thurston/Pierce subspecies of the *Mazama* pocket gopher face threats from loss or fragmentation of habitat. Historically, pocket gophers probably persisted by continually recolonizing habitat patches after local extinctions. However, widespread development and conversion of habitat has resulted in widely separated populations, and intervening habitat corridors are now gone, with the effect of impeding or stopping much of the natural recolonization that historically occurred (Stinson 2005, p. 46). Although pocket gophers are not known to have low genetic diversity, small population sizes at most sites, coupled with disjunct and fragmented habitat, may contribute to further population declines. Little is known about the local or rangewide reproductive success of pocket gophers found in Washington State (USFWS, 2016).

**Stressor:** Predation and Pest Control (USFWS, 2016)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** Predation: Predation influences the distribution, abundance, and diversity of species in ecological communities. Generally, predation leads to changes in both the population size of the predator and that of the prey. In unfavorable environments, prey species are stressed or living at low population densities such that predation is likely to have negative effects on all prey species, thus lowering species richness. In addition, when a nonnative predator is introduced to the ecosystem, negative effects on the prey population may be higher than those from co-evolved native predators. The effect of predation may be magnified when populations are small, and the disproportionate effect of predation on declining populations has been shown to drive rare species even further towards extinction (Woodworth 1999, pp. 74, 75). Predation has an impact on populations of the four Thurston/Pierce subspecies of *Mazama* pocket gopher. Urbanization, particularly in the south Puget Sound region, has resulted in not only habitat loss, but also increased exposure to feral and domestic cats and dogs. Domestic cats are known to have serious impacts on small mammals and birds and have been implicated in the decline of several endangered and threatened mammals, including marsh rabbits (*Sylvilagus palustris*) in Florida and the salt-marsh harvest mouse (*Reithrodontomys raviventris*) in California (Ogan and Jurek 1997, p. 89). Domestic cats and dogs have been specifically identified as common predators of pocket gophers (Case and Jasch 1994, p. B-21; Henderson 1981, p. 233; Wight 1918, p. 21) and at least two pocket gopher locations were found as a result of house cats bringing home pocket gopher carcasses (WDFW 2001). Informal interviews with area biologists document multiple incidents of domestic pet predation on pocket gophers (Chan, in litt. 2013; Clouse, in litt. 2012 Skriletz 2013 in litt., Wood 2013 in litt.). There is also one recorded instance of a WDFW biologist being presented with a dead *Mazama* pocket gopher by a dog during an east Olympia, Washington, site visit in 2006 (Burke Museum 2012 McAllister 2013 in litt.). Some local populations of the pocket gopher occur in areas where people recreate with their dogs, bringing these potential predators into environments that may otherwise be relatively free of them, consequently increasing the risks to individual pocket gophers and populations that may be small and isolated. The four Thurston/Pierce subspecies of *Mazama* pocket gopher occur in rapidly developing areas. Local populations that survive commercial and residential development (adjacent to and within habitat) are potentially vulnerable to extirpation by domestic and feral cats and dogs (Case and Jasch 1994, p. B-21; Henderson 1981, p. 233). As stated previously, predation is a natural part of the pocket gopher's life history; however, the effect of predation may be magnified when populations are small and habitat is fragmented. The disproportionate effect of additional predation on declining populations has been shown to drive rare species even

further towards extinction (Woodworth 1999, pp. 74, 75). Predation, particularly from nonnative species, will likely continue to be a threat to the four Thurston/Pierce subspecies of the Mazama pocket gopher now and in the future. This is particularly likely where development abuts gopher habitat, resulting in increased numbers of cats and dogs in the vicinity, and in areas where people recreate with their dogs – particularly if dogs are off-leash and not prevented from harassing wildlife. In such areas, where local populations of pocket gophers are already small, this additional predation pressure (above natural levels of predation) is expected to further negatively impact population numbers. Pest Control: Pocket gophers are often considered a pest because they sometimes damage crops and seedling trees, and their mounds can create a nuisance. Several site locations were found as a result of trapping conducted on Christmas tree farms, a nursery, and in a livestock pasture (WDFW 2001). The type locality for the Cathlamet pocket gopher is on a commercial tree farm. Pocket gophers from Thurston County were used in a rodenticide experiment as recently as 1995 (Witmer et al. 1996a, p. 97). In Washington State it is currently illegal to trap or poison Mazama pocket gophers, or to trap or poison moles where they overlap with Mazama pocket gopher populations, but not all property owners are cognizant of these laws, nor are most citizens capable of differentiating between moles, pocket gophers, or the signs of their habitation (e.g., soil disturbance). In light of this, it is reasonable to believe that mole trapping or poisoning still has the potential to adversely affect pocket gopher populations. Local populations that survive commercial and residential development (adjacent to and within habitat) may be subsequently extirpated by trapping or poisoning. Lethal control by trapping or poisoning is most likely to be a threat to the four Thurston/Pierce subspecies where their ranges overlap residential properties (USFWS, 2016).

### **Recovery**

### **References**

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Designation of Critical Habitat for Mazama Pocket Gophers

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## SPECIES ACCOUNT: *Thomomys mazama tumuli* (Tenino pocket gopher)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Threatened; Pacific Region (R1) (USFWS, 2016)

### **Physical Description**

Adult Mazama pocket gophers are reddish brown to black above, and the underparts are lead-colored with buff-colored tips. The lips, nose, and patches behind the ears are black; the wrists are white. Adults range from 7 to 9 inches (189 to 220 millimeters (mm)) in total length, with tails that range from 2 to 3 inches (45 to 85 mm)(Verts and Carraway 2000, p.2). Mazama pocket gophers are morphologically similar to other species of pocket gophers that exploit a subterranean existence. They are stocky and tubular in shape, with short necks, powerful limbs, long claws, and tiny ears and eyes. Their short, nearly hairless tails are highly sensitive and probably assist when navigating tunnels. The “pockets” are external, fur-lined cheek pouches on either side of the mouth that are used to transport nesting material and plant cuttings. Mazama pocket gophers reach reproductive age in the spring of the year after their birth and produce litters between spring and early summer. Litter size ranges from one to nine (Wight 1918, p. 14), with an average of five (Scheffer 1938, p. 222). They do not hibernate in winter; they remain active throughout the year (Case and Jasch 1994, p. B-20) (USFWS, 2016).

### **Taxonomy**

Although the species *Thomomys mazama*, or Mazama pocket gopher, includes numerous subspecies that are found in the States of Washington, Oregon, and California, only the subspecies found in the State of Washington have recently been considered for listing. The Mazama pocket gopher complex consists of 15 subspecies, eight of which occur only in Washington, five of which occur only in Oregon, one that occurs only in California, and one subspecies with a distribution that spans the boundary between Oregon and California (Hall 1981, p. 467). The first pocket gophers collected in western Washington were considered subspecies of the northern pocket gopher (*Thomomys talpoides*)(Goldman 1939), until 1960 when the complex of pocket gophers found in western Washington was determined to be more similar to the western pocket gopher (*T. mazama*)(Johnson and Benson 1960, p. 20). Eight western Washington subspecies of Mazama pocket gopher (*T. mazama*, ssp. *couchi*, *glacialis*, *louiei*, *melanops*, *pugetensis*, *tacomensis*, *tumuli*, and *yelmensis*) have been identified (Hall 1981, p. 467). *Thomomys mazama* is recognized as a valid species by the Integrated Taxonomic Information System (ITIS 2012). Although there have been suggestions that potential changes to the classification of some of these subspecies should be considered, we have no information to suggest that any of the presently recognized subspecies are the subject of serious dispute. We follow the subspecies designations of Verts and Carraway (2000), as this text represents the currently accepted taxonomy for the species *T. mazama*. Verts and Carraway (2000, p.1) recognize *T. m. glacialis*, *pugetensis*, *tumuli*, and *yelmensis* as separate subspecies (the Roy Prairie, Olympia, Tenino, and Yelm pocket gophers, respectively) based on morphological characteristics, distribution, and differences in number of chromosomes. Due to the close proximity of the four subspecies located in Thurston and Pierce Counties, and the fact that at least three of them occur in the same clade, we refer to these four subspecies (*T. m. glacialis*,

pugetensis, tumuli, and yelmensis) as “the four Thurston/Pierce subspecies” of the Mazama pocket gopher (USFWS, 2016).

**Current Range**

Tenino pocket gophers (*Thomomys mazama tumuli*) were originally found in the vicinity of the Rocky Prairie NAP, near Tenino (Dalquest and Scheffer 1942, p. 96), a relatively small prairie area. Gophers still reside there, but WDFW researchers have not seen consistent occupancy of the area in recent years (Olson, in litt. 2010), suggesting that the activity intermittently detected in the NAP may be attributable to individuals dispersing from a currently unidentified nearby source. Soil series and soil series complexes in this area that may support pocket gophers include Everett, Nisqually, Norma, Spanaway, and Spanaway-Nisqually complex. In Washington, Mazama pocket gophers are found west of the Cascade Mountain Range, in the Olympic Mountains and in the Puget Sound trough, with an additional single locality known from Wahkiakum County (Verts and Carraway 2000, p.3). Their populations are concentrated in well-drained friable soils often associated with glacial outwash (USFWS, 2016).

**Critical Habitat Designated**

Yes; 4/9/2014.

**Legal Description**

On April 9, 2014, the U.S. Fish and Wildlife Service (Service) designated critical habitat for three subspecies of the Mazama pocket gopher (the Olympia pocket gopher, *Thomomys mazama pugetensis*; the Tenino pocket gopher, *T. m. tumuli*; and the Yelm pocket gopher, *T. m. yelmensis*) under the Endangered Species Act of 1973, as amended (Act). In total, approximately 1,607 acres (650 hectares) in Thurston County, Washington, fall within the boundaries of the critical habitat designation for the Olympia, Tenino, and Yelm pocket gophers. The effect of this regulation was to designate critical habitat for the Olympia, Tenino, and Yelm subspecies of the Mazama pocket gopher found in Thurston County, Washington, under the Act.

**Critical Habitat Designation**

Tenino Pocket Gopher Critical Habitat—Rocky Prairie Unit. This unit consists of 399 ac (162 ha) and is owned by one commercial land owner and Burlington Northern Santa Fe Railroad. The Rocky Prairie Unit is located north of the city of Tenino, Thurston County, Washington; is likely occupied by the Tenino pocket gopher; and contains the physical or biological features essential to the conservation of the species due to the underlying soil series or soil series complex (Everett, Nisqually, Spanaway, and Spanaway-Nisqually complex), suitable forb and grass vegetation present onsite, and its large size.

**Primary Constituent Elements/Physical or Biological Features**

Critical habitat for the Tenino pocket gopher is designated in Thurston County, Washington. Within this area, the primary constituent elements of the physical or biological features essential to the conservation of Tenino pocket gopher consist of two components:

(i) Friable, loamy, and deep soils, some with relatively greater content of sand, gravel, or silt, all generally on slopes less than 15 percent in the following soil series or soil series complex: (A) Alderwood; (B) Cagey; (C) Everett; (D) Indianola; (E) Kapowsin; (F) Nisqually; (G) Norma; (H) Spanaway; (I) Spanaway-Nisqually complex; and (J) Yelm.

(ii) Areas equal to or larger than 50 ac (20 ha) in size that provide for breeding, foraging, and dispersal activities, found in the soil series listed in paragraph (2)(i) of this entry that have: (A) Less than 10 percent woody vegetation cover; (B) Vegetative cover suitable for foraging by gophers. Pocket gophers' diets include a wide variety of plant material, including leafy vegetation, succulent roots, shoots, tubers, and grasses. Forbs and grasses that Mazama pocket gophers are known to eat include, but are not limited to: *Achillea millefolium* (common yarrow), *Agoseris* spp. (agoseris), *Cirsium* spp. (thistle), *Bromus* spp. (brome), *Camassia* spp. (camas), *Collomia linearis* (tiny trumpet), *Epilobium* spp. (several willowherb spp.), *Eriophyllum lanatum* (woolly sunflower), *Gayophytum diffusum* (groundsmoke), *Hypochaeris radicata* (hairy cat's ear), *Lathyrus* spp. (peavine), *Lupinus* spp. (lupine), *Microsteris gracilis* (slender phlox), *Penstemon* spp. (penstemon), *Perideridia gairdneri* (Gairdner's yampah), *Phacelia heterophylla* (varileaf phacelia), *Polygonum douglasii* (knotweed), *Potentilla* spp. (cinquefoil), *Pteridium aquilinum* (bracken fern), *Taraxacum officinale* (common dandelion), *Trifolium* spp. (clover), and *Viola* spp. (violet); and (C) Few, if any, barriers to dispersal. Barriers to dispersal may include, but are not limited to, forest edges, roads (paved and unpaved), abrupt elevation changes, Scot's broom thickets, highly cultivated lawns, inhospitable soil types or substrates, development and buildings, slopes greater than 35 percent, and open water.

### **Special Management Considerations or Protections**

Critical habitat does not include manmade structures (such as buildings, aqueducts, runways, roads, and other paved areas) and the land on which they are located existing within the legal boundaries on May 9, 2014.

The physical or biological features essential to the conservation of each subspecies may require special management considerations or protection to restore, protect, and maintain the essential features found there. The physical or biological features in this subunit are threatened by: Loss of habitat through conversion to incompatible uses, such as pit mining; development on adjacent or surrounding areas; the loss of natural disturbance processes and invasion by woody plants; predation; small or isolated populations as a result of habitat fragmentation; habitat degradation or destruction as the result of the inadequacy of existing regulatory mechanisms; and control as a pest species. We additionally evaluated this area as if it were presently unoccupied by the Tenino pocket gopher, and have determined that it is nonetheless essential to the conservation of the species.

The physical or biological features essential to the conservation of the four Thurston/Pierce subspecies of the Mazama pocket gopher may require special management considerations or protection to control or prevent the establishment of invasive woody plants, which create shade and compete for light, food and nutrients otherwise utilized by the forb, bulb, and grass species that the gophers require for forage. Management may be implemented using hand tools or mechanical methods, prescribed fire, and the judicious use of herbicides. Although several management techniques are being implemented on public lands, we may need to improve our outreach to educate private landowners on controlling their pets and appropriately managing grazing on their properties, as well as to developing incentives for landowners who agree to conserve habitat. Incentives would create protected areas, through agreements or acquisitions. These would include corridors between existing protected habitat areas that may require management, enhancement actions, and long-term maintenance.

### **Life History**

**Feeding Narrative**

Adult: Pocket gophers are generalist herbivores and their diet includes a wide variety of plant material, including leafy vegetation, succulent roots, shoots, and tubers. In natural settings pocket gophers play a key ecological role by aerating soils, activating the seed bank, and stimulating plant growth, though they can be considered pests in agricultural systems. In prairie and meadow ecosystems, pocket gopher activity plays an important role in maintaining species richness and diversity. Foraging primarily takes place below the surface of the soil, where pocket gophers snip off roots of plants before occasionally pulling the whole plant below ground to eat or store in caches. If above-ground foraging occurs, it's usually within a few feet of an opening and forage plants are quickly cut into small pieces and carried back to the nest or cache (Wight 1918, p. 12). Any water they need is obtained from their food (Gettinger 1984, pp. 749-750; Wight 1918, p. 13). The probability of pocket gopher occupancy is much higher in areas with less than 10 percent woody vegetation cover (Olson 2011a, p. 16), presumably because such vegetation will shade out the forbs, bulbs, and grasses that pocket gophers prefer to eat, and high densities of woody plants make travel both below and above the ground difficult (USFWS, 2016).

**Reproduction Narrative**

Adult: Pocket gophers reach sexual maturity during the spring of the year following their birth, and generally produce one litter per year (Case and Jasch 1994, p. B-20), though timing of sexual maturity has been shown to vary with habitat quality (Patton and Brylski 1987, p. 502; Patton and Smith 1990, p. 76). Gestation lasts approximately 18 days (Andersen 1978, p. 421; Schramm 1961, p. 169). Young are born in the spring to early summer (Wight 1918, p. 13), and are reared by the female. Aside from the breeding season, males and females remain segregated in their own tunnel systems. There are 1-9 pups per litter (averaging 5), born without hair, pockets, or teeth, and they must be kept warm by the mother or "packed" in dried vegetation (Case and Jasch 1994, p. B-20; Wight 1918, p. 14). Juvenile pelage starts growing in at just over a week (Andersen 1978, p. 420). The young eat vegetation in the nest within three weeks of birth, with eyes and ears opening and pockets developing at about a month (Andersen 1978, p. 420; Wight 1918, p. 14). At six weeks they are weaned, fighting with siblings, and nearly ready to disperse (Andersen 1978, p. 420; Wight 1918, p. 15), which usually occurs at about two months of age (Stinson 2005, p. 26). They attain their adult weight between four and five months of age (Andersen 1978, pp. 419, 421). Most pocket gophers live only a year or two, with few living to three or four years of age (Hansen 1962, pp. 152, 153; Livezey and Verts 1979, p. 39) (USFWS, 2016).

**Spatial Arrangements of the Population**

Adult: Clumped (USFWS, 2016)

**Habitat Narrative**

Adult: The Mazama pocket gopher (pocket gopher) is associated with glacial outwash prairies in western Washington, an ecosystem of conservation concern (Hartway and Steinberg 1997, p. 1), as well as alpine and subalpine meadows and other meadow-like openings at lower elevations. Steinberg and Heller (1997, p. 46) found that pocket gophers are even more patchily distributed than are prairies, as there are some seemingly high quality prairies within the species' range that lack pocket gophers; e.g., Mima Mounds Natural Area Preserve (NAP), and 13th Division Prairie on Joint Base Lewis-McChord (JBLM). Pocket gopher distribution is affected by the rock

content of soils, drainage, forage availability, and climate (Case and Jasch 1994, p. B-21; Hafner et al. 1998, p. 279; Reichman 2007, pp. 273-274; Steinberg and Heller 1997, p. 45; Stinson 2005, p. 31; WDFW 2009). Prairie and meadow habitats used by pocket gophers have a naturally patchy distribution. In their prairie habitats, there is an even patchier distribution of soil rockiness which may further restrict the total area that pocket gophers can utilize (Steinberg and Heller 1997, p. 45; WDFW 2009). We assume that meadow soils have a similarly patchy distribution of rockiness, though the soil surveys to support this are, at this time, incomplete. In western Washington, pocket gophers currently occupy the following soils series: Alderwood, Cagey, Carstairs, Everett, Everett-Spanaway complex, Everett-Spanaway-Spana complex, Godfrey, Grove, Indianola, Kapowsin, McKenna, Murnen, Nisqually, Norma, Shelton, Spana, Spana-Spanaway-Nisqually complex, Spanaway, Spanaway-Nisqually complex, and Yelm. No soil survey information is currently available for occupied sites in the Olympic National Park, so the soils occupied there are unknown. We purposely avoid using specific map unit names, because we know that there are imperfections in soil mapping. Maps are based on the technology, standards, and tools available at the time soil surveys were conducted, sometimes up to 50 years ago. We recognize that soil survey boundaries may be adjusted in the future, and that soil series names may be added or removed to soil survey maps and databases. As a result, the overlap of pocket gopher locations with soil series names may be different in the future. The soils information presented here is based on best scientific data available at the time of listing. We also recognize that some of these soil series or soil series complexes are not typically either deep or well-drained. For a variety of reasons, mapped soil types may or may not have all of the characteristics described by the U.S. Department of Agriculture, Natural Resources Conservation Service, and the actual soils that occur on sites may have characteristics that make them more or less habitable by pocket gophers. These reasons may include: map boundary or transcription errors, map projection errors or differences, map identification or typing errors, soil or hydrological manipulations that have occurred since mapping took place, and small-scale inclusions that are different from the mapped soil. Because soils are mapped at large scales, mapped soils may not identify smaller inclusions. Any of the soil series or soil series complexes listed above could potentially be suitable for the four Thurston/Pierce subspecies of the Mazama pocket gopher. And, the four Thurston/Pierce subspecies of the Mazama pocket gopher may also inhabit soil series not included in the above list. Although some soils are sandier, more gravelly, or may have more or less silt than described, most all soils used by pocket gophers are friable (easily pulverized or crumbled), loamy, and deep, and generally have slopes less than 15 percent. There have been reports of pocket gophers (subspecies unknown) occurring on other types of soils, on managed forest lands in Capitol State Forest (owned by the Washington State Department of Natural Resources, WDNR) and Vail Forest (owned by Weyerhaeuser) in Thurston County. These were subsequently determined to be moles (*Scapanus* spp.), based on trapping conducted in these areas by the Washington State Department of Fish and Wildlife (WDFW) during 2012 (Thompson, pers. comm. 2012b). A study of the relationship between soil rockiness and pocket gopher distribution revealed a strong negative correlation between the proportion of medium-sized rocks in the soil, and the presence of pocket gophers (eight of nine prairies sampled); medium sized rocks were considered greater than 0.5 inch (12.7 mm), but less than 2 inches (50.8 mm) in diameter (Steinberg 1996, p. 32). In observations of pocket gopher distribution on JBLM, pocket gophers did not occur in areas with a high percentage of Scot's broom cover (*Cytisus scoparius*), or where mole populations were particularly dense (Steinberg 1995, p. 26). A more recent study on JBLM also found that pocket gopher presence was negatively associated with Scot's broom; however, the researcher found no relationship between pocket gopher presence and mole

density (Olson 2011a, pp. 12, 13). Pocket gopher burrows consist of a series of main runways, off which lateral tunnels lead to the surface of the ground (Wight 1918, p. 7). Pocket gophers dig their burrows using their sharp teeth and claws and then push the soil out through the lateral tunnels (Case and Jasch 1994, p. B-20; Wight 1918, p. 8). Nests containing dried vegetation are generally located near the center of each pocket gopher's home tunnel system (Wight 1918, p. 10). Food caches and store piles are usually placed near the nest, and excrement is piled into blind tunnels or loop tunnels, and then covered with dirt, leaving the nest and main runways clean (Wight 1918, p. 11). (USFWS, 2016).

***Dispersal/Migration*****Motility/Mobility**

Adult: High (USFWS, 2016)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory (USFWS, 2016)

**Dispersal**

Adult: Low (USFWS, 2016)

**Immigration/Emigration**

Adult: Unlikely (USFWS, 2016)

**Dispersal/Migration Narrative**

Adult: Pocket gophers have limited dispersal capabilities (Williams and Baker 1976, p. 303). Mazama pocket gophers are smaller in size than other sympatric or peripatric *Thomomys* species (Verts and Carraway 2000, p. 1). Both dispersal distance and home range size are therefore likely to be smaller than for other *Thomomys* species. Dispersal distances may vary based on surface or soil conditions and size of the animal. For other, larger, *Thomomys* species, dispersal distances average about 131 feet (40 meters) (Barnes Jr. 1973, pp. 168, 169; Daly and Patton 1990, pp. 1286, 1288; Williams and Baker 1976, p. 306). Initial results from research being conducted on JBLM indicate that juvenile pocket gophers usually make movements from 13.1 to 32.8 feet (4-10 meters), though these may not be dispersal movements. One juvenile made a distinct dispersal movement of 525 feet (160 meters) in a single day (Olson 2012, p. 5). Suitable dispersal habitat is free of barriers to movement, and may need to contain foraging habitat if an animal is required to make a long-distance dispersal movement. Potential barriers include, but are not limited to, forest edges, roads (paved and unpaved), abrupt elevation changes, Scot's broom thickets (Olson 2012, p. 3), highly cultivated lawns, inhospitable soil types or substrates (Olson 2008, p. 4), development and buildings, slopes greater than 35 percent, and open water. Barriers may be permeable, meaning that they impede movement from place to place without completely blocking it, or they may be impermeable, meaning they cannot be crossed. Permeable barriers, as well as lower quality dispersal habitats, may present a risk of mortality for animals that use them (e.g., open areas where predation risk is increased, or a paved area where vehicular mortality is high). The WDFW conducted a study to determine dispersal distances of juvenile pocket gophers on JBLM. Twenty-eight juveniles were radio-collared and tracked for 17 to 56 days, with all but three animals tracked for more than 30 days. Of these, only nine gophers moved more than 32.8 feet (10 meters), and 10 gophers were never found more than 13.1 feet (4 meters) from any previous location (Olson 2012, p. 5). Only one

animal dispersed what would be considered a larger distance, moving 525 feet (160 meters) in a single day.

### ***Population Information and Trends***

#### **Population Trends:**

Decreasing (USFWS, 2016)

#### **Population Narrative:**

There are few data on historical or current population sizes of Mazama pocket gopher (pocket gopher) populations in Washington, although several local populations and one subspecies are believed to be extinct. Knowledge of the past status of the pocket gopher is limited to distributional information. Recent surveys have focused on determining current distribution, primarily in response to development applications. In addition, in 2012, WDFW initiated a five county-wide distribution survey. Because the object of all of these surveys has mainly been presence/absence only, total population numbers for each subspecies are unknown. And, the precise boundaries of each subspecies' range are not currently known. Local population estimates have been reported but are based on using apparent gopher mounds to delineate the number of territories, a method that has not been validated (Stinson 2005, pp. 40, 41). Olson (2011a, p. 2) evaluated this methodology on pocket gopher populations at the Olympia Airport and Wolf Haven International. Although there was a positive relationship between the number of mounds and number of pocket gophers, the relationship varies spatially, temporally, and demographically (Olson 2011a, pp. 2, 39). Based on the results of Olson's 2011 study, we believe past population estimates (Stinson 2005) may have been too high. As there is no generally accepted standard survey protocol to determine population size for pocket gophers, it is not currently possible to obtain an estimate of subspecies population sizes or trends. Overall habitat availability has declined, however, and habitat has a finite ability to support pocket gophers. For these reasons, the Service concludes that the overall population trend of each of the four Thurston/Pierce subspecies of the Mazama pocket gopher is negative. Increased survey effort since 2007 has resulted in the identification of numerous additional occupied sites located on private lands, especially in Thurston County (WDFW 2013a). Some of these new detections are adjacent to other known occupied sites, such as the population at the Olympia Airport. The full extent of these smaller discontinuous sites is currently unknown, and no research has been done to determine whether or not these aggregations are "stepping stone" sites that may facilitate dispersal into nearby unoccupied suitable habitat, or if they are population sinks (sites that do not add to the overall population through recruitment). Others of these additional occupied sites are separate locations, seemingly unassociated (physically) with known populations (Tirhi, in litt. 2008). The largest known expanse of areas occupied by any subspecies in Washington occur on JBLM (Roy Prairie and Yelm pocket gophers), and at the Olympia and Shelton airports (Olympia and Shelton pocket gophers, respectively). A translocated population occurs on Wolf Haven International's land near Tenino, Washington. Between 2005 and 2008, over 200 gophers from a variety of areas in Thurston County (some from around Olympia Airport (Olympia pocket gopher, *T. m. pugetensis*)) and some from near the intersection of Rich Road and Yelm Highway (assumed to be Olympia pocket gophers) were released into the 38 acres (15 ha) mounded prairie site. Based on the best available information, we do not believe the property previously supported pocket gophers. Today pocket gophers continue to occupy the site (Tirhi, in litt. 2011); however, current population estimates are not available. Another site, West Rocky Prairie Wildlife Area, has received a total of 560 translocated pocket gophers

(*T. m. pugetensis*) from the Olympia Airport between 2009 and 2011. Initial translocation efforts were unsuccessful; a majority of the pocket gophers died within three days due to predation (Olson 2009, p. 3). Modified release techniques used in 2010 and 2011 resulted in improved survival rates (Olson 2011b, p. 4). It is too soon to know if the population will become self-sustaining, or if additional translocations of gophers will be necessary.

### ***Threats and Stressors***

**Stressor:** Destruction, Modification, or Curtailment of Habitat and Range (USFWS, 2016)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** The primary long term threats to the pocket gopher are the loss, conversion, and degradation of habitat, particularly to urban development, successional changes to grassland habitat, and the spread of invasive plants. The threats also include increased predation pressure, which is closely linked to habitat degradation. The prairies of south Puget Sound are one of the rarest ecosystems in the United States (Dunn and Ewing 1997b, p. v; Noss et al. 1995, p. I-2). Dramatic changes have occurred on the landscape over the last 150 years, including a 90 to 95 percent reduction in the extent of the prairie ecosystem. In the south Puget Sound region, where most of western Washington's prairies historically occurred, less than 10 percent of the original prairie persists, and only three percent remains dominated by native vegetation (Crawford and Hall 1997, pp. 13, 14). Development: Native prairies and grasslands have been severely reduced throughout the range of the four Thurston/Pierce subspecies of *Mazama* pocket gopher, especially as a result of conversion to residential and commercial development and agriculture. Prairie habitat continues to be lost, particularly to residential development (Stinson 2005, p. 70), by removal and fragmentation of native vegetation, and the excavation, and/or heavy equipment-caused compaction of surfaces and conversion to non-habitat (e.g., buildings, pavement, other infrastructure), rendering soils unsuitable for burrowing. Residential development is associated with increased infrastructure, such as new road construction, which is one of the primary causes of landscape fragmentation (Watts et al. 2007, p. 736). Activities that accompany low-density development are correlated with decreased levels of biodiversity, mortality to wildlife, and facilitated introduction of nonnative invasive species (Trombulak and Frissell 2001; Watts et al. 2007, p. 736). In the south Puget Sound lowlands, the glacial outwash soils and gravels underlying the prairies are deep and valued for use in construction and road building, which leads to their degradation and destruction. In the south Puget Sound, Nisqually loamy soils appear to support high densities of pocket gophers (Stinson, in litt. 2010a Olson 2008, p. 6), the vast majority of which occur in developed areas of Thurston County, or within the Urban Growth Areas for the cities of Olympia, Tumwater, and Lacey (WDFW 2009), where future development is most likely to occur. Where pocket gopher populations presumably extended across an undeveloped expanse of open prairie (Dalquest and Scheffer 1942, pp. 95, 96), areas currently occupied by the four Thurston/Pierce subspecies of the *Mazama* pocket gopher are now isolated to small fragmented patches due to development and conversion of suitable habitat to incompatible uses. The presumed extinction of the Tacoma pocket gopher is likely linked directly to residential and commercial development, which has replaced nearly all pocket gopher habitats in the historical range of the subspecies (Stinson 2005, pp. 18, 34, 46). One of the historical Tacoma pocket gopher sites was converted to a large gravel pit and golf course (Steinberg 1996, pp. 24, 27; Stinson 2005, pp. 47, 120). In addition, two gravel pits are now operating on part of the site recognized as the type locality for the Roy Prairie pocket gopher

(Stinson 2005, p. 42), and another is in operation near Tenino (Stinson, in litt. 2010b) in the vicinity of the type locality for the Tenino pocket gopher. Multiple pocket gopher sites in Pierce and Thurston Counties may be, or have been, lost to gravel pit development, golf course development, or residential and commercial development (Stinson, in litt. 2005; Stinson 2005, pp. 26, 42; Stinson, in litt. 2010b). Multiple prairies that used to contain uninterrupted expanses of prairie habitat suitable for pocket gophers within the range of the four Thurston/Pierce subspecies have been developed to cities, neighborhoods, agricultural lands, or military bases, and/or negatively impacted by such development, including Baker Prairie, Bush Prairie, Chambers Prairie, Frost Prairie, Grand Mound Prairie, Little Chambers Prairie, Marion Prairie, Roy Prairie, Ruth Prairie, Woods Prairie, Violet Prairie, and Yelm Prairie. Some of these prairie areas still contain smaller areas that support pocket gophers, and some appear to no longer support pocket gophers at all (WDFW 2012). Where their properties coincide with pocket gopher occupancy, many private lands developers and landowners in Thurston County have agreed to create set-asides or agree to other mitigation activities in order to obtain development permits from the County (Tirhi, in litt. 2008). However, it is unknown if any pocket gophers will remain on these sites due to the small size of the set-asides, extensive grading in some areas adjacent to set-asides, lack of dedicated funding for enforcement or monitoring of set-aside maintenance (Thurston County Long Range Planning and Resource Stewardship, in litt. 2011, p. 2), and lack of control of predation by domestic or feral cats and dogs. In addition, some landowners have received variances from Thurston County that allowed development to occur without a requirement to set aside areas for pocket gophers. A population of Olympia pocket gophers is located at and around the Port of Olympia's Olympia Airport, which is sited on the historical Bush Prairie. Gophers on Bush Prairie are currently vulnerable to negative impacts from proposed future development by the Port of Olympia and ongoing development by adjacent landowners. The Port of Olympia has plans to develop large portions of the existing grassland that likely supports the largest population of the Olympia pocket gopher in Washington (Stinson 2007, in litt.; Port of Olympia and WDFW 2008, p.1; Port of Olympia 2012). The Olympia Airport is realigning the airport runway, which is in known occupied habitat. They continue to work with the Service and WDFW on mitigating airport expansion activities that may negatively impact gophers (Tirhi, in litt. 2010). The Olympia pocket gopher has a population at the Olympia Airport that spans several hundred acres, and there are two translocated populations: one at West Rocky Prairie Wildlife Area (some individuals from the Olympia Airport) and one at Wolf Haven (individuals from the Olympia Airport and some from near the intersection of Rich Road and Yelm Highway). The population centered on the Olympia Airport could be negatively impacted by plans for development both on and off the airport, while the two translocated populations are currently secure from intense commercial and residential development pressures as they occur on conserved lands. The Roy Prairie pocket gopher is known to occur across a large expanse of prairie on JBLM, which is currently secure from the threat of development. The Tenino pocket gopher has a single known population, which has been detected during surveys on the Rocky Prairie NAP, although the intermittent nature of these detections suggests it must be part of a larger metapopulation that occurs across nearby areas that have not been accessible for surveys. No known development poses a threat to the NAP, but any future conversion of the surrounding area to incompatible land use would likely hinder the recovery of this subspecies. The Yelm pocket gophers on Tenalquot prairie (which is owned in large part by JBLM) and Scatter Creek Wildlife Area are also secure from such residential and commercial development, but the Yelm pocket gopher habitat on Rock Prairie north of Old Highway 99 is in an area that is likely to be developed soon, which may negatively affect any local populations in the vicinity. Loss or Curtailment of Natural Disturbance Processes: The suppression and loss of ecological

disturbance regimes across vast portions of the landscape, such as fire, has resulted in altered vegetation structure in the prairies and meadows and has facilitated invasion by native and nonnative woody vegetation, rendering habitat unusable for the four Thurston/Pierce subspecies of *Mazama* pocket gopher. The basic ecological processes that maintain prairies and meadows have disappeared from, or have been altered on, all but a few protected and managed sites. Historically, the prairies and meadows of the south Puget Sound region are thought to have been actively maintained by native peoples, who lived here for at least 10,000 years before the arrival of Euro-American settlers (Boyd 1986; Christy and Alverson 2011, p. 93). Frequent burning reduced the encroachment and spread of shrubs and trees (Boyd 1986; Chappell and Kagan 2001, p. 42), favoring open grasslands with a variety of native plants and animals. Following Euro-American settlement of the region in the mid-19th century, fire was actively suppressed on grasslands, allowing encroachment by woody vegetation into the remaining prairie habitat and oak woodlands (Agee 1993, p. 360; Altman et al. 2001, p. 262; Boyd 1986; Franklin and Dyrness 1973, p. 122; Kruckeberg 1991, p. 287). Fires on the prairie create a mosaic of vegetation conditions, which serve to maintain native prairie plant communities. In some prairie patches fires will kill encroaching woody vegetation and reset succession back to bare ground, creating early successional vegetation conditions suitable for many native prairie species. Early successional forbs and grasses are favored by pocket gophers. The historical fire frequency on prairies has been estimated to be 3 to 5 years (Foster 2005, p. 8). On sites where regular fires occur, there is a high complement of native plants and fewer invasive species. These types of fires maintain the native short-statured plant communities favored by pocket gophers. The result of fire suppression has been the invasion of the prairies and oak woodlands by native and nonnative plant species (Dunn and Ewing 1997a, p. v; Tveten and Fonda 1999, p. 146), notably woody plants such as the native Douglas-fir (*Pseudotsuga menziesii*) and the nonnative Scot's broom. On tallgrass prairies in midwestern North America, fire suppression has led to degradation and the loss of native grasslands (Curtis 1959, pp. 296, 298; Panzer 2002, p. 1297). On northwestern prairies, fire suppression has allowed Douglas-fir to encroach on and outcompete native prairie vegetation for light, water, and nutrients (Stinson 2005, p. 7). This increase in woody vegetation and nonnative plant species has resulted in less available prairie habitat overall and habitat that is unsuitable for and avoided by many native prairie species, including pocket gophers (Olson 2011a, pp. 12, 16; Pearson et al. 2005, pp. 2, 27; Tveten and Fonda 1999, p. 155). Pocket gophers prefer early successional vegetation as forage. Woody plants shade out the forbs and grasses that pocket gophers prefer to eat, and high densities of woody plants make travel both below and above the ground difficult. In locations with poor forage, pocket gophers tend to have larger territories, which may be difficult or impossible to establish in densely forested areas. The probability of pocket gopher occupancy is much higher in areas with less than 10 percent woody vegetation cover (Olson 2011a, p. 16). On JBLM alone, over 16,000 acres (6,477 ha) of prairie has converted to Douglas-fir forest since the mid-19th century (Foster and Shaff 2003, p. 284). Where controlled burns or direct tree removal are not used as a management tool, this encroachment will continue to cause the loss of open grassland habitats for pocket gophers and is an ongoing threat to the species. Restoration in some of the south Puget Sound grasslands has resulted in temporary control of Scot's broom and other invasive plants through the careful and judicious use of herbicides, mowing, grazing, and fire. Fire has been used as a management tool to maintain native prairie composition and structure and is generally acknowledged to improve the health and composition of grassland habitat by providing a short-term nitrogen addition, which results in a fertilizer effect to vegetation, thus aiding grasses and forbs to sprout. Unintentional fires ignited by military training burn patches of prairie grasses and forbs on JBLM on an annual basis. These light ground fires create a mosaic of

conditions within the grassland, maintaining a low vegetative structure of native and nonnative plant composition, and patches of bare soil. Because of the topography of the landscape, fires create a patchy mosaic of areas that burn completely, some areas that do not burn, and areas where consumption of the vegetation is mixed in its effects to the habitat. One of the benefits of fire in grasslands is that it tends to kill regenerating conifers, and reduces the cover of nonnative shrubs such as Scot's broom, although Scot's broom seed stored in the soil can be stimulated by fire (Agee 1993, p. 367). Fire also improves conditions for many native bulb-forming plants, such as *Camassia* spp. (Agee and Dunwiddie 1984). On sites where regular fires occur, such as on JBLM, there is a high complement of native plants and fewer invasive species. These types of fires maintain the native, short-statured plant communities favored by pocket gophers. Management practices such as intentional burning and mowing require expertise in timing and technique to achieve desired results. If applied at the wrong season, frequency, or scale, fire and mowing can be detrimental to the restoration of native prairie species. Excessive and high-intensity burning can result in a lack of vegetation or encourage regrowth of nonnative grasses. Where such burning has occurred over a period of more than 50 years on the artillery ranges of JBLM, prairies are covered by nonnative forbs and grasses instead of native perennial bunchgrasses (Tveten and Fonda 1999, pp. 154, 155). Pocket gophers are not commonly found in areas colonized by Douglas-fir trees because pocket gophers require forbs and grasses of an early successional stage for food (Witmer et al. 1996a, p. 96). Pocket gophers observed on JBLM did not occur in areas with high cover of Scot's broom (Steinberg 1995, p. 26). A more recent study on JBLM also found that pocket gopher presence was negatively associated with Scot's broom (Olson 2011a, pp. 12, 13, 16). Some subspecies may disperse through forested areas or may temporarily establish territories on forest edges, but there is currently not enough data available to determine how common this behavior may be or which subspecies employ it. The four Thurston/Pierce subspecies of the *Mazama* pocket gopher occur on prairie-type habitats, many of which, if not actively managed to maintain vegetation in an early-successional state, have been invaded by shrubs and trees that either preclude pocket gophers or limit their ability to fully occupy the landscape. Typical management at civilian airports prevents woody vegetation from encroaching onto surrounding areas for flight safety reasons. Woody vegetation encroachment is therefore not a threat at civilian airports. Military Training: Pocket gopher populations occurring on JBLM are exposed to differing levels of training activities on the base. The Department of Defense's (DOD) proposed actions under their "Grow the Army" initiative include stationing 5,700 new soldiers, new combat service support units, a combat aviation brigade, facility demolition and construction to support the increased troop levels, and additional aviation, maneuver, and live fire training (75 FR 55313, September 10, 2010). The increased training activities will affect nearly all training areas at JBLM, resulting in an increased risk of accidental fires, and habitat destruction and degradation attributable to vehicle use in occupied areas, mounted and dismounted training, bivouac activities, and digging. Even though the training areas on the base are degraded, with implementation of agreed-upon conservation measures, these areas still provide habitat for the Roy Prairie and Yelm pocket gopher. JBLM's recently signed Endangered Species Management Plan (ESMP) for the *Mazama* pocket gopher will serve to minimize threats across the base by redirecting some training activities to areas outside of occupied habitat, designating areas where no vehicles are permitted, designating areas where vehicles will remain on roads only, and designating areas where no digging is allowed, among other conservation measures. JBLM has further committed to enhancing and expanding suitable habitat for the Roy Prairie and Yelm pocket gophers in "priority habitat" areas on base (areas that were proposed as critical habitat); enforcing restrictions on recreational use of occupied habitat by dog owners and horseback riders; and continuing to support the off-base

recovery of the four Thurston/Pierce subspecies of the *Mazama* pocket gopher. Several moderate- to large-sized areas supporting pocket gophers have been identified on JBLM. These areas are within the historical ranges of the Roy Prairie (Pierce County) and Yelm (Thurston County) pocket gophers. Their absence from some sites of what is presumed to have been formerly suitable habitat may be related to compaction of the soil due to years of mechanized vehicle training (Steinberg 1995, p. 36). Training infrastructure (e.g., roads, firing ranges, bunkers) also degrades pocket gopher habitat and may lead to reduced use of these areas by pocket gophers. For example, JBLM has plans to add a third rifle range on the south impact area where it overlaps with a densely occupied pocket gopher site. The area may be usable by pocket gophers when the project is completed; however, construction of the rifle range may result in removal of forage and direct mortality of pocket gophers through crushing of burrows (Stinson, in litt. 2011). Recent survey access to the center of the artillery impact area on 91st Division Prairie, where bombardment is presumably of the highest intensity, did detect some unspecified level of occupancy by the Roy Prairie pocket gopher (WDFW 2013b, enclosure 1, p. 6). This apparently suitable central portion of the 91st Division Prairie is subject to repeated and ongoing bombardment, which may create an ecological trap for dispersing juveniles. JBLM training areas have varying levels of use; some allow excavation and off-road vehicle use, while other areas have restrictions that limit off-road vehicle use. The ESMP specifically requires coordination between the JBLM Fish and Wildlife personnel and the JBLM entities responsible for training activities (e.g., Range Support, battalion commanders, and/or first field grade officers) to ensure all parties are aware of where occupied areas occur in relation to training activities, the effects of training, and the potential ramifications of habitat destruction or animal mortality. Since military training has the potential to directly or indirectly harm or harass pocket gophers, we conclude that these activities will negatively impact the Roy Prairie and Yelm pocket gophers. JBLM has committed to operational restrictions on portions of the base in order to avoid and minimize potential impacts to Roy Prairie and Yelm pocket gophers. Currently-occupied areas will be buffered from training activities, with an emphasis on occupied habitat in “priority habitat” areas. Regular surveys will be conducted with the goals of determining distribution, protecting pocket gophers and their habitat from disturbance or destruction, and determining population status. Where possible, JBLM will alleviate training pressure by transferring activities to unoccupied areas where encroaching forest has been removed. This strategy has the effect of both releasing large areas of land that were historically prairie and providing unoccupied areas where training is free of the risk of negatively impacting Roy Prairie or Yelm pocket gophers. While the Service fully supports the implementation of these impact minimization efforts and will continue to collaborate with DOD to address all aspects of training impacts on the species, not all adverse impacts on pocket gophers can be fully avoided. Military training continues to pose a threat to the Roy Prairie and Yelm subspecies at this time. No military training occurs in the ranges of the Olympia or Tenino subspecies of the *Mazama* pocket gopher (USFWS, 2016).

**Stressor:** Poor Connectivity Between Small and Isolated Populations (USFWS, 2016)

**Exposure:**

**Response:**

**Consequence:** Isolated genetics

**Narrative:** Most species’ populations fluctuate naturally, responding to various factors such as weather events, disease, and predation. Populations that are small, fragmented, or isolated by habitat loss or modification of naturally patchy habitat, and other human-related factors, are more vulnerable to extirpation by natural randomly occurring events, cumulative effects, and to genetic effect (collectively known as small population effects). These effects can include genetic

drift (loss of recessive alleles), founder effects (over time, an increasing percentage of the population inheriting a narrow range of traits), and genetic bottlenecks leading to increasingly lower genetic diversity, with consequent negative effects on evolutionary potential. To date, of the eight subspecies of Mazama pocket gopher in Washington, only the Olympic pocket gopher has been documented as having low genetic diversity (Welch and Kenagy 2008, p. 7), although the six other extant subspecies have local populations that are small, fragmented, and physically isolated from one another. The four Thurston/Pierce subspecies of the Mazama pocket gopher face threats from loss or fragmentation of habitat. Historically, pocket gophers probably persisted by continually recolonizing habitat patches after local extinctions. However, widespread development and conversion of habitat has resulted in widely separated populations, and intervening habitat corridors are now gone, with the effect of impeding or stopping much of the natural recolonization that historically occurred (Stinson 2005, p. 46). Although pocket gophers are not known to have low genetic diversity, small population sizes at most sites, coupled with disjunct and fragmented habitat, may contribute to further population declines. Little is known about the local or rangewide reproductive success of pocket gophers found in Washington State (USFWS, 2016).

**Stressor:** Predation and Pest Control (USFWS, 2016)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** Predation: Predation influences the distribution, abundance, and diversity of species in ecological communities. Generally, predation leads to changes in both the population size of the predator and that of the prey. In unfavorable environments, prey species are stressed or living at low population densities such that predation is likely to have negative effects on all prey species, thus lowering species richness. In addition, when a nonnative predator is introduced to the ecosystem, negative effects on the prey population may be higher than those from co-evolved native predators. The effect of predation may be magnified when populations are small, and the disproportionate effect of predation on declining populations has been shown to drive rare species even further towards extinction (Woodworth 1999, pp. 74, 75). Predation has an impact on populations of the four Thurston/Pierce subspecies of Mazama pocket gopher. Urbanization, particularly in the south Puget Sound region, has resulted in not only habitat loss, but also increased exposure to feral and domestic cats and dogs. Domestic cats are known to have serious impacts on small mammals and birds and have been implicated in the decline of several endangered and threatened mammals, including marsh rabbits (*Sylvilagus palustris*) in Florida and the salt-marsh harvest mouse (*Reithrodontomys raviventris*) in California (Ogan and Jurek 1997, p. 89). Domestic cats and dogs have been specifically identified as common predators of pocket gophers (Case and Jasch 1994, p. B-21; Henderson 1981, p. 233; Wight 1918, p. 21) and at least two pocket gopher locations were found as a result of house cats bringing home pocket gopher carcasses (WDFW 2001). Informal interviews with area biologists document multiple incidents of domestic pet predation on pocket gophers (Chan, in litt. 2013; Clouse, in litt. 2012 Skriletz 2013 in litt., Wood 2013 in litt.). There is also one recorded instance of a WDFW biologist being presented with a dead Mazama pocket gopher by a dog during an east Olympia, Washington, site visit in 2006 (Burke Museum 2012 McAllister 2013 in litt.). Some local populations of the pocket gopher occur in areas where people recreate with their dogs, bringing these potential predators into environments that may otherwise be relatively free of them, consequently increasing the risks to individual pocket gophers and populations that may be small and isolated. The four Thurston/Pierce subspecies of Mazama pocket gopher occur in rapidly

developing areas. Local populations that survive commercial and residential development (adjacent to and within habitat) are potentially vulnerable to extirpation by domestic and feral cats and dogs (Case and Jasch 1994, p. B-21; Henderson 1981, p. 233). As stated previously, predation is a natural part of the pocket gopher's life history; however, the effect of predation may be magnified when populations are small and habitat is fragmented. The disproportionate effect of additional predation on declining populations has been shown to drive rare species even further towards extinction (Woodworth 1999, pp. 74, 75). Predation, particularly from nonnative species, will likely continue to be a threat to the four Thurston/Pierce subspecies of the *Mazama* pocket gopher now and in the future. This is particularly likely where development abuts gopher habitat, resulting in increased numbers of cats and dogs in the vicinity, and in areas where people recreate with their dogs – particularly if dogs are off-leash and not prevented from harassing wildlife. In such areas, where local populations of pocket gophers are already small, this additional predation pressure (above natural levels of predation) is expected to further negatively impact population numbers. Pest Control: Pocket gophers are often considered a pest because they sometimes damage crops and seedling trees, and their mounds can create a nuisance. Several site locations were found as a result of trapping conducted on Christmas tree farms, a nursery, and in a livestock pasture (WDFW 2001). The type locality for the Cathlamet pocket gopher is on a commercial tree farm. Pocket gophers from Thurston County were used in a rodenticide experiment as recently as 1995 (Witmer et al. 1996a, p. 97). In Washington State it is currently illegal to trap or poison *Mazama* pocket gophers, or to trap or poison moles where they overlap with *Mazama* pocket gopher populations, but not all property owners are cognizant of these laws, nor are most citizens capable of differentiating between moles, pocket gophers, or the signs of their habitation (e.g., soil disturbance). In light of this, it is reasonable to believe that mole trapping or poisoning still has the potential to adversely affect pocket gopher populations. Local populations that survive commercial and residential development (adjacent to and within habitat) may be subsequently extirpated by trapping or poisoning. Lethal control by trapping or poisoning is most likely to be a threat to the four Thurston/Pierce subspecies where their ranges overlap residential properties (USFWS, 2016).

### **Recovery**

### **References**

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Designation of Critical Habitat for *Mazama* Pocket Gophers

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## SPECIES ACCOUNT: *Thomomys mazama yelmensis* (Yelm pocket gopher)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Threatened; Pacific Region (R1) (USFWS, 2016)

### **Physical Description**

Adult Mazama pocket gophers are reddish brown to black above, and the underparts are lead-colored with buff-colored tips. The lips, nose, and patches behind the ears are black; the wrists are white. Adults range from 7 to 9 inches (189 to 220 millimeters (mm)) in total length, with tails that range from 2 to 3 inches (45 to 85 mm)(Verts and Carraway 2000, p.2). Mazama pocket gophers are morphologically similar to other species of pocket gophers that exploit a subterranean existence. They are stocky and tubular in shape, with short necks, powerful limbs, long claws, and tiny ears and eyes. Their short, nearly hairless tails are highly sensitive and probably assist when navigating tunnels. The “pockets” are external, fur-lined cheek pouches on either side of the mouth that are used to transport nesting material and plant cuttings. Mazama pocket gophers reach reproductive age in the spring of the year after their birth and produce litters between spring and early summer. Litter size ranges from one to nine (Wight 1918, p. 14), with an average of five (Scheffer 1938, p. 222). They do not hibernate in winter; they remain active throughout the year (Case and Jasch 1994, p. B-20) (USFWS, 2016).

### **Taxonomy**

Although the species *Thomomys mazama*, or Mazama pocket gopher, includes numerous subspecies that are found in the States of Washington, Oregon, and California, only the subspecies found in the State of Washington have recently been considered for listing. The Mazama pocket gopher complex consists of 15 subspecies, eight of which occur only in Washington, five of which occur only in Oregon, one that occurs only in California, and one subspecies with a distribution that spans the boundary between Oregon and California (Hall 1981, p. 467). The first pocket gophers collected in western Washington were considered subspecies of the northern pocket gopher (*Thomomys talpoides*)(Goldman 1939), until 1960 when the complex of pocket gophers found in western Washington was determined to be more similar to the western pocket gopher (*T. mazama*)(Johnson and Benson 1960, p. 20). Eight western Washington subspecies of Mazama pocket gopher (*T. mazama*, ssp. *couchi*, *glacialis*, *louiei*, *melanops*, *pugetensis*, *tacomensis*, *tumuli*, and *yelmensis*) have been identified (Hall 1981, p. 467). *Thomomys mazama* is recognized as a valid species by the Integrated Taxonomic Information System (ITIS 2012). Although there have been suggestions that potential changes to the classification of some of these subspecies should be considered, we have no information to suggest that any of the presently recognized subspecies are the subject of serious dispute. We follow the subspecies designations of Verts and Carraway (2000), as this text represents the currently accepted taxonomy for the species *T. mazama*. Verts and Carraway (2000, p.1) recognize *T. m. glacialis*, *pugetensis*, *tumuli*, and *yelmensis* as separate subspecies (the Roy Prairie, Olympia, Tenino, and Yelm pocket gophers, respectively) based on morphological characteristics, distribution, and differences in number of chromosomes. Due to the close proximity of the four subspecies located in Thurston and Pierce Counties, and the fact that at least three of them occur in the same clade, we refer to these four subspecies (*T. m. glacialis*,

pugetensis, tumuli, and yelmensis) as “the four Thurston/Pierce subspecies” of the Mazama pocket gopher (USFWS, 2016).

**Current Range**

In Washington, Mazama pocket gophers are found west of the Cascade Mountain Range, in the Olympic Mountains and in the Puget Sound trough, with an additional single locality known from Wahkiakum County (Verts and Carraway 2000, p.3). Their populations are concentrated in well-drained friable soils often associated with glacial outwash. Yelm pocket gophers (*Thomomys mazama yelmensis*) were originally found on prairies in the area of Grand Mound, Vail, and Rochester (Dalquest and Scheffer 1944, p. 446). Surveys conducted during 1993 and 1994 found no pocket gophers near the towns of Vail or Rochester (Steinberg 1995, p. 28). More recent surveys have reported pocket gophers near Grand Mound, Littlerock, Rainier, Rochester, and Vail (Krippner 2011, p. 31), though WDFW biologists question the validity of the reports near Littlerock and Vail (WDFW 2013b, enclosure 1, p. 3). Soil series and soil series complexes in and around these areas that may support pocket gophers include Alderwood, Everett, Godfrey, Kapowsin, McKenna, Nisqually, Norma, Spana, Spanaway, Spanaway-Nisqually complex, and Yelm.(USFWS, 2016).

**Critical Habitat Designated**

Yes; 4/9/2014.

**Legal Description**

On April 9, 2014, the U.S. Fish and Wildlife Service (Service) designated critical habitat for three subspecies of the Mazama pocket gopher (the Olympia pocket gopher, *Thomomys mazama pugetensis*; the Tenino pocket gopher, *T. m. tumuli*; and the Yelm pocket gopher, *T. m. yelmensis*) under the Endangered Species Act of 1973, as amended (Act). In total, approximately 1,607 acres (650 hectares) in Thurston County, Washington, fall within the boundaries of the critical habitat designation for the Olympia, Tenino, and Yelm pocket gophers. The effect of this regulation was to designate critical habitat for the Olympia, Tenino, and Yelm subspecies of the Mazama pocket gopher found in Thurston County, Washington, under the Act.

**Critical Habitat Designation**

Yelm Pocket Gopher Critical Habitat—Tenalquot Prairie Subunit. This subunit consists of 289 ac (117 ha) and contains lands owned by one commercial landowner and The Nature Conservancy. This subunit is located northwest of the city of Rainier, Thurston County, Washington. As proposed, subunit 1–E (now the Tenalquot Prairie Subunit) included 1,505 ac (609 ha) of JBLM land, which has been exempted based on a completed ESMP. This 4(a)(3)(B)(i) exemption, based on this species-specific management plan, has been determined to provide a conservation benefit to the Yelm pocket gopher. The Tenalquot Prairie Subunit is occupied by the Yelm pocket gopher and contains the physical or biological features essential to the conservation of the species due to the underlying soil series (Spanaway), suitable forb and grass vegetation present onsite, and its large size. The physical or biological features in this subunit are threatened by: Loss of habitat through conversion to incompatible uses, such as development; the loss of natural disturbance processes and invasion by woody plants; inadequacy of existing regulatory mechanisms; and control as a pest species.

Yelm Pocket Gopher Critical Habitat—Rock Prairie Subunit. This subunit consists of 243 ac (98 ha) and contains lands owned by one private residential and commercial landowner. As proposed

(subunit 1–H), this subunit included 378 ac (153 ha) of private ranch land, which has been excluded under section 4(b)(2) of the Act (see Exclusions for details). The Rock Prairie Subunit is likely occupied by the Yelm pocket gopher and contains the physical or biological features essential to the conservation of the species due to the underlying soil series or soil series complex (Spanaway and Spanaway-Nisqually complex), suitable forb and grass vegetation present onsite, and its size. The physical or biological features in this subunit are threatened by: Loss of habitat through conversion to incompatible uses, such as development; the loss of natural disturbance processes and invasion by woody plants; predation; inadequacy of existing regulatory mechanisms; and control as a pest species. We additionally evaluated this area as if it were presently unoccupied by the Yelm pocket gopher, and have determined that it is nonetheless essential to the conservation of the species.

### **Primary Constituent Elements/Physical or Biological Features**

Critical habitat for the Yelm pocket gopher is designated in Thurston County, Washington. Within these areas, the primary constituent elements of the physical or biological features essential to the conservation of the Yelm pocket gopher consist of two components:

(i) Friable, loamy, and deep soils, some with relatively greater content of sand, gravel, or silt, all generally on slopes less than 15 percent in the following soil series or soils series complex: (A) Alderwood; (B) Cagey; (C) Everett; (D) Godfrey; (E) Indianola; (F) Kapowsin; (G) McKenna; (H) Nisqually; (I) Norma; (J) Spanaway; (K) Spanaway-Nisqually complex; and (L) Yelm.

(ii) Areas equal to or larger than 50 ac (20 ha) in size that provide for breeding, foraging, and dispersal activities, found in the soil series listed in paragraph (2)(i) of this entry that have: (A) Less than 10 percent woody vegetation cover; (B) Vegetative cover suitable for foraging by gophers. Pocket gophers' diets include a wide variety of plant material, including leafy vegetation, succulent roots, shoots, tubers, and grasses. Forbs and grasses that Mazama pocket gophers are known to eat include, but are not limited to: *Achillea millefolium* (common yarrow), *Agoseris* spp. (agoseris), *Cirsium* spp. (thistle), *Bromus* spp. (brome), *Camassia* spp. (camas), *Collomia linearis* (tiny trumpet), *Epilobium* spp. (several willowherb spp.), *Eriophyllum lanatum* (woolly sunflower), *Gayophytum diffusum* (groundsmoke), *Hypochaeris radicata* (hairy cat's ear), *Lathyrus* spp. (peavine), *Lupinus* spp. (lupine), *Microsteris gracilis* (slender phlox), *Penstemon* spp. (penstemon), *Perideridia gairdneri* (Gairdner's yampah), *Phacelia heterophylla* (varileaf phacelia), *Polygonum douglasii* (knotweed), *Potentilla* spp. (cinquefoil), *Pteridium aquilinum* (bracken fern), *Taraxacum officinale* (common dandelion), *Trifolium* spp. (clover), and *Viola* spp. (violet); and (C) Few, if any, barriers to dispersal. Barriers to dispersal may include, but are not limited to, forest edges, roads (paved and unpaved), abrupt elevation changes, Scot's broom thickets, highly cultivated lawns, inhospitable soil types or substrates, development and buildings, slopes greater than 35 percent, and open water.

### **Special Management Considerations or Protections**

Critical habitat does not include manmade structures (such as buildings, aqueducts, runways, roads, and other paved areas) and the land on which they are located existing within the legal boundaries on May 9, 2014.

The physical or biological features essential to the conservation of each subspecies may require special management considerations or protection to restore, protect, and maintain the essential features found there.

The physical or biological features essential to the conservation of the four Thurston/Pierce subspecies of the Mazama pocket gopher may require special management considerations or protection to control or prevent the establishment of invasive woody plants, which create shade and compete for light, food and nutrients otherwise utilized by the forb, bulb, and grass species that the gophers require for forage. Management may be implemented using hand tools or mechanical methods, prescribed fire, and the judicious use of herbicides. Although several management techniques are being implemented on public lands, we may need to improve our outreach to educate private landowners on controlling their pets and appropriately managing grazing on their properties, as well as to developing incentives for landowners who agree to conserve habitat. Incentives would create protected areas, through agreements or acquisitions. These would include corridors between existing protected habitat areas that may require management, enhancement actions, and long-term maintenance.

### ***Life History***

#### **Feeding Narrative**

Adult: Pocket gophers are generalist herbivores and their diet includes a wide variety of plant material, including leafy vegetation, succulent roots, shoots, and tubers. In natural settings pocket gophers play a key ecological role by aerating soils, activating the seed bank, and stimulating plant growth, though they can be considered pests in agricultural systems. In prairie and meadow ecosystems, pocket gopher activity plays an important role in maintaining species richness and diversity. Foraging primarily takes place below the surface of the soil, where pocket gophers snip off roots of plants before occasionally pulling the whole plant below ground to eat or store in caches. If above-ground foraging occurs, it's usually within a few feet of an opening and forage plants are quickly cut into small pieces and carried back to the nest or cache (Wight 1918, p. 12). Any water they need is obtained from their food (Gettinger 1984, pp. 749-750; Wight 1918, p. 13). The probability of pocket gopher occupancy is much higher in areas with less than 10 percent woody vegetation cover (Olson 2011a, p. 16), presumably because such vegetation will shade out the forbs, bulbs, and grasses that pocket gophers prefer to eat, and high densities of woody plants make travel both below and above the ground difficult (USFWS, 2016).

#### **Reproduction Narrative**

Adult: Pocket gophers reach sexual maturity during the spring of the year following their birth, and generally produce one litter per year (Case and Jasch 1994, p. B-20), though timing of sexual maturity has been shown to vary with habitat quality (Patton and Brylski 1987, p. 502; Patton and Smith 1990, p. 76). Gestation lasts approximately 18 days (Andersen 1978, p. 421; Schramm 1961, p. 169). Young are born in the spring to early summer (Wight 1918, p. 13), and are reared by the female. Aside from the breeding season, males and females remain segregated in their own tunnel systems. There are 1-9 pups per litter (averaging 5), born without hair, pockets, or teeth, and they must be kept warm by the mother or "packed" in dried vegetation (Case and Jasch 1994, p. B-20; Wight 1918, p. 14). Juvenile pelage starts growing in at just over a week (Andersen 1978, p. 420). The young eat vegetation in the nest within three weeks of birth, with eyes and ears opening and pockets developing at about a month (Andersen 1978, p. 420; Wight 1918, p. 14). At six weeks they are weaned, fighting with siblings, and nearly ready to disperse (Andersen 1978, p. 420; Wight 1918, p. 15), which usually occurs at about two months of age (Stinson 2005, p. 26). They attain their adult weight between four and five months of age

(Andersen 1978, pp. 419, 421). Most pocket gophers live only a year or two, with few living to three or four years of age (Hansen 1962, pp. 152, 153; Livezey and Verts 1979, p. 39) (USFWS, 2016).

### **Spatial Arrangements of the Population**

Adult: Clumped (USFWS, 2016)

### **Habitat Narrative**

Adult: The Mazama pocket gopher (pocket gopher) is associated with glacial outwash prairies in western Washington, an ecosystem of conservation concern (Hartway and Steinberg 1997, p. 1), as well as alpine and subalpine meadows and other meadow-like openings at lower elevations. Steinberg and Heller (1997, p. 46) found that pocket gophers are even more patchily distributed than are prairies, as there are some seemingly high quality prairies within the species' range that lack pocket gophers; e.g., Mima Mounds Natural Area Preserve (NAP), and 13th Division Prairie on Joint Base Lewis-McChord (JBLM). Pocket gopher distribution is affected by the rock content of soils, drainage, forage availability, and climate (Case and Jasch 1994, p. B-21; Hafner et al. 1998, p. 279; Reichman 2007, pp. 273-274; Steinberg and Heller 1997, p. 45; Stinson 2005, p. 31; WDFW 2009). Prairie and meadow habitats used by pocket gophers have a naturally patchy distribution. In their prairie habitats, there is an even patchier distribution of soil rockiness which may further restrict the total area that pocket gophers can utilize (Steinberg and Heller 1997, p. 45; WDFW 2009). We assume that meadow soils have a similarly patchy distribution of rockiness, though the soil surveys to support this are, at this time, incomplete. In western Washington, pocket gophers currently occupy the following soils series: Alderwood, Cagey, Carstairs, Everett, Everett-Spanaway complex, Everett-Spanaway-Spana complex, Godfrey, Grove, Indianola, Kapowsin, McKenna, Murnen, Nisqually, Norma, Shelton, Spana, Spana-Spanaway-Nisqually complex, Spanaway, Spanaway-Nisqually complex, and Yelm. No soil survey information is currently available for occupied sites in the Olympic National Park, so the soils occupied there are unknown. We purposely avoid using specific map unit names, because we know that there are imperfections in soil mapping. Maps are based on the technology, standards, and tools available at the time soil surveys were conducted, sometimes up to 50 years ago. We recognize that soil survey boundaries may be adjusted in the future, and that soil series names may be added or removed to soil survey maps and databases. As a result, the overlap of pocket gopher locations with soil series names may be different in the future. The soils information presented here is based on best scientific data available at the time of listing. We also recognize that some of these soil series or soil series complexes are not typically either deep or well-drained. For a variety of reasons, mapped soil types may or may not have all of the characteristics described by the U.S. Department of Agriculture, Natural Resources Conservation Service, and the actual soils that occur on sites may have characteristics that make them more or less habitable by pocket gophers. These reasons may include: map boundary or transcription errors, map projection errors or differences, map identification or typing errors, soil or hydrological manipulations that have occurred since mapping took place, and small-scale inclusions that are different from the mapped soil. Because soils are mapped at large scales, mapped soils may not identify smaller inclusions. Any of the soil series or soil series complexes listed above could potentially be suitable for the four Thurston/Pierce subspecies of the Mazama pocket gopher. And, the four Thurston/Pierce subspecies of the Mazama pocket gopher may also inhabit soil series not included in the above list. Although some soils are sandier, more gravelly, or may have more or less silt than described, most all soils used by pocket gophers are friable (easily pulverized or crumbled), loamy, and deep, and generally have

slopes less than 15 percent. There have been reports of pocket gophers (subspecies unknown) occurring on other types of soils, on managed forest lands in Capitol State Forest (owned by the Washington State Department of Natural Resources, WDNR) and Vail Forest (owned by Weyerhaeuser) in Thurston County. These were subsequently determined to be moles (*Scapanus* spp.), based on trapping conducted in these areas by the Washington State Department of Fish and Wildlife (WDFW) during 2012 (Thompson, pers. comm. 2012b). A study of the relationship between soil rockiness and pocket gopher distribution revealed a strong negative correlation between the proportion of medium-sized rocks in the soil, and the presence of pocket gophers (eight of nine prairies sampled); medium sized rocks were considered greater than 0.5 inch (12.7 mm), but less than 2 inches (50.8 mm) in diameter (Steinberg 1996, p. 32). In observations of pocket gopher distribution on JBLM, pocket gophers did not occur in areas with a high percentage of Scot's broom cover (*Cytisus scoparius*), or where mole populations were particularly dense (Steinberg 1995, p. 26). A more recent study on JBLM also found that pocket gopher presence was negatively associated with Scot's broom; however, the researcher found no relationship between pocket gopher presence and mole density (Olson 2011a, pp. 12, 13). Pocket gopher burrows consist of a series of main runways, off which lateral tunnels lead to the surface of the ground (Wight 1918, p. 7). Pocket gophers dig their burrows using their sharp teeth and claws and then push the soil out through the lateral tunnels (Case and Jasch 1994, p. B-20; Wight 1918, p. 8). Nests containing dried vegetation are generally located near the center of each pocket gopher's home tunnel system (Wight 1918, p. 10). Food caches and store piles are usually placed near the nest, and excrement is piled into blind tunnels or loop tunnels, and then covered with dirt, leaving the nest and main runways clean (Wight 1918, p. 11). (USFWS, 2016).

***Dispersal/Migration*****Motility/Mobility**

Adult: High (USFWS, 2016)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory (USFWS, 2016)

**Dispersal**

Adult: Low (USFWS, 2016)

**Immigration/Emigration**

Adult: Unlikely (USFWS, 2016)

**Dispersal/Migration Narrative**

Adult: Pocket gophers have limited dispersal capabilities (Williams and Baker 1976, p. 303). Mazama pocket gophers are smaller in size than other sympatric or peripatric *Thomomys* species (Verts and Carraway 2000, p. 1). Both dispersal distance and home range size are therefore likely to be smaller than for other *Thomomys* species. Dispersal distances may vary based on surface or soil conditions and size of the animal. For other, larger, *Thomomys* species, dispersal distances average about 131 feet (40 meters) (Barnes Jr. 1973, pp. 168, 169; Daly and Patton 1990, pp. 1286, 1288; Williams and Baker 1976, p. 306). Initial results from research being conducted on JBLM indicate that juvenile pocket gophers usually make movements from 13.1 to 32.8 feet (4-10 meters), though these may not be dispersal movements. One juvenile

made a distinct dispersal movement of 525 feet (160 meters) in a single day (Olson 2012, p. 5). Suitable dispersal habitat is free of barriers to movement, and may need to contain foraging habitat if an animal is required to make a long-distance dispersal movement. Potential barriers include, but are not limited to, forest edges, roads (paved and unpaved), abrupt elevation changes, Scot's broom thickets (Olson 2012, p. 3), highly cultivated lawns, inhospitable soil types or substrates (Olson 2008, p. 4), development and buildings, slopes greater than 35 percent, and open water. Barriers may be permeable, meaning that they impede movement from place to place without completely blocking it, or they may be impermeable, meaning they cannot be crossed. Permeable barriers, as well as lower quality dispersal habitats, may present a risk of mortality for animals that use them (e.g., open areas where predation risk is increased, or a paved area where vehicular mortality is high). The WDFW conducted a study to determine dispersal distances of juvenile pocket gophers on JBLM. Twenty-eight juveniles were radio-collared and tracked for 17 to 56 days, with all but three animals tracked for more than 30 days. Of these, only nine gophers moved more than 32.8 feet (10 meters), and 10 gophers were never found more than 13.1 feet (4 meters) from any previous location (Olson 2012, p. 5). Only one animal dispersed what would be considered a larger distance, moving 525 feet (160 meters) in a single day.

### ***Population Information and Trends***

#### **Population Trends:**

Decreasing (USFWS, 2016)

#### **Population Narrative:**

There are few data on historical or current population sizes of *Mazama* pocket gopher (pocket gopher) populations in Washington, although several local populations and one subspecies are believed to be extinct. Knowledge of the past status of the pocket gopher is limited to distributional information. Recent surveys have focused on determining current distribution, primarily in response to development applications. In addition, in 2012, WDFW initiated a five county-wide distribution survey. Because the object of all of these surveys has mainly been presence/absence only, total population numbers for each subspecies are unknown. And, the precise boundaries of each subspecies' range are not currently known. Local population estimates have been reported but are based on using apparent gopher mounds to delineate the number of territories, a method that has not been validated (Stinson 2005, pp. 40, 41). Olson (2011a, p. 2) evaluated this methodology on pocket gopher populations at the Olympia Airport and Wolf Haven International. Although there was a positive relationship between the number of mounds and number of pocket gophers, the relationship varies spatially, temporally, and demographically (Olson 2011a, pp. 2, 39). Based on the results of Olson's 2011 study, we believe past population estimates (Stinson 2005) may have been too high. As there is no generally accepted standard survey protocol to determine population size for pocket gophers, it is not currently possible to obtain an estimate of subspecies population sizes or trends. Overall habitat availability has declined, however, and habitat has a finite ability to support pocket gophers. For these reasons, the Service concludes that the overall population trend of each of the four Thurston/Pierce subspecies of the *Mazama* pocket gopher is negative. Increased survey effort since 2007 has resulted in the identification of numerous additional occupied sites located on private lands, especially in Thurston County (WDFW 2013a). Some of these new detections are adjacent to other known occupied sites, such as the population at the Olympia Airport. The full extent of these smaller discontinuous sites is currently unknown, and no research has been

done to determine whether or not these aggregations are “stepping stone” sites that may facilitate dispersal into nearby unoccupied suitable habitat, or if they are population sinks (sites that do not add to the overall population through recruitment). Others of these additional occupied sites are separate locations, seemingly unassociated (physically) with known populations (Tirhi, in litt. 2008). The largest known expanse of areas occupied by any subspecies in Washington occur on JBLM (Roy Prairie and Yelm pocket gophers), and at the Olympia and Shelton airports (Olympia and Shelton pocket gophers, respectively). A translocated population occurs on Wolf Haven International’s land near Tenino, Washington. Between 2005 and 2008, over 200 gophers from a variety of areas in Thurston County (some from around Olympia Airport (Olympia pocket gopher, *T. m. pugetensis*)) and some from near the intersection of Rich Road and Yelm Highway (assumed to be Olympia pocket gophers) were released into the 38 acres (15 ha) mounded prairie site. Based on the best available information, we do not believe the property previously supported pocket gophers. Today pocket gophers continue to occupy the site (Tirhi, in litt. 2011); however, current population estimates are not available. Another site, West Rocky Prairie Wildlife Area, has received a total of 560 translocated pocket gophers (*T. m. pugetensis*) from the Olympia Airport between 2009 and 2011. Initial translocation efforts were unsuccessful; a majority of the pocket gophers died within three days due to predation (Olson 2009, p. 3). Modified release techniques used in 2010 and 2011 resulted in improved survival rates (Olson 2011b, p. 4). It is too soon to know if the population will become self-sustaining, or if additional translocations of gophers will be necessary.

### ***Threats and Stressors***

**Stressor:** Destruction, Modification, or Curtailment of Habitat and Range (USFWS, 2016)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** The primary long term threats to the pocket gopher are the loss, conversion, and degradation of habitat, particularly to urban development, successional changes to grassland habitat, and the spread of invasive plants. The threats also include increased predation pressure, which is closely linked to habitat degradation. The prairies of south Puget Sound are one of the rarest ecosystems in the United States (Dunn and Ewing 1997b, p. v; Noss et al. 1995, p. I-2). Dramatic changes have occurred on the landscape over the last 150 years, including a 90 to 95 percent reduction in the extent of the prairie ecosystem. In the south Puget Sound region, where most of western Washington’s prairies historically occurred, less than 10 percent of the original prairie persists, and only three percent remains dominated by native vegetation (Crawford and Hall 1997, pp. 13, 14). Development: Native prairies and grasslands have been severely reduced throughout the range of the four Thurston/Pierce subspecies of *Mazama* pocket gopher, especially as a result of conversion to residential and commercial development and agriculture. Prairie habitat continues to be lost, particularly to residential development (Stinson 2005, p. 70), by removal and fragmentation of native vegetation, and the excavation, and/or heavy equipment-caused compaction of surfaces and conversion to non-habitat (e.g., buildings, pavement, other infrastructure), rendering soils unsuitable for burrowing. Residential development is associated with increased infrastructure, such as new road construction, which is one of the primary causes of landscape fragmentation (Watts et al. 2007, p. 736). Activities that accompany low-density development are correlated with decreased levels of biodiversity, mortality to wildlife, and facilitated introduction of nonnative invasive species (Trombulak and Frissell 2001; Watts et al. 2007, p. 736). In the south Puget Sound lowlands, the glacial outwash

soils and gravels underlying the prairies are deep and valued for use in construction and road building, which leads to their degradation and destruction. In the south Puget Sound, Nisqually loamy soils appear to support high densities of pocket gophers (Stinson, in litt. 2010a Olson 2008, p. 6), the vast majority of which occur in developed areas of Thurston County, or within the Urban Growth Areas for the cities of Olympia, Tumwater, and Lacey (WDFW 2009), where future development is most likely to occur. Where pocket gopher populations presumably extended across an undeveloped expanse of open prairie (Dalquest and Scheffer 1942, pp. 95, 96), areas currently occupied by the four Thurston/Pierce subspecies of the *Mazama* pocket gopher are now isolated to small fragmented patches due to development and conversion of suitable habitat to incompatible uses. The presumed extinction of the Tacoma pocket gopher is likely linked directly to residential and commercial development, which has replaced nearly all pocket gopher habitats in the historical range of the subspecies (Stinson 2005, pp. 18, 34, 46). One of the historical Tacoma pocket gopher sites was converted to a large gravel pit and golf course (Steinberg 1996, pp. 24, 27; Stinson 2005, pp. 47, 120). In addition, two gravel pits are now operating on part of the site recognized as the type locality for the Roy Prairie pocket gopher (Stinson 2005, p. 42), and another is in operation near Tenino (Stinson, in litt. 2010b) in the vicinity of the type locality for the Tenino pocket gopher. Multiple pocket gopher sites in Pierce and Thurston Counties may be, or have been, lost to gravel pit development, golf course development, or residential and commercial development (Stinson, in litt. 2005; Stinson 2005, pp. 26, 42; Stinson, in litt. 2010b). Multiple prairies that used to contain uninterrupted expanses of prairie habitat suitable for pocket gophers within the range of the four Thurston/Pierce subspecies have been developed to cities, neighborhoods, agricultural lands, or military bases, and/or negatively impacted by such development, including Baker Prairie, Bush Prairie, Chambers Prairie, Frost Prairie, Grand Mound Prairie, Little Chambers Prairie, Marion Prairie, Roy Prairie, Ruth Prairie, Woods Prairie, Violet Prairie, and Yelm Prairie. Some of these prairie areas still contain smaller areas that support pocket gophers, and some appear to no longer support pocket gophers at all (WDFW 2012). Where their properties coincide with pocket gopher occupancy, many private lands developers and landowners in Thurston County have agreed to create set-asides or agree to other mitigation activities in order to obtain development permits from the County (Tirhi, in litt. 2008). However, it is unknown if any pocket gophers will remain on these sites due to the small size of the set-asides, extensive grading in some areas adjacent to set-asides, lack of dedicated funding for enforcement or monitoring of set-aside maintenance (Thurston County Long Range Planning and Resource Stewardship, in litt. 2011, p. 2), and lack of control of predation by domestic or feral cats and dogs. In addition, some landowners have received variances from Thurston County that allowed development to occur without a requirement to set aside areas for pocket gophers. A population of Olympia pocket gophers is located at and around the Port of Olympia's Olympia Airport, which is sited on the historical Bush Prairie. Gophers on Bush Prairie are currently vulnerable to negative impacts from proposed future development by the Port of Olympia and ongoing development by adjacent landowners. The Port of Olympia has plans to develop large portions of the existing grassland that likely supports the largest population of the Olympia pocket gopher in Washington (Stinson 2007, in litt.; Port of Olympia and WDFW 2008, p.1; Port of Olympia 2012). The Olympia Airport is realigning the airport runway, which is in known occupied habitat. They continue to work with the Service and WDFW on mitigating airport expansion activities that may negatively impact gophers (Tirhi, in litt. 2010). The Olympia pocket gopher has a population at the Olympia Airport that spans several hundred acres, and there are two translocated populations: one at West Rocky Prairie Wildlife Area (some individuals from the Olympia Airport) and one at Wolf Haven (individuals from the Olympia Airport and some from near the intersection of Rich Road and Yelm

Highway). The population centered on the Olympia Airport could be negatively impacted by plans for development both on and off the airport, while the two translocated populations are currently secure from intense commercial and residential development pressures as they occur on conserved lands. The Roy Prairie pocket gopher is known to occur across a large expanse of prairie on JBLM, which is currently secure from the threat of development. The Tenino pocket gopher has a single known population, which has been detected during surveys on the Rocky Prairie NAP, although the intermittent nature of these detections suggests it must be part of a larger metapopulation that occurs across nearby areas that have not been accessible for surveys. No known development poses a threat to the NAP, but any future conversion of the surrounding area to incompatible land use would likely hinder the recovery of this subspecies. The Yelm pocket gophers on Tenalquot prairie (which is owned in large part by JBLM) and Scatter Creek Wildlife Area are also secure from such residential and commercial development, but the Yelm pocket gopher habitat on Rock Prairie north of Old Highway 99 is in an area that is likely to be developed soon, which may negatively affect any local populations in the vicinity.

**Loss or Curtailment of Natural Disturbance Processes:** The suppression and loss of ecological disturbance regimes across vast portions of the landscape, such as fire, has resulted in altered vegetation structure in the prairies and meadows and has facilitated invasion by native and nonnative woody vegetation, rendering habitat unusable for the four Thurston/Pierce subspecies of *Mazama* pocket gopher. The basic ecological processes that maintain prairies and meadows have disappeared from, or have been altered on, all but a few protected and managed sites. Historically, the prairies and meadows of the south Puget Sound region are thought to have been actively maintained by native peoples, who lived here for at least 10,000 years before the arrival of Euro-American settlers (Boyd 1986; Christy and Alverson 2011, p. 93). Frequent burning reduced the encroachment and spread of shrubs and trees (Boyd 1986; Chappell and Kagan 2001, p. 42), favoring open grasslands with a variety of native plants and animals. Following Euro-American settlement of the region in the mid-19th century, fire was actively suppressed on grasslands, allowing encroachment by woody vegetation into the remaining prairie habitat and oak woodlands (Agee 1993, p. 360; Altman et al. 2001, p. 262; Boyd 1986; Franklin and Dyrness 1973, p. 122; Kruckeberg 1991, p. 287). Fires on the prairie create a mosaic of vegetation conditions, which serve to maintain native prairie plant communities. In some prairie patches fires will kill encroaching woody vegetation and reset succession back to bare ground, creating early successional vegetation conditions suitable for many native prairie species. Early succession forbs and grasses are favored by pocket gophers. The historical fire frequency on prairies has been estimated to be 3 to 5 years (Foster 2005, p. 8). On sites where regular fires occur, there is a high complement of native plants and fewer invasive species. These types of fires maintain the native short-statured plant communities favored by pocket gophers. The result of fire suppression has been the invasion of the prairies and oak woodlands by native and nonnative plant species (Dunn and Ewing 1997a, p. v; Tveten and Fonda 1999, p. 146), notably woody plants such as the native Douglas-fir (*Pseudotsuga menziesii*) and the nonnative Scot's broom. On tallgrass prairies in midwestern North America, fire suppression has led to degradation and the loss of native grasslands (Curtis 1959, pp. 296, 298; Panzer 2002, p. 1297). On northwestern prairies, fire suppression has allowed Douglas-fir to encroach on and outcompete native prairie vegetation for light, water, and nutrients (Stinson 2005, p. 7). This increase in woody vegetation and nonnative plant species has resulted in less available prairie habitat overall and habitat that is unsuitable for and avoided by many native prairie species, including pocket gophers (Olson 2011a, pp. 12, 16; Pearson et al. 2005, pp. 2, 27; Tveten and Fonda 1999, p. 155). Pocket gophers prefer early successional vegetation as forage. Woody plants shade out the forbs and grasses that pocket gophers prefer to eat, and high densities of

woody plants make travel both below and above the ground difficult. In locations with poor forage, pocket gophers tend to have larger territories, which may be difficult or impossible to establish in densely forested areas. The probability of pocket gopher occupancy is much higher in areas with less than 10 percent woody vegetation cover (Olson 2011a, p. 16). On JBLM alone, over 16,000 acres (6,477 ha) of prairie has converted to Douglas-fir forest since the mid-19th century (Foster and Shaff 2003, p. 284). Where controlled burns or direct tree removal are not used as a management tool, this encroachment will continue to cause the loss of open grassland habitats for pocket gophers and is an ongoing threat to the species. Restoration in some of the south Puget Sound grasslands has resulted in temporary control of Scot's broom and other invasive plants through the careful and judicious use of herbicides, mowing, grazing, and fire. Fire has been used as a management tool to maintain native prairie composition and structure and is generally acknowledged to improve the health and composition of grassland habitat by providing a short-term nitrogen addition, which results in a fertilizer effect to vegetation, thus aiding grasses and forbs to sprout. Unintentional fires ignited by military training burn patches of prairie grasses and forbs on JBLM on an annual basis. These light ground fires create a mosaic of conditions within the grassland, maintaining a low vegetative structure of native and nonnative plant composition, and patches of bare soil. Because of the topography of the landscape, fires create a patchy mosaic of areas that burn completely, some areas that do not burn, and areas where consumption of the vegetation is mixed in its effects to the habitat. One of the benefits of fire in grasslands is that it tends to kill regenerating conifers, and reduces the cover of nonnative shrubs such as Scot's broom, although Scot's broom seed stored in the soil can be stimulated by fire (Agee 1993, p. 367). Fire also improves conditions for many native bulb-forming plants, such as *Camassia* spp. (Agee and Dunwiddie 1984). On sites where regular fires occur, such as on JBLM, there is a high complement of native plants and fewer invasive species. These types of fires maintain the native, short-statured plant communities favored by pocket gophers. Management practices such as intentional burning and mowing require expertise in timing and technique to achieve desired results. If applied at the wrong season, frequency, or scale, fire and mowing can be detrimental to the restoration of native prairie species. Excessive and high-intensity burning can result in a lack of vegetation or encourage regrowth of nonnative grasses. Where such burning has occurred over a period of more than 50 years on the artillery ranges of JBLM, prairies are covered by nonnative forbs and grasses instead of native perennial bunchgrasses (Tveten and Fonda 1999, pp. 154, 155). Pocket gophers are not commonly found in areas colonized by Douglas-fir trees because pocket gophers require forbs and grasses of an early successional stage for food (Witmer et al. 1996a, p. 96). Pocket gophers observed on JBLM did not occur in areas with high cover of Scot's broom (Steinberg 1995, p. 26). A more recent study on JBLM also found that pocket gopher presence was negatively associated with Scot's broom (Olson 2011a, pp. 12, 13, 16). Some subspecies may disperse through forested areas or may temporarily establish territories on forest edges, but there is currently not enough data available to determine how common this behavior may be or which subspecies employ it. The four Thurston/Pierce subspecies of the *Mazama* pocket gopher occur on prairie-type habitats, many of which, if not actively managed to maintain vegetation in an early-successional state, have been invaded by shrubs and trees that either preclude pocket gophers or limit their ability to fully occupy the landscape. Typical management at civilian airports prevents woody vegetation from encroaching onto surrounding areas for flight safety reasons. Woody vegetation encroachment is therefore not a threat at civilian airports. Military Training: Pocket gopher populations occurring on JBLM are exposed to differing levels of training activities on the base. The Department of Defense's (DOD) proposed actions under their "Grow the Army" initiative include stationing 5,700 new soldiers, new combat service support units, a combat aviation

brigade, facility demolition and construction to support the increased troop levels, and additional aviation, maneuver, and live fire training (75 FR 55313, September 10, 2010). The increased training activities will affect nearly all training areas at JBLM, resulting in an increased risk of accidental fires, and habitat destruction and degradation attributable to vehicle use in occupied areas, mounted and dismounted training, bivouac activities, and digging. Even though the training areas on the base are degraded, with implementation of agreed-upon conservation measures, these areas still provide habitat for the Roy Prairie and Yelm pocket gopher. JBLM's recently signed Endangered Species Management Plan (ESMP) for the Mazama pocket gopher will serve to minimize threats across the base by redirecting some training activities to areas outside of occupied habitat, designating areas where no vehicles are permitted, designating areas where vehicles will remain on roads only, and designating areas where no digging is allowed, among other conservation measures. JBLM has further committed to enhancing and expanding suitable habitat for the Roy Prairie and Yelm pocket gophers in "priority habitat" areas on base (areas that were proposed as critical habitat); enforcing restrictions on recreational use of occupied habitat by dog owners and horseback riders; and continuing to support the off-base recovery of the four Thurston/Pierce subspecies of the Mazama pocket gopher. Several moderate- to large-sized areas supporting pocket gophers have been identified on JBLM. These areas are within the historical ranges of the Roy Prairie (Pierce County) and Yelm (Thurston County) pocket gophers. Their absence from some sites of what is presumed to have been formerly suitable habitat may be related to compaction of the soil due to years of mechanized vehicle training (Steinberg 1995, p. 36). Training infrastructure (e.g., roads, firing ranges, bunkers) also degrades pocket gopher habitat and may lead to reduced use of these areas by pocket gophers. For example, JBLM has plans to add a third rifle range on the south impact area where it overlaps with a densely occupied pocket gopher site. The area may be usable by pocket gophers when the project is completed; however, construction of the rifle range may result in removal of forage and direct mortality of pocket gophers through crushing of burrows (Stinson, in litt. 2011). Recent survey access to the center of the artillery impact area on 91st Division Prairie, where bombardment is presumably of the highest intensity, did detect some unspecified level of occupancy by the Roy Prairie pocket gopher (WDFW 2013b, enclosure 1, p. 6). This apparently suitable central portion of the 91st Division Prairie is subject to repeated and ongoing bombardment, which may create an ecological trap for dispersing juveniles. JBLM training areas have varying levels of use; some allow excavation and off-road vehicle use, while other areas have restrictions that limit off-road vehicle use. The ESMP specifically requires coordination between the JBLM Fish and Wildlife personnel and the JBLM entities responsible for training activities (e.g., Range Support, battalion commanders, and/or first field grade officers) to ensure all parties are aware of where occupied areas occur in relation to training activities, the effects of training, and the potential ramifications of habitat destruction or animal mortality. Since military training has the potential to directly or indirectly harm or harass pocket gophers, we conclude that these activities will negatively impact the Roy Prairie and Yelm pocket gophers. JBLM has committed to operational restrictions on portions of the base in order to avoid and minimize potential impacts to Roy Prairie and Yelm pocket gophers. Currently-occupied areas will be buffered from training activities, with an emphasis on occupied habitat in "priority habitat" areas. Regular surveys will be conducted with the goals of determining distribution, protecting pocket gophers and their habitat from disturbance or destruction, and determining population status. Where possible, JBLM will alleviate training pressure by transferring activities to unoccupied areas where encroaching forest has been removed. This strategy has the effect of both releasing large areas of land that were historically prairie and providing unoccupied areas where training is free of the risk of negatively impacting Roy Prairie or Yelm pocket gophers.

While the Service fully supports the implementation of these impact minimization efforts and will continue to collaborate with DOD to address all aspects of training impacts on the species, not all adverse impacts on pocket gophers can be fully avoided. Military training continues to pose a threat to the Roy Prairie and Yelm subspecies at this time. No military training occurs in the ranges of the Olympia or Tenino subspecies of the Mazama pocket gopher (USFWS, 2016).

**Stressor:** Poor Connectivity Between Small and Isolated Populations (USFWS, 2016)

**Exposure:**

**Response:**

**Consequence:** Isolated genetics

**Narrative:** Most species' populations fluctuate naturally, responding to various factors such as weather events, disease, and predation. Populations that are small, fragmented, or isolated by habitat loss or modification of naturally patchy habitat, and other human-related factors, are more vulnerable to extirpation by natural randomly occurring events, cumulative effects, and to genetic effect (collectively known as small population effects). These effects can include genetic drift (loss of recessive alleles), founder effects (over time, an increasing percentage of the population inheriting a narrow range of traits), and genetic bottlenecks leading to increasingly lower genetic diversity, with consequent negative effects on evolutionary potential. To date, of the eight subspecies of Mazama pocket gopher in Washington, only the Olympic pocket gopher has been documented as having low genetic diversity (Welch and Kenagy 2008, p. 7), although the six other extant subspecies have local populations that are small, fragmented, and physically isolated from one another. The four Thurston/Pierce subspecies of the Mazama pocket gopher face threats from loss or fragmentation of habitat. Historically, pocket gophers probably persisted by continually recolonizing habitat patches after local extinctions. However, widespread development and conversion of habitat has resulted in widely separated populations, and intervening habitat corridors are now gone, with the effect of impeding or stopping much of the natural recolonization that historically occurred (Stinson 2005, p. 46). Although pocket gophers are not known to have low genetic diversity, small population sizes at most sites, coupled with disjunct and fragmented habitat, may contribute to further population declines. Little is known about the local or rangewide reproductive success of pocket gophers found in Washington State (USFWS, 2016).

**Stressor:** Predation and Pest Control (USFWS, 2016)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** Predation: Predation influences the distribution, abundance, and diversity of species in ecological communities. Generally, predation leads to changes in both the population size of the predator and that of the prey. In unfavorable environments, prey species are stressed or living at low population densities such that predation is likely to have negative effects on all prey species, thus lowering species richness. In addition, when a nonnative predator is introduced to the ecosystem, negative effects on the prey population may be higher than those from co-evolved native predators. The effect of predation may be magnified when populations are small, and the disproportionate effect of predation on declining populations has been shown to drive rare species even further towards extinction (Woodworth 1999, pp. 74, 75). Predation has an impact on populations of the four Thurston/Pierce subspecies of Mazama pocket gopher. Urbanization, particularly in the south Puget Sound region, has resulted in not only habitat loss, but also increased exposure to feral and domestic cats and dogs. Domestic cats are known to

have serious impacts on small mammals and birds and have been implicated in the decline of several endangered and threatened mammals, including marsh rabbits (*Sylvilagus palustris*) in Florida and the salt-marsh harvest mouse (*Reithrodontomys raviventris*) in California (Ogan and Jurek 1997, p. 89). Domestic cats and dogs have been specifically identified as common predators of pocket gophers (Case and Jasch 1994, p. B-21; Henderson 1981, p. 233; Wight 1918, p. 21) and at least two pocket gopher locations were found as a result of house cats bringing home pocket gopher carcasses (WDFW 2001). Informal interviews with area biologists document multiple incidents of domestic pet predation on pocket gophers (Chan, in litt. 2013; Clouse, in litt. 2012 Skriletz 2013 in litt., Wood 2013 in litt.). There is also one recorded instance of a WDFW biologist being presented with a dead *Mazama* pocket gopher by a dog during an east Olympia, Washington, site visit in 2006 (Burke Museum 2012 McAllister 2013 in litt.). Some local populations of the pocket gopher occur in areas where people recreate with their dogs, bringing these potential predators into environments that may otherwise be relatively free of them, consequently increasing the risks to individual pocket gophers and populations that may be small and isolated. The four Thurston/Pierce subspecies of *Mazama* pocket gopher occur in rapidly developing areas. Local populations that survive commercial and residential development (adjacent to and within habitat) are potentially vulnerable to extirpation by domestic and feral cats and dogs (Case and Jasch 1994, p. B-21; Henderson 1981, p. 233). As stated previously, predation is a natural part of the pocket gopher's life history; however, the effect of predation may be magnified when populations are small and habitat is fragmented. The disproportionate effect of additional predation on declining populations has been shown to drive rare species even further towards extinction (Woodworth 1999, pp. 74, 75). Predation, particularly from nonnative species, will likely continue to be a threat to the four Thurston/Pierce subspecies of the *Mazama* pocket gopher now and in the future. This is particularly likely where development abuts gopher habitat, resulting in increased numbers of cats and dogs in the vicinity, and in areas where people recreate with their dogs – particularly if dogs are off-leash and not prevented from harassing wildlife. In such areas, where local populations of pocket gophers are already small, this additional predation pressure (above natural levels of predation) is expected to further negatively impact population numbers. Pest Control: Pocket gophers are often considered a pest because they sometimes damage crops and seedling trees, and their mounds can create a nuisance. Several site locations were found as a result of trapping conducted on Christmas tree farms, a nursery, and in a livestock pasture (WDFW 2001). The type locality for the Cathlamet pocket gopher is on a commercial tree farm. Pocket gophers from Thurston County were used in a rodenticide experiment as recently as 1995 (Witmer et al. 1996a, p. 97). In Washington State it is currently illegal to trap or poison *Mazama* pocket gophers, or to trap or poison moles where they overlap with *Mazama* pocket gopher populations, but not all property owners are cognizant of these laws, nor are most citizens capable of differentiating between moles, pocket gophers, or the signs of their habitation (e.g., soil disturbance). In light of this, it is reasonable to believe that mole trapping or poisoning still has the potential to adversely affect pocket gopher populations. Local populations that survive commercial and residential development (adjacent to and within habitat) may be subsequently extirpated by trapping or poisoning. Lethal control by trapping or poisoning is most likely to be a threat to the four Thurston/Pierce subspecies where their ranges overlap residential properties (USFWS, 2016).

### **Recovery**

### **References**

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Designation of Critical Habitat for Mazama Pocket Gophers

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## **SPECIES ACCOUNT: *Urocitellus brunneus* (= *Spermophilus b.b.*) (Northern Idaho ground squirrel)**

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### ***Species Taxonomic and Listing Information***

**Commonly-used Acronym:** NIDGS

**Listing Status:** Threatened; 04/05/2000; Pacific Region (R1) (USFWS, 2016)

### **Physical Description**

The northern Idaho ground squirrel (*Urocitellus brunneus*) is smaller than most ground squirrels at about 9 inches. The northern Idaho ground squirrel's fur is dark reddish-gray (due to a mixture of black unbanded and yellowish-red banded guard hairs), with reddish-brown spots on its coat. It has a short, narrow tail, tan feet and ears, grey-brown throat and a creamy white eye ring (USFWS, 2016).

### **Taxonomy**

The first NIDGS specimens were collected in 1913 by L. E. Wyman, and described by A.H. Howell as a subspecies of the Washington ground squirrel (*Citellus townsendii brunneus*; Howell 1938, pp. 72-73). Subsequently, Howell (1938, pp. 72-73) reclassified the Idaho ground squirrel as a full species, *Citellus brunneus*. In 1949 (p. 300), Hershkovitz demonstrated that *Spermophilus* is the correct name for the genus. In 1991 (entire), Yensen determined that *Spermophilus brunneus* consisted of two taxonomically distinct subspecies based on morphology, pelage, and life history differences that also included biogeographical separation; the NIDGS and neighboring southern Idaho ground squirrel (SIDGS; *Spermophilus brunneus endemicus*; genus recognized as *Urocitellus* ; Helgen et al. 2009, p. 297). Yensen (1991, p. 583) suggested that the two subspecies were close to species-level separation, and subsequent genetic work has indicated that they could be validated as separate species (Gill and Yensen 1992, p. 155; Yensen and Sherman 1997, p. 1; Gavin et al. 1999, p. 163; Hoisington 2007, p. iii). In 2009, Helgen et al. (p. 297) revised the genus *Spermophilus* and placed the NIDGS into the genus *Urocitellus* . Given the NIDGS is threatened throughout its range, it is not considered a distinct population segment (USFWS, 2011).

### **Current Range**

This species is endemic to a five-county area of west-central Idaho (Yensen and Sherman 1997). The northern subspecies (*brunneus*) presently is known only from Valley and Adams counties at elevations of 1,150-1,550 meters; most populations are small and often isolated by several kilometers (Yensen 1991). The southern subspecies (*endemicus*) has a patchy distribution at lower elevations (670-975 meters) north of the Payette River in Gem, Payette, and Washington counties. The species is apparently extirpated in the area between the extant populations of the northern and southern subspecies (Yensen 1984, 1991, Yensen et al. 1991, Yensen and Sherman 1997).

### **Critical Habitat Designated**

Yes;

### ***Life History***

**Feeding Narrative**

Juvenile: Feeds on green vegetation, seeds.; Food Habits: Herbivore (Adult, Immature), Granivore (Adult, Immature) (NatureServe, 2015)

Adult: Feeds on green vegetation, seeds.; Food Habits: Herbivore (Adult, Immature), Granivore (Adult, Immature). Southern populations emerge in late January or early February and cease above-ground activity in late June or early July; northern populations are active above ground from late March or early April until late July or early August (Yensen 1991). Activity is constrained by time of snow melt and vegetation dessication. (NatureServe, 2015)

**Reproduction Narrative**

Adult: Mating occurs soon after spring emergence; males guard sexually receptive females from other males; after mating, female excludes male from female burrow; gestation lasts about 3 weeks; litter size is 2-10 (average around 6-7); young are weaned in 3 weeks (Yensen 1991, Spahr et al. 1991).; May be limited by competition from Columbian ground squirrel (Spahr et al. 1991). Badgers and prairie falcons are the primary predators.; (NatureServe, 2015). The northern Idaho ground squirrel emerges in late March or early April and remains active above ground until July or early August (Yensen 1991). Emergence during this period begins with adult males, followed by adult females, and then yearlings. The northern Idaho ground squirrel becomes reproductively active within the first 2 weeks of emergence (Yensen 1991). Females and males are sexually mature the first spring after birth. They produce one litter per year of between two and seven pups, depending on the fitness of the female. Females that survive the first winter live, on average, nearly twice as long as males (3.2 years for females and 1.7 years for males). Individual females have lived for as long as 8 years. Males normally die at a younger age due to behavior associated with reproductive activity. During the mating period, males move considerable distances in search of receptive females and often fight with other males for copulations, thereby exposing themselves to predation by raptors, such as prairie falcons (*Falco mexicanus*), goshawks (*Accipiter gentilis*), and red-tailed hawks (*Buteo jamaicensis*). Significantly more males die or disappear during the 2-week mating period than during the rest of the 12- to 14-week period of above-ground activity (Sherman and Yensen 1994). Seasonal torpor or hibernation generally occurs in early to mid-July for males and females, and late July to early August for juveniles (Yensen and Sherman 1997) (USFWS, 2003).

**Tolerance Ranges/Thresholds**

Adult: Moderate (inferred from NatureServe, 2015)

**Site Fidelity**

Adult: Moderate (inferred from NatureServe, 2015)

**Habitat Narrative**

Adult: Northern populations are associated with shallow rocky soils in xeric meadows surrounded by ponderosa pine and Douglas-fir forest; southern populations inhabit low rolling hills and valleys now dominated by annual grassland with relict big sagebrush and bunch grasses (Yensen et al. 1991, Yensen 1991). This squirrel may occur on slopes and rarely on ridges (Yensen 1984). It burrows extensively in shallow rocky soils, but nest burrows are located in adjacent areas with deeper (>1 meter) well-drained soils (Yensen et al. 1991). Grassland/herbaceous. Burrowing in or using soil (NatureServe, 2015). Moderate ecological

integrity of the community, site fidelity and tolerance ranges are inferred based on the specific habitat requirements of the species, relatively large geographic area it inhabits and the number of known populations.

### ***Dispersal/Migration***

#### **Motility/Mobility**

Adult: High (USFWS, 2011)

#### **Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory (Natureserve, 2015)

#### **Dispersal**

Adult: Moderate (USFWS, 2011)

#### **Dispersal/Migration Narrative**

Adult: Nonmigrant: Y; (NatureServe, 2015). Idaho ground squirrel dispersal corridors have been reduced or eliminated, further constricting the subspecies into smaller isolated areas (Yensen and Sherman 1997). Fire suppression has allowed conifers to invade once suitable meadow habitats, thereby shrinking the size of forb/grass meadows or closing grassy dispersal/migration corridors to nearby meadow sites. These changes have isolated the dry meadows with suitable shallow soils and preferred forage and burrow habitat where the northern Idaho ground squirrel finds refuge from the Columbian ground squirrel. Habitat fragmentation and reduced opportunities for dispersal among habitats prevents gene flow and results in considerable population differences (Yensen and Sherman 1997, Sherman and Runge 2002). The loss of dispersal corridors has caused some isolated population sites to become extirpated in recent years (Sherman and Yensen 1994; U.S. Fish and Wildlife Service 1996). Additionally, small populations at several remaining sites are likely to become extirpated (Sherman and Yensen 1994; Mangel and Tier 1994). The rate, timing, and age of dispersal of northern Idaho ground squirrels from their natal sites is unknown. However, in order to establish successfully functioning metapopulations, it is important to know how far northern Idaho ground squirrels disperse under a variety of conditions. Timing and age of dispersal is important for planning successful translocation, supplementation, and reintroduction programs. It may be possible to utilize information from surrogate ground squirrel species being studied (e.g., southern Idaho ground squirrels, Pauite ground squirrel, etc.) (USFWS, 2003). Data from studies of the southern Idaho ground squirrel (SIDGS; *Spermophilus brunneus endimicus*; genus recognized as *Urocitellus*1 ; Helgen et al. 2009, p. 297), which is classified as a candidate species (i.e. candidate for protection under the Endangered Species Act), indicates that dispersal is undertaken by young of the year midway through their active period (i.e. while they are above ground; Panek 2005, p. 39). While less is known regarding NIDGS dispersal timing, at one occupied location in 2011 it was determined that NIDGS pups were dispersing in mid July (Rautsaw in litt. 2011b, p. 13). Regarding dispersal distances, SIDGSs have been documented dispersing up to distances of 2.4 kilometers (km) (1.5 miles (mi); Panek 2005, p. 32). Caution should be used when comparing dispersal results from SIDGSs for NIDGSs given the different habitat requirements for each subspecies (NIDGSs are found in meadow/ forested habitats, while SIDGSs are found in shrub steppe habitats). These different habitats requirements may influence the dispersal distances for each subspecies (USFWS, 2011).

***Population Information and Trends*****Population Trends:**

Decreasing (NatureServe, 2015); Increasing (USFWS, 2011)

**Resiliency:**

Moderate (inferred from NatureServe, 2015 and USFWS, 2011)

**Representation:**

Moderate (inferred from NatureServe, 2015 and USFWS, 2011)

**Redundancy:**

Moderate (inferred from NatureServe, 2015 and USFWS, 2011)

**Number of Populations:**

21 - 80 (NatureServe, 2015)

**Population Size:**

2500 - 100,000 individuals (NatureServe, 2015)

**Population Narrative:**

A significant decline has occurred in area of occupancy, number of subpopulations, and population size (USFWS 2002, 2004). See information for subspecies *brunneus* and *endemicus*. Decline of 30-70%. Total adult population size appears to be at least several thousand individuals (Yensen 2001, USFWS 2002). See information for subspecies *brunneus* and *endemicus*. Based on locations mapped on a coarse scale (Yensen and Sherman 1997), this species occurs in at least few dozen distinct areas; these include at least a few hundred occupied sites. See information for subspecies *brunneus* and *endemicus*. (NatureServe, 2015). In 1985, the total NIDGS population was estimated to be 5,000 squirrels scattered among 18 known population sites (Yensen 1985, p. 29). In 2002, two years after listing, the population estimate for the NIDGS was 450 to 500 individuals (Haak 2002, p. 10). In 2010, NIDGSs occupied 56 sites, an increase of 34 sites compared to the 22 sites detected in 2002 (Evans Mack 2010a, p. ii). Modeled population results, combined with squirrels detected on surveys, estimate the minimum pre-pup population was 1,560 in 2010, down slightly from the 1,618 estimated in 2009 (Evans Mack 2010a, p. ii; Evans Mack in litt. 2010, p. 2; Evans Mack and Bond 2010, p. 6). The decrease in population from 2009 to 2010 is attributed to fewer sites surveyed in 2010 as opposed to a true population decrease (Evans Mack in litt. 2011b, p. 2). Overall, the 10-year NIDGS population trend is increasing while its distribution across the landscape continues to expand (Figure 1; Evans Mack in litt. 2011b, p. 2; Evans Mack 2010a, pp. 6, 10) (USFWS, 2011). Moderate resiliency, representation and redundancy are based on the number of use sites and their relatively wide ranging geography as well as the overall population size.

***Threats and Stressors***

**Stressor:** Meadow invasion (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat/increased predation

**Narrative:** Northern Idaho ground squirrels rely on meadow habitat connected within a matrix of ponderosa pine and/ or Douglas fir forested habitat. The primary threat to the NIDGS identified in the 2000 listing rule and 2003 Recovery Plan was, and appears to continue to be, meadow invasion by conifers (Rautsaw in litt. 2011b, p. 1; Evans Mack in litt. 2010, p. 5; USFWS 2009a, p. 2; USFWS 2003, p. 11; USFWS 2000a, p. 17779; Jensen and Sherman 1997, p. 3). Once open stands of conifers with an herbaceous understory have been replaced by dense stands of trees lacking an understory as a result of logging and fire suppression in post-settlement times (USFWS 2003, pp. 11-12; Burns and Zborowski 1996, entire; Crane and Fischer 1986 and Steele et al. 1986 in Jensen and Sherman 1997, p. 3). This has reduced the amount of suitable NIDGS habitat, while at the same time further isolating populations and reducing genetic exchange among populations. With limited connectivity for dispersal opportunities, small and isolated NIDGS populations are also likely more susceptible to the effects of predation (USFWS, 2011).

**Stressor:** Land use changes (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Development and habitat conversion are historical and ongoing threats to NIDGS populations, especially on private lands (Evans Mack in litt. 2010, p. 5; USFWS 2003, p. 11; USFWS 2000a, pp. 17781-17782). Half of the currently known sites occupied by NIDGSs occur on private land, comprising an estimated 439 ha (1,085 ac) of occupied habitat (Evans Mack in litt. 2010, p. 5). The land incorporating the entire Round Valley NIDGS metapopulation is presently for sale, and a subdivision and private home have been developed in Round Valley in the last 6 years (Evans Mack in litt. 2010, p. 5). In addition, Potlatch Forest Holdings Inc. is advertising private timber land for sale in the Mud Creek drainage along Price Valley and Mud Creek roads where there are known occupied NIDGS locations (Evans Mack in litt. 2010, p. 5). This conversion of once open space occupied by NIDGSs to housing developments on private land is a continuous and expanding threat to the species (Evans Mack in litt. 2010, p. 5) (USFWS, 2011).

**Stressor:** Motorized recreation (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** A threat to NIDGS habitat not discussed in the 2000 Final Listing Rule, but that has materialized since then, is off highway vehicle (OHV) use. Cross-country OHV use can detrimentally impact NIDGS habitat through soil compaction, removal of vegetation, and physical disturbance or harm to individuals (USFS 2007, pp. 3-183). While this threat has not been quantified, anecdotal evidence exists of NIDGS habit disturbance by OHVs in certain areas (Rautsaw in litt. 2011c, entire). While it's unlikely this threat is operating at the landscape level, isolated OHV cross-country use has the potential to negatively affect NIDGSs and their habitat through localized events, potentially threatening small and/ or isolated populations (USFWS, 2011).

**Stressor:** Recreational shooting (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** Illegal recreational shooting continues to be a threat to the NIDGS, though quantification of take remains unknown; therefore population effects are unclear (Evans Mack 2010a, p. 6). In 2009, an illegal shooting case was documented and brought to trial in Adams County, where the person charged pleaded guilty to illegally taking (shooting) a NIDGS (Evans Mack in litt. 2009, p. 1). In addition, NIDGSs are commonly mistaken for COGSs, which are still legal to shoot, both of which are often found occurring together in the same general vicinities. This potential confusion between the two species further increases the likelihood of continued illegal shooting of NIDGSs (USFWS, 2011).

**Stressor:** Scientific Collections and Translocations (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** While NIDGSs are actively monitored through live trapping, only 5 mortalities out of 2,490 trap events (<0.2%) have occurred in the past 8 years (Evans Mack in litt. 2010, p. 6). In 2005, a translocation attempt may have led to the loss of 9-13 NIDGSs (Evans Mack in litt. 2010, p. 6). Additional translocation attempts have not been carried out since then and concerns remain regarding the high mortality rate of translocated animals, low overall success, and diminished priority due to recent genetic findings (USFWS, 2011).

**Stressor:** Predation (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** At the time of listing, the threats to the NIDGS associated with Factor C include predation, especially at smaller and more isolated populations (USFWS 2000a, pp. 17782-17783). The state of knowledge on disease and predation has not changed significantly since listing or the completion of the 2003 Recovery Plan. While disease is not considered a threat, it is presently unknown if plague (*Yersinia pestis*) occurs within any NIDGS populations (Evans Mack in litt. 2010, p. 6). Fleas have been documented at one NIDGS population, which has undergone population increases and decreases, though it's unknown if fleas are the source of the population changes (Evans Mack in litt., 2010a, p. 6). Domestic dogs have recently been identified as a localized threat at two NIDGS sites on private land (Evans Mack in litt. 2010, p. 6). Additional interactions between domestic dogs, feral cats, and NIDGSs are likely to continue as once open private lands near occupied NIDGS habitat are converted to residential developments. In addition, the closure of Slaughter Campground by the PNF at the Lost Valley metapopulation site was primarily due to the negative effects domestic dogs were having on NIDGSs (Rautsaw in litt. 2011b, p. 32). Badgers continue to be a predation concern, primarily to small and isolated populations that are more susceptible to the effects of localized predation events. To reduce the threat of predation on NIDGS populations, limited mammalian predator control, primarily for badgers (*Taxidea taxus*), has taken place periodically from 2003-2009. While quantification of control actions are reported annually, its effectiveness at reducing predation to NIDGSs is unknown because it never has been measured (Evans Mack in litt. 2010, p. 6). Other predators to NIDGSs include raptors and weasels (*Mustela frenata*) (USFWS, 2011).

**Stressor:** Inadequacy of existing regulatory mechanisms (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Illegal Take or Possession Northern Idaho ground squirrels are a Federally threatened species, with illegal take regulated under Section 9(a)(1) of the Act. While hunting for several other species of ground squirrels in Idaho is unregulated by the State, the NIDGS is considered a protected non-game species under State Law for which it is illegal to take: no person shall take or possess those species of wildlife classified as Protected Nongame, or Threatened or Endangered at any time or in any manner, except as provided in Sections 36- 106(e) and 36- 1107, Idaho Code, by Commission rule, or IDAPA 13.01.10, ?Rules Governing the Importation, Possession, Release, Sale or Salvage of Wildlife,? Subsection 100.06.b (IDFG 2005, p. B-5). Even though it is illegal to shoot NIDGSs, illegal take continues to pose a threat to the species. In 2009, a person was charged and sentenced with the illegal killing of a NIDGS. The sentencing was minimal due to the lack of knowledge by the defendant regarding the presence of NIDGSs in the vicinity of the infraction, and as a result, additional signage has been erected at key locations within their range warning of the presence of a threatened species. Additional public outreach regarding the illegality of shooting NIDGSs is needed to further reduce this threat since people commonly mistake NIDGSs for COGSs, which are legal to shoot (USFWS, 2011). While regulatory mechanisms for protecting NIDGS habitat are lacking on Idaho State lands, at this time we do not possess the information linking this lack of regulatory mechanisms as a threat to the species. We recommend the MOA between IDFG and IDL be continued and appropriately applied on State lands. Conservation measures may need to be better developed to address crosscountry OHV use through occupied NIDGS habitat on State of Idaho lands. We encourage the IDL to take advantage of opportunities to enhance NIDGS habitat on State endowment lands while adhering to the Idaho Constitution mandate to secure the maximum long term financial return for the State of Idaho (USFWS, 2011).

**Stressor:** Private land development (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** As is discussed under Factor A, the development of occupied NIDGS habitat on private lands continues to be a threat to the species. Comprehensive plans for Adams and Valley Counties, where all of the known NIDGS occupied habitat occurs, contain goals of protecting wildlife and their habitats (Valley County 2010, pp. 11-12; Adams County 2006, p. 37). Even with these goals in place, private lands containing occupied NIDGS habitat continue to be developed in those Counties (see 2.3.2.1, Present or threatened destruction, modification or curtailment of its habitat or range). While IDFG continues to provide technical comments to various agencies, including local Counties regarding the effects of land use changes on NIDGSs and their habitat (Evans Mack 2010b, p. 12), the inadequacy of existing regulatory mechanisms regarding private land development continues to be a threat to the species (USFWS, 2011).

**Stressor:** Columbian Ground Squirrel Competition (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** In 2010, it was found that COGSs occurred at 24 sites occupied by NIDGSs (Evans Mack 2010a, p. 6). It's been noted by Evans Mack and Bond (2010, p. 7) that COGS expansion at certain NIDGS sites is likely a result of habitat treatments for the benefit of NIDGSs. As the PNF conducts habitat treatments for the benefit of NIDGSs, we expect COGSs to also favorably respond by

expanding their range into once unsuitable habitat. In addition, COGSs may displace NIDGSs in other parts of their range where habitat treatments have not occurred. Therefore the threat still exists for COGSs displacing NIDGSs from occupied habitat (USFWS, 2011).

**Stressor:** Forage Competition Between NIDGSs and Livestock (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** While potential forage competition between NIDGSs and livestock (cattle) was not identified as a threat factor at the time of listing, it was identified as a research priority (USFWS 2003, p. 24). Given most occupied NIDGS sites are also grazed by cattle, a pilot study to document the diets of NIDGSs and cattle at 2 occupied sites was conducted. In the recent preliminary study, diet comparison results indicate that there is low dietary overlap between NIDGSs and cattle, with NIDGS diets consisting of a higher proportion of forbs (herbaceous flowering plant) compared to a higher proportion of graminoids (grasses) in cattle diets (Yensen et al. 2010, entire). Further study of these results is needed to answer additional questions raised in this preliminary study (Yensen et al. 2010, p. 6) such as season of use by cattle/NIDGS overlap and intensity of grazing. Domestic sheep grazing also occurs within portions of the range of NIDGSs. Domestic sheep have been known to alter the vegetation cover components in sagebrush ecosystems (Mueggler 1950, entire; Laycock 1967, entire). Spring grazing by domestic sheep has been shown to lead to a reduction of perennial forbs and grasses, while fall domestic sheep grazing has been shown to be less detrimental to the perennial forb and grass vegetation component (Mueggler 1950, pp. 314-315; Laycock 1967, p. 213; Bork et al. 1998, p. 299). Both perennial forbs and grasses are important diet components for NIDGSs. Given the likely dietary overlap between domestic sheep and NIDGSs, there is concern that domestic sheep grazing may negatively affect NIDGS habitat (Rautsaw in litt. 2011b, p. 35). Additional information is needed regarding the timing and extent of overlap of domestic sheep grazing in occupied and suitable NIDGS habitat to determine the extent of this potential threat (USFWS, 2011).

**Stressor:** Roadway mortality (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:** Loss of individuals

**Narrative:** Mortality of NIDGSs from vehicles on roads has occurred near occupied sites on U.S. Forest Service and County roadways, and a U.S. highway, although total mortality has not been quantified (Evans Mack in litt. 2010, p. 7). Speed limits and timing restrictions have been identified as conservation measures, though they have not always been adhered to or implemented (Evans Mack in litt. 2010, p. 7). While vehicle induced NIDGS mortality is a potential threat, especially to smaller and isolated populations, additional study is needed to better quantify the amount of NIDGS mortality that occurs from vehicle collisions (USFWS, 2011).

**Stressor:** Idaho Department of Lands Land Exchange (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** In 2009, the Boy Scouts of America (BSA) approached the Service regarding entering into a safe harbor agreement, or similar agreement, related to a potential land swap between the BSA and IDL to establish a Boy Scott summer recreation camp (BSA in litt 2009, entire). The

section of IDL land the BSA had proposed to acquire is adjacent to the large Lost Valley Reservoir NIDGS population. In 2010, this colony's population was estimated at 154 individuals (Evans Mack 2010a, p. 15). The proposed human access route from the identified section of IDL land to the Reservoir would also cross through occupied NIDGS habitat, thereby greatly increasing human disturbance to NIDGSs. While the present status of this land swap is unknown, as it is currently proposed by the BSA it would constitute a threat to the relatively large and important Lost Valley Reservoir NIDGS population and impede NIDGS recovery (Womack in litt. 2010, entire) (USFWS, 2011).

**Stressor:** Small Populations and Naturally Occurring Events (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:** Extinction

**Narrative:** Due to the threats discussed in this 5-year review, along with the fact that small and isolated NIDGS populations remain throughout their range, the NIDGS still likely has little resilience to naturally occurring events (USFWS, 2011).

**Stressor:** Climate change (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:** Loss of habitat

**Narrative:** Predicted changes of climate could result in a wide-range of potential outcomes for NIDGSs and their habitat. The effects to the NIDGS in either the short or long-term in a focused geographic area cannot be reasonably discerned without a specific aspect of its ecology or physiology linked to a confidently predicted climate change variable (e.g., water temperature tolerance of fish, or early snowmelt reducing wolverine denning). Increasing temperatures and drought could affect fire frequency and intensity and the susceptibility of forest vegetation to disease. This rise in temperatures may also affect the timing of NIDGSs entering and exiting seasonal torpor in response to vegetative timing changes from climate change; or may cause a response by NIDGSs to move up in elevation as lower elevation habitats become less suitable. Additional information is needed to better determine the response of the NIDGS to a changing climate (USFWS, 2011).

## ***Recovery***

### **Delisting Criteria:**

Delisting may be considered when the following recovery criteria have been met: 1. Of the 17 potential metapopulations that have been identified within the probable historical distribution, there must be at least 10 metapopulations, each maintaining an average effective population size of greater than 500 individuals for 5 consecutive years (USFWS, 2003).

2. The area occupied by a minimum of 10 potential metapopulations must be protected. In order for an area to be deemed protected, it must be: (1) owned or managed by a government agency with appropriate management standards in place; (2) managed by a conservation organization that identifies maintenance of the subspecies as the primary objective for the area; or, (3) on private lands with a long-term conservation easement or covenant that commits present and future landowners to the perpetuation of the subspecies (USFWS, 2003).

3. Plans have been completed for the continued ecological management of habitats for a minimum of 10 potential metapopulation sites (USFWS, 2003).
4. A post-delisting monitoring plan covering a minimum of 10 potential metapopulation sites has been completed and is ready for implementation (USFWS, 2003).

**Recovery Actions:**

- Protect and increase all extant potential metapopulation sites (USFWS, 2003).
- Establish additional metapopulations and dispersal corridors (USFWS, 2003).
- Develop and execute a population and habitat management plan for each potential metapopulation site (USFWS, 2003).
- Accelerate and complete habitat enhancement projects on the Payette National Forest (USFWS, 2003).
- Develop and implement a transplantation effort to increase genetic diversity in each metapopulation (USFWS, 2003).
- Fully implement a long-term intensive and extensive metapopulation and habitat monitoring plan to evaluate success of recovery efforts (USFWS, 2003).
- Continue surveying efforts to locate new populations (USFWS, 2003).
- Conduct research to fill data gaps to ensure recovery (USFWS, 2003).
- Establish a captive propagation program as a hedge against extinction while wild populations are being reestablished, to provide an additional source for increasing genetic diversity of wild populations, provide a source for establishing new populations, provide research opportunities, and contribute to public education (USFWS, 2003).
- Increase efforts to enhance the outreach program for conservation of the northern Idaho ground squirrel (USFWS, 2003).
- Develop and maintain a comprehensive database to be used for monitoring the success of recovery (USFWS, 2003).
- Establish and maintain an interagency recovery coordinator and technical working group to coordinate recovery efforts (USFWS, 2003).
- Explore and initiate conservation options on private lands (USFWS 2017).
- Revise the Recovery Plan based on recommendations outlined in the 2011 5-year status review (USFWS 2017).
- Continue annual NIDGS population monitoring utilizing the long-term monitoring plan developed (USFWS 2017).
- Revise the existing suitable habitat model for the species, initially developed in 2007, utilizing recently collected annual monitoring and habitat data (USFWS 2017).

***Conservation Measures and Best Management Practices:***

- 1. Continue and increase habitat treatments for NIDGSs Given the primary threat continues to be meadow invasion by conifers, additional work is still needed to enhance and maintain habitat for the NIDGS. We encourage the PNF to continue their existing and ongoing efforts to enhance and maintain suitable habitat conditions for NIDGSs on National Forest lands. In addition, we support additional habitat treatments to benefit NIDGSs on non-federal lands (USFWS, 2011).
- 2. Explore and initiate conservation options on private lands As mentioned in the inadequacy of existing regulatory mechanisms, the development of occupied NIDGS habitat on private lands continues to be a threat to the species. Approximately 50% of known occupied habitat occurs on private land (Evans Mack in litt. 2010, p. 5). Options for conservation may include outright

acquisition, conservation easements, or long-term Safe Harbor Agreements, such as the 15-year agreement for the OX ranch, signed in 2009, that enrolls 3,150 ha (7,783 ac) of privately owned land (USFWS, 2011).

- 3. Revise the Recovery Plan The NIDGS Technical Working Group requested that the Service update the NIDGS Recovery Plan (NIDGS TWG in litt. 2010, entire). There are several aspects of the 2003 Recovery Plan for NIDGS that need revision that have been identified by the Technical Working Group, including: (1) Identify realistic population targets for recovery; (2) Clarify and/or redefine primary and secondary metapopulation areas; (3) Shift metapopulation site boundaries and re-assign occupied sites to better reflect recovery potential; (4) Provide an enhanced discussion of the role of private lands to recovery; (5) Discuss risks to squirrels that weren't identified initially (e.g. large vehicle traffic); (6) Discuss NIDGS suitable habitat and provide a copy of the suitable habitat model; (7) Expand the Probable Historic Distribution boundary based on new locations of squirrels; and (8) Acknowledge the diminished role of translocation as a recovery tool (NIDGS TWG in litt. 2010, p. 1) (USFWS, 2011).
- 4. Continue the NIDGS coordinator position As part of the 2003 Recovery Plan, recovery measure D (Coordinate the NIDGS Recovery Program), an interagency recovery coordinator position was established in 2003 (USFWS 2003, pp. 26-27). The primary responsibilities for the NIDGS recovery coordinator are to (1) coordinate and integrate ongoing interagency recovery programs, and (2) monitor NIDGS populations (Evans Mack 2010b, p. 1; Evans Mack 2011, p. 3). The coordinator is an IDFG employee, whose work is carried out through an agreement with, and partially funded by, the Service. From October 2008 through September 2010, coordinator accomplishments included; interagency program coordination; technical working group coordination and attendance; providing technical assistance regarding NIDGSs to 16 different agencies and private entities; providing information and education regarding NIDGSs through presentations and media outreach; securing funding for recovery actions; managing NIDGS tabular and spatial data; and, overseeing annual monitoring efforts (Evans Mack 2010b, entire) (USFWS, 2011).
- 5. Continue annual NIDGS population monitoring Since listing in 2000, annual NIDGS population monitoring has been conducted utilizing various methods. Beginning in 2002, population monitoring has been overseen by the NIDGS interagency program coordinator. In 2004, standardized field protocols were developed for monitoring NIDGS populations (Evans Mack 2004, entire). Protocols are updated as needed, with the latest update occurring in January, 2011 (Evans Mack 2011, entire). Presently, annual population monitoring includes examining demography and population trends at 5 intensive monitoring sites (Evans Mack 2011, p. 3), surveying previously known NIDGS sites to assess NIDGS occupancy, and surveying areas identified as suitable habitat for new populations (Evans Mack 2010a, p. 1) (USFWS, 2011).
- 6. Address information gaps. In their January, 2010 letter to the Service, the NIDGS Technical Working Group identified gaps in knowledge (NIDGS TWG in litt. 2010, entire). They include: a. Diet of northern Idaho ground squirrels and potential effects of forage competition with livestock and Columbian ground squirrels b. Other potential impacts of livestock grazing c. Other potential effects of competition with Columbian ground squirrels d. Monitoring effectiveness of habitat treatments to squirrel recovery, including timing of habitat treatments and maintenance treatments e. The impacts of predators and illegal hunting on northern Idaho ground squirrel populations f. Dispersal patterns and the importance of open habitat corridors for dispersal (USFWS, 2011).
- 7. Develop an updated Population Viability Analysis (PVA) model In 1993, a computer population viability simulation program was constructed utilizing recruitment and death values recorded over 8 years from 1 intensively studied NIDGS population site (Gavin et al. 1999, entire; Sherman and Yensen 1994, entire). Utilizing the variables of no natural immigration, and beginning the population viability analysis with 50 individuals (30 less than the actual population size of 80) the model

calculated that all but 1 of 100 population sites would become extinct in 20 years. In 1999, the Service contracted with the U.S. Geological Survey to develop a 2nd population model for NIDGS (Runge 1999, entire). Using the assumptions of a closed population and overwintering survival of the female and pups, this model predicted population extinction within 7 years (using 1999 demographic trend information) if no conservation measures were taken. It's been 12 years since a NIDGS population model was developed and the species has not gone extinct. Many recovery actions have been implemented by the agencies involved in recovery for the species. Information gathered from the annual interagency monitoring of NIDGS populations, demographics, and trends has been used to refine annual population estimates. Utilizing the data gathered from the annual monitoring of NIDGSs, an updated model, such as from a population viability analysis, could prove informative for future recovery planning and prioritization purposes (USFWS, 2011).

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## SPECIES ACCOUNT: *Urocyon littoralis catalinae* (Santa Catalina Island Fox)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Threatened; 03/05/2004; California/Nevada Region (R8) (USFWS, 2016)

### **Physical Description**

The Santa Catalina Island fox (*Urocyon littoralis catalinae*) is the diminutive relative of the gray fox (*U. cinereoargentea*), weighing between 1.8 and 3.0 kilograms (kg) (3 and 6 pounds [lb.]), and stands approximately 30 centimeters (cm) (1 foot [ft.]) tall (USFWS 2015a). The island fox has dark pelage and a linear measurement about 25 percent smaller than that of the mainland gray fox. Body length, including head and tail, ranges from 59 to 79 cm (1.9 to 2.6 ft.). Tail length alone ranges from 11 to 29 cm (4.3 to 11.4 inches [in.]). Height at the shoulder is from 12 to 15 cm (4.7 to 5.9 in.) (USFWS 2015b). The dorsal coloration is grayish-white and black, and the base of the ears and sides of the neck and limbs are cinnamon-rufous in color (USFWS 2015a). The tail has a contrasting thin black stripe on the dorsal side, with a mane of stiff hairs. This is the smallest fox species known from the United States. Adult males weigh 2.00 kg (4.4 lb.) on average, and adult females weigh 1.88 kg (4.14 lb.). The chin, lips, nose, and areas around the eyes are lined in black, and the sides of the rostrum are grey. The ears, neck, and sides of the limbs are cinnamon colored. The underside of the tail is a rusty color. Fur color may differ among islands and be highly variable among individuals, ranging from overall greyish to honey brown and red. Island grey foxes molt once a year during the fall months from August to November. At that time, the fur coat fades in color and the fur tips curl at the ends. Young foxes tend to have a paler but thicker dorsal fur coat compared to adults. In addition, the ears are darker in color compared to adult foxes. The Santa Catalina Island fox display sexual dimorphism, with males being heavier and larger than their female counterparts (USFWS 2015a).

### **Taxonomy**

The island foxes (*U. littoralis*) are important members of the *Urocyon* family because they share similar genetic makeup to the mainland grey fox, but have rapidly evolved into a smaller, separate species (NPS 2003). The island fox is the largest of the Channel Islands native mammals, but one of the smallest canid species in the world (NPS 2003). The Santa Catalina Island fox is one of six subspecies of island fox. The other subspecies, all of which inhabit islands off the California coast, are the San Clemente Island fox (*U. l. clementae*), San Nicolas Island fox (*U. l. dickeyi*), San Miguel Island fox (*U. l. littoralis*), Santa Cruz Island fox (*U. l. santacruzae*), and Santa Rosa Island fox (*U. l. santarosae*) (USFWS 2015a). The Santa Catalina Island fox is differentiated from other island fox subspecies by morphologic, genetic, and geographic distinctions; each of the six subspecies are limited in range to a single island (69 FR 10335). Genetic information indicates that all island foxes are descended from one colonization event, and can be distinguished by snout size, number of vertebrae, and tail length (USFWS 2015a; NPS 2003).

### **Historical Range**

The Santa Catalina Island fox range is restricted to the Santa Catalina Island, Los Angeles County, California (USFWS 2015a). Archeological and geological evidence suggests that foxes arrived on

the three northern islands about 10,400 to 16,000 years ago, and dispersed to the three southern islands (including Santa Catalina) 2,200 to 4,300 years ago (NatureServe 2015). It is also theorized that Native Americans may have translocated foxes from the northern islands to the southern islands (NatureServe 2015). Together with the fossil record, DNA restriction fragment evidence indicates that San Clemente Island was the first southern Channel Island colonized, probably by immigrants from San Miguel Island. Historically, the population of the Santa Catalina Island fox has been divided by a vertical isthmus, creating an eastern and western subpopulation (USFWS 2015a).

**Current Range**

Currently, Santa Catalina Island fox is endemic to Santa Catalina Island, where it occupies approximately 35 square kilometers (km<sup>2</sup>) (less than about 22 square miles [sq. mi.]) (NatureServe 2015). By 1994, Santa Catalina Island fox population had increased to an estimated 1,342 individuals. However, the population experienced catastrophic decline of more than 90 percent from 1999 to 2000, when it was estimated at fewer than 100 individuals. Recovery has occurred fairly quickly, with the total population in 2013 estimated at 1,852 individuals, of which 1,594 were adults (USFWS 2015a).

**Distinct Population Segments Defined**

No

**Critical Habitat Designated**

No;

***Life History*****Feeding Narrative**

Adult: Island foxes consume a wide variety of insects, such as grasshoppers, katydid, and Jerusalem crickets (*Stenopelmatus fuscus*) when they are seasonally available. They have also been known to prey on deer mice (*Peromyscus maniculatus*), house mice (*Mus musculus*), and introduced rodent species like black rats (*Rattus rattus*). As a solitary hunter, the island fox depends on stealth and agility to apprehend its prey (CIPF 2006). Like other species of fox, adults will co-forage with pups until they are self-sufficient, at approximately 4 to 5 months of age (USFWS 2015). Deer mice (*Peromyscus maniculatus*) are an important food resource during the breeding season, because they are an energy-rich food source that adult foxes can bring back to their pups. In addition to insects and mammals, island foxes have been known to prey on ground-nesting birds, such as horned larks (*Eremophila alpestris*) and western meadowlarks (*Sturnella neglecta*). Less commonly in the island fox diet are amphibians, reptiles, and the carrion of marine mammals. In addition to insectivorous and carnivorous appetite, the island fox also forages on native plants such as that of the summer holly (*Comarostaphylis* sp.), toyon (*Heteromeles* sp.), beavertail cactus (*Opuntia* sp.), stone fruit (*Prunus* sp.), rose (*Rosa* sp.), nightshade (*Solanum* sp.), huckleberry (*Vaccinium* sp.), and manzanita (*Arctostaphylos* sp.) genera (USFWS 2015). Island foxes have been in direct competition with feral cats, who are known to be more ravenous and aggressive than island foxes (USFWS 2015).

**Reproduction Narrative**

Adult: Island fox courtship activities occur from late January to early March, and breeding occurs once per year between February and early March. Following gestation for 50 to 53 days, a litter

of one to five pups (typically two) is born between late April and May (NatureServe 2015). In a captive breeding program, the observed average litter size was 2.4 (USFWS 2015). Island foxes give birth to their young under shrubs in dens, or in the side of ravines (USFWS 2015). Young foxes depend on their parents to provide shelter during the early stages of parental care. Both parents display a high level of parental care, foraging for their young and keeping them close throughout the summer months until August or September. Their shelters, commonly known as canid compounds, consist of multi-entrance tunnels, leading to intertwining chambers for quick entry and exit. Island foxes require habitat with dense topsoil to create dens (USFWS 2015). Deer mice (*Peromyscus maniculatus*) are important food items during the period of weaning and pup growth (NPS 2003). Some pups stay in their natal territory for up to 2 years, and others disperse earlier (USFWS 2015). Reproductive success is low for the species, reducing the chance for populations to recover following catastrophic decline (NPS 2003). In addition, the species lifespan is short, between 5 and 10 years. Most breeding involves older animals. On San Miguel Island, only 16 percent of juvenile females bred over a 5-year period, in contrast to 60 percent of older females (NPS 2003).

**Geographic or Habitat Restraints or Barriers**

Adult: Low vegetation types present less coverage and may render foxes more vulnerable to predation (USFWS 2015). They have also been known to avoid ravines and nonnative grasslands, because of difficult foraging conditions. The Santa Catalina Island is surrounded by water, a geographical barrier. The distance between islands make it unlikely to access new quality habitat on neighboring islands (USFWS 2015).

**Spatial Arrangements of the Population**

Adult: Random (USFWS 2015).

**Environmental Specificity**

Adult: Broad/generalist; community with key requirements common.

**Tolerance Ranges/Thresholds**

Adult: Moderate

**Site Fidelity**

Adult: High

**Dependency on Other Individuals or Species for Habitat**

Adult: Observed territory configuration changes after the death and replacement of paired male foxes, but not after the death and replacement of paired females or juveniles, indicating that adult males are involved in territory formation and maintenance (USFWS 2015). Island foxes rely on the abandoned burrows of other species for shelter. These foxes often salvage pre dug holes (NPS 2003).

**Habitat Narrative**

Adult: Santa Catalina Island foxes are habitat generalists; they occupy a wide range of topography, with habitat preferences that include valley and foothill native grasslands, southern coastal dunes, coastal sage scrub, coastal bluff, island chaparral, maritime cactus scrub, southern coastal oak woodland, southern riparian woodland, pine forests, and coastal marsh habitat types. Other island fox subspecies have been known to avoid ravines and scrub oak

patches. In addition, island foxes may use grasslands less than other habitats, even though insect prey is abundant in grasslands, because grasslands are denser and may be more difficult to forage in, and more vulnerable to aerial predation (USFWS 2015). Santa Catalina Island foxes require hollow structures, including ground holes, hollow trees, rock piles, shrubs, caves, and manmade structures for dens. If a fox is unable to find a suitable pre-hollowed area, it will dig a deep hole in the ground. These holes, called dens, serve as protection from predators, harsh weather conditions, and other dangers. Some dens start from burrows of other species, and some are created by the island foxes themselves (NPS 2003). The canid dens can consist of multi-entry complexes. Their designated territories are marked with feces and urine. Observed territory configuration changes after the death and replacement of paired male foxes, but not after the death and replacement of paired females or juveniles. This indicates that adult males play an important role in territory formation and maintenance (USFWS 2015). The USFWS did not designate any critical habitat for the island fox because: 1) the island fox is a habitat generalist and an opportunistic omnivore; 2) prior to predation by golden eagles and the outbreak of disease, habitat did not appear to be a limiting factor despite human-induced habitat changes; and 3) the primary reasons for the listing of the fox were predation and disease (USFWS 2015).

***Dispersal/Migration*****Motility/Mobility**

Adult: Moderate

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory (NatureServe 2015).

**Dispersal**

Adult: Low

**Immigration/Emigration**

Adult: No

**Dependency on Other Individuals or Species for Dispersal**

Adult: No

**Dispersal/Migration Narrative**

Adult: Santa Catalina foxes have moderate mobility and are nonmigratory canid species. They have been observed to have low dispersal and no immigration/emigration, due to their geographic isolation (USFWS 2015), with a home range of 0.80 to 1.6 kilometers (0.5 to 1.0 mile) (NPS 2003). The two subpopulations of Santa Catalina Island foxes are separated by an isthmus on the island, lessening potential dispersal and home range (NPS 2003). The island fox's territory is marked by feces and urine and can move location throughout the fox's lifetime (NPS 2003). Island foxes are known to have extended parental care, with the male fox playing an important role in the rearing of the young (NPS 2003). Some pups disperse at 7 months, while others remain on their natal territories into their second year (NPS 2003). Island foxes have small territories with high densities and shorter dispersal distances than the mainland gray fox (*U. cinereoargentea*). The home range and size configuration of the island fox is dependent on landscape features, fox population density, resource distribution, sex of the animal, habitat

type, and season. Males are more involved in territory formation and maintenance than female foxes. Recorded home ranges of other subspecies of island fox varied from 0.15 to 0.87 km<sup>2</sup> (0.06 to 0.34 sq. mi.), and averaged 0.55 km<sup>2</sup> (0.21 sq. mi.) in size during a period of moderate to high fox density (equivalent to seven island foxes per km<sup>2</sup> [18 per sq. mi.], from Santa Cruz Island and San Clemente Island) (USFWS 2015).

**Additional Life History Information**

Adult: Island foxes have small territories with high densities and shorter dispersal distances than the mainland gray fox (*U. cinereoargentea*). The home range and size configuration of the island fox is dependent on landscape features, fox population density, resource distribution, sex of the animal, habitat type, and season. Males are more involved in territory formation and maintenance than female foxes. Recorded home ranges of other subspecies of island fox varied from 0.15 to 0.87 km<sup>2</sup> (0.06 to 0.34 sq. mi.), and averaged 0.55 km<sup>2</sup> (0.21 sq. mi.) in size during a period of moderate to high fox density (equivalent to seven island foxes per km<sup>2</sup> [18 per sq. mi.], from Santa Cruz Island and San Clemente Island) (USFWS 2015).

***Population Information and Trends*****Population Trends:**

Decreasing (NatureServe 2015)

**Species Trends:**

Decreasing; due to the introduction of CDV (USFWS 2015).

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Population Growth Rate:**

Declining (USFWS 2015)

**Number of Populations:**

One extant population; two subpopulations (NPS 2003; NatureServe 2015).

**Population Size:**

Estimated at 1,852 individuals, of which 1,594 were adults in 2013 (USFWS 2015).

**Minimum Viable Population Size:**

The minimum viable population size is undescribed; however, the carrying capacity of Santa Catalina Island is approximately 1,500 foxes. Although not specified for the Southern Channel Islands, captive breeding programs modeled for the Northern Channel Islands required an on-island captive population of 20 breeding pairs (USFWS 2015).

**Resistance to Disease:**

Low

**Adaptability:**

Low

**Additional Population-level Information:**

The Santa Catalina Island fox population is composed of two subpopulations that are separated by an isthmus, with the eastern subpopulation at greater risk of extinction (due to CDV) than the western subpopulation (NPS 2003).

**Population Narrative:**

The Santa Catalina Island fox population is composed of two subpopulations that are separated by a vertical isthmus, with the eastern subpopulation at greater risk of extinction, specifically from CDV, than the western subpopulation (NPS 2003). Although the risk of extinction is greater for the easternmost foxes, the entire population is steadily declining due to interactions with humans, disease, competition with feral cats, and loss of genetic diversity (USFWS 2015). With low resiliency, representation, and redundancy, the Santa Catalina Island fox is critically imperiled, facing the threat of near extinction (USFWS 2015). Densities between 1988 and 1991 ranged from 2.6 island foxes per km<sup>2</sup> (6.7 per sq. mi.) in grassland to 12.7 island foxes per km<sup>2</sup> (32.9 per sq. mi.) in scrub/dune habitats. By the mid-90s, Santa Catalina Island fox populations had increased to an estimated 1,342 individuals. However, the population experienced catastrophic decline of more than 90 percent from 1999 to 2000, when it was estimated at fewer than 100 individuals. Recovery has occurred fairly quickly, with the total population in 2013 estimated at 1,852 individuals, of which 1,594 were adults (USFWS 2015). If recovery criteria are met, the Santa Catalina Island fox could be recovered by 2024, with an estimated carrying capacity of 1,500 foxes, or 20 breeding pairs (USFWS 2015).

**Threats and Stressors**

**Stressor:** The threat of disease, such as that posed by ear tumors and canine distemper virus (CDT)

**Exposure:** Direct; potential for pathogen introduction from the movement of wild and domestic mammals.

**Response:** Reduced fitness; inability to compete for resources, find mates, and defend territory.

**Consequence:** High mortality rates.

**Narrative:** The threat of disease, such as ear tumors and canine distemper virus (CDV), is a continued concern for the Santa Catalina Island fox. The catastrophic decline on Santa Catalina was caused by CDV. The recent finding of ear tumors in Santa Catalina Island foxes, confirmed to be a source of mortality in wild foxes, is of high enough frequency to be considered a concern. The response to decreased fitness leads to an inability to compete for resources, find mates, and defend territory. Island foxes on all islands are vaccinated against CDV and rabies, the two diseases for which active mitigation measures could not be implemented in a timely manner once an outbreak was detected. The number of foxes vaccinated on each island is generally the number required to start a captive breeding program (75 to 100), were the population to be affected by an epidemic. Disease remains a concern for Santa Catalina Island foxes, because the island has high accessibility and a sizable human population (USFWS 2015).

**Stressor:** Competition with feral cats

**Exposure:** Direct

**Response:** Increased incidence of competition with feral cats leads to starvation and decreased viability.

**Consequence:** Death of individuals and greatly reduced populations.

**Narrative:** Competition with feral cats has led to starvation of juvenile foxes, increasing mortality rates, and a reduction in overall population numbers. The introduction of feral cats to the island has greatly disrupted the ecosystem for native predators (USFWS 2015).

**Stressor:** Mortality or injury from vehicular strikes

**Exposure:** Direct

**Response:** Reduced population size; less dispersal potential.

**Consequence:** Reduction in population numbers; decreased reproductive success; and higher susceptibility to mortality/extirpation.

**Narrative:** Death from vehicle collision on roads is the largest known source of mortality on San Clemente Islands, account for a minimum of 26 foxes per year between the years 1991 and 1995. Between 2003 and 2007, the annual average of foxes killed by vehicles was 4 per year, but the number of foxes killed has increased in the past several years as the fox numbers have increased: in 2011, 16 foxes were killed by vehicles, and in 2013, 12 foxes were killed by vehicles. It is likely that some foxes were hit and later succumbed to their injuries, or that there are juveniles who did not survive following the death of the mother. Because there is not have a method to document this type of mortality, the actual annual mortality due to vehicles is likely higher. Road mortality may impact island fox population dynamics (USFWS 2015).

**Stressor:** Inbreeding, loss of adaptive potential, loss of heterozygosity, and small population size

**Exposure:** Direct, indirect.

**Response:** Decrease in heterogeneity, population bottleneck.

**Consequence:** Reduction in population numbers, decreased reproductive success, increased genetic effects of population bottleneck, higher susceptibility to mortality/extirpation.

**Narrative:** Inbreeding and loss of heterozygosity have the potential to affect viability-related fitness traits in island foxes. Loss of genetic variation (adaptive potential) significantly affects the likelihood of persistence of the island fox over longer time frames (USFWS 2015).

**Stressor:** Habitat or range destruction, modification, or curtailment

**Exposure:** Indirect

**Response:** Decreased home range, increased interaction between foxes and humans.

**Consequence:** Increased mortality.

**Narrative:** Destruction, modification, or curtailment of habitat or range has a large effect on the success of island fox populations. The frequent interactions with domestic animals and humans has reduced home range and increased incidental mortalities of island foxes (USFWS 2015).

### ***Recovery***

#### **Reclassification Criteria:**

The following conditions would most likely result in a determination that downlisting of the Santa Catalina Island fox is warranted:

Santa Catalina Island fox exhibits demographic characteristics consistent with long-term viability, meaning it has no more than a 5 percent risk of quasi-extinction over a 50-year period. 1. Quasi-extinction is a population size of fewer than 30 individuals. 2. The risk of extinction is calculated based on a combined 90 percent confidence interval for a 3-year running average of population size estimates. 3. Five percent or less at risk is sustained for at least 5 years, during which time the population trend is not declining (USFWS 2015).

**Delisting Criteria:**

In addition to the reclassification criteria, the following conditions would mostly likely result in a determination that delisting the Santa Catalina Island fox is warranted:

Land managers are able to respond in a timely fashion to potential or incipient disease outbreaks, and to other identified threats, using the best available technology. A disease management strategy is developed, approved, and implemented by the land manager(s) in collaboration with USFWS, including review by the appropriate Integrated Island Fox Recovery Team Technical Expertise Group or the equivalent. This strategy must include: 1. Identification of a portion of Santa Catalina Island fox that will be vaccinated against CDV and rabies, the diseases posing the greatest risk and for which vaccines are safe, effective, and available. Vaccinations to be provided and numbers vaccinated will be developed in consultation with appropriate subject-matter experts; 2. Identification of actual and potential pathogens of island foxes, and the means by which these can be prevented from decimating fox populations; 3. Measures to prevent diseases in island foxes; 4. A monitoring program that provides for timely detection of a disease outbreak, and an associated emergency response strategy as recommended by the appropriate subject-matter experts; and 5. A process for updating the disease management strategy as new information arises (USFWS 2015).

**Recovery Actions:**

- Reduce mortality for each subspecies of island fox to ensure that populations persist at sustainable levels (USFWS 2015).
- Conduct captive breeding and reintroduction of island foxes to increase population size (USFWS 2015).
- Establish island fox monitoring strategies (USFWS 2015).
- The long-term conservation strategy identifies actions that would further the conservation of the island fox. At this time, these activities are not essential for preventing extinction and are not required for downlisting or delisting a particular island fox subspecies; however, these activities could substantially enhance the long-term conservation of the species and may also increase our scientific understanding of the island fox. The USFWS has identified the following long-term conservation actions:
- Establish a mainland captive island fox population on which to conduct research to better understand fox behavior, ecology, reproduction, disease, and vaccine efficacy (USFWS 2015).
- Establish, expand, and continue island fox education and outreach programs (USFWS 2015).
- Assessing the demographic impact of other threats such as mortality from vehicle strikes, competition with feral cats, and emerging disease issues (USFWS 2015).
- Restoring island habitat (USFWS 2015).
- Establishment of conservation agreements (USFWS 2015).

***Conservation Measures and Best Management Practices:***

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***Additional Threshold Information:***

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## SPECIES ACCOUNT: *Ursus arctos horribilis* (Grizzly bear)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Threatened/Experimental Population (unoccupied); 07/28/1975, 11/17/2000, 07/28/1975; Mountain-Prairie Region (R6) (USFWS, 2016)

### **Physical Description**

A large bear. Color ranges from pale yellowish to dark brown; usually white tips on the hairs, especially on the back, resulting in a frosted or grizzled effect; facial profile concave; claws on front feet of adults about 4 inches long and curved; noticeable hump above shoulders; head and body of adults about 6-8 feet, height at shoulders 3-4.5 feet (Burt and Grossenheider 1964). Grizzly Bears reach weights of 180-680 kg (400-1,500 lb); the male is on average 1.8 times as heavy as the female, an example of sexual dimorphism. (USFWS, 2016; NatureServe, 2015)

### **Taxonomy**

Grizzly bears (*Ursus arctos horribilis*) are vertebrates that belong to the Class Mammalia, Order Carnivora, and Family Ursidae. The grizzly bear is a member of the brown bear species (*U. arctos*) that occurs in North America, Europe, and Asia; the subspecies *U. a. horribilis* is limited to North America (Rausch 1963; Servheen 1999). Early taxonomic descriptions of *U. arctos* based primarily on skull measurements described more than 90 subspecies (Merriam 1918), but this was later revised to 2 subspecies in North America: *U. a. middendorfi* on the islands of the Kodiak archipelago in Alaska and *U. a. horribilis* in the rest of North America (Rausch 1963). The two North American subspecies approach of Rausch (1963) is generally accepted by taxonomists today, and is the approach we use. (USFWS, 2015)

### **Historical Range**

Formerly throughout western North America, north from northern Mexico; northwestern Africa, all of the Palearctic from western Europe, Near and Middle East through the northern Himalayas to western and northern China and Chukot (Russia) and Hokkaido (Japan) (Wozencraft, in Wilson and Reeder 1993). (NatureServe, 2015)

### **Current Range**

In North America, present range includes Alaska, northern and western Canada, northern Continental Divide in Montana, Cabinet/Yaak mountains in Montana/Idaho, Selkirk Mountains in Idaho/Washington, Northern Cascades in Washington, and Yellowstone area, Wyoming/Montana/Idaho. Some bears in the Cabinet-Yaak ecosystem of Montana and Idaho and Selkirk ecosystem of Idaho and Washington mingle in the Purcell Mountains in southern British Columbia, and movement data indicate that the Cabinet-Yaak and Selkirk populations are connected to a much larger population (several hundred bears) extending north into British Columbia (USFWS 1999). (NatureServe, 2015)

### **Distinct Population Segments Defined**

Yes; Greater Yellowstone Ecosystem, North Cascades Ecosystem, U.S.A. (portions of ID and MT-experimental population), U.S.A (lower 48 states except where listed as an experiment population) (USFWS, 2016)

### **Critical Habitat Designated**

Yes;

### ***Life History***

#### **Feeding Narrative**

Adult: Grizzly bears are opportunistic omnivores with high diet variability among individuals, seasons, and years (Mattson et al. 1991a; Mattson et al. 1991b; Schwartz et al. 2003b; LeFranc et al. 1987; Felicetti et al. 2003; Felicetti et al. 2004). In all areas, vegetal matter is a dominant portion of the diet. Feeds on carrion, fish (especially coastal populations), large and small mammals, insects, fruit, grasses, bark, roots, mushrooms, and garbage. May cache food (and guard it). In the Yellowstone region, ungulate remains were a major portion of early season scats; graminoids dominated in May and June, and whitebark pine seeds were most important in late season scats; berries composed a minor portion of scats in all seasons (Mattson et al. 1991). May feed on insect aggregations (e.g., army cutworm moths, ladybird beetles); in Shoshone National Forest, Yellowstone ecosystem, alpine insect aggregations are an important source of food, especially in the absence of high-quality foraging alternatives in July and August of most years (Mattson et al. 1991). In Waterton Lakes National Park, Alberta, main food was roots of *Hedysarum sulphurescens* in spring and autumn, *Erythronium grandiflorum* corms and green vegetation (mainly umbellifers) from June through early August; *Vaccinium* fruits were important in late July and August (Hamer et al. [1991]). Sometimes preys on black bear and conspecifics (Mattson et al., 1992, J. Mamm. 73:422-425). Tends to be predominantly crepuscular with the least activity during midday, but much individual variation. In preparation for hibernation, bears increase their food intake dramatically during a stage called hyperphagia (Craighead and Mitchell 1982). Hyperphagia is defined simply as overeating (in excess of daily metabolic demands) and occurs throughout the 2-4 months prior to den entry (i.e., August–November). During hyperphagia, excess food is deposited as fat, and grizzly bears may gain as much as 1.65 kg/day (3.64 lb/day) (Craighead and Mitchell 1982). (USFWS, 2011; NatureServe, 2015)

#### **Reproduction Narrative**

Adult: Mating occurs from May-July with a peak in mid-June (Craighead and Mitchell 1982; Nowak and Paradiso 1983). Although females mate from mid-May through early July, their fertilized embryos are not implanted into the uterus until late fall, once enough nutrition is attained to survive the winter and nurse cubs for 2-3 months inside the den (Schwartz et al. 2003a, 2003b, 2006a). Litter size varies from 1-4 cubs (average 2) (Schwartz et al. 2003b). Gestation lasts about 184 days. Cubs are born in the den in late January or early February and remain with the female for 2-3 years before the mother will again mate and produce another litter (Schwartz et al. 2003b). Breeding interval generally is 2-4 years. In North America, first parturition occurs at 5-6 years in the south, 6-9 years in the north. A few live as long as 20-25 years. Long life span, late sexual maturity, protracted reproductive cycles.; May congregate in areas with abundant food; otherwise solitary except when breeding or caring for young. (USFWS, 2011; NatureServe, 2015)

#### **Geographic or Habitat Restraints or Barriers**

Adult: Major water barriers; arbitrarily set at those greater than 5 kilometers across. (NatureServe, 2015)

#### **Environmental Specificity**

Adult: Low (inferred from NatureServe, 2015)

**Tolerance Ranges/Thresholds**

Adult: High (inferred from NatureServe, 2015)

**Site Fidelity**

Adult: Low (USFWS, 2011)

**Habitat Narrative**

Adult: Now found mostly in arctic tundra, alpine tundra, and subalpine mountain forests. Once found in a wide variety of habitats including: open prairie, brushlands, riparian woodlands, and semidesert scrub. Ranges widely at the landscape level. Most populations require huge areas of suitable habitat. Common only where food is abundant and concentrated (e.g., salmon runs, caribou calving grounds). Typically digs own hibernation den, usually on steep northern slope where snow accumulates. Young are born in den in cave, crevice, hollow tree, hollow dug under rock, or similar site. Use of summit or ridge for mating (in May-June) reported for Banff National Park, Alberta, but not elsewhere (Hamer and Herrero 1990). In the Northwest Territories, Canada, all dens were on well-drained slopes; the majority of dens faced south (25), followed by west (13), east (10), and north (8); most dens were constructed under cover of tall shrubs (*Betula glandulosa* and *Salix*), the root structures of which supported ceilings of dens; esker habitat was selected more than expected by chance (McLoughlin et al. 2002). In Spain, remnant deciduous forests and upland creek drainages were prime feeding areas (Clevenger et al. 1992). In the lower 48 States, annual home range sizes for female grizzly bears are approximately 400 sq km (150 sq mi) (LeFranc et al. 1987). For males, annual home ranges vary from 286-1,398 sq km (110-540 sq mi), but average approximately 800 sq km (309 sq mi) (LeFranc et al. 1987). (USFWS, 2011; NatureServe, 2015)

***Dispersal/Migration*****Motility/Mobility**

Adult: High (NatureServe, 2015)

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Non-migratory but large home range resulting in local migration (NatureServe, 2015)

**Dispersal**

Adult: High (NatureServe, 2015)

**Dispersal/Migration Narrative**

Adult: In North America, often exhibits discrete elevational movements from spring to fall, following seasonal food availability (LeFranc et al. 1987); generally at lower elevations in spring and higher elevations in mid-summer and winter. Home range exhibits much variation among different individuals, areas, and seasons; male range generally is larger than that of female; annual range varies from less than 25 sq km (Kodiak Island) to more than 2000 sq km (LeFranc et al. 1987). (NatureServe, 2015)

***Population Information and Trends***

**Population Trends:**

Short-term trends indicate declines of 10 to 30% (NatureServe, 2015)

**Resiliency:**

Medium (inferred from USFWS, 2011)

**Representation:**

High (inferred from USFWS, 2011)

**Redundancy:**

Medium (inferred from USFWS, 2011)

**Number of Populations:**

81 to >300 (NatureServe, 2015)

**Population Size:**

~50,000 in North America; 31,000 - 36,000 in the United States (NatureServe, 2015)

**Minimum Viable Population Size:**

Short-term fitness (i.e., survival and reproduction rates) can be attained by maintaining an effective population size of at least 50 individuals (Frankel and Soule 1981). This corresponds to an inbreeding rate of 1% per generation, which is considered “tolerable” by animal breeders (Frankel and Soule 1981). For long-term fitness (i.e., the ability to adapt), Franklin (1980) posited that an effective population size of at least 500 is needed to prevent the loss of genetic diversity over time through the natural processes of inbreeding, drift, and mutation. (USFWS, 2011)

**Population Narrative:**

Short-term population trends indicate declines of 10 to 30%. In North America, there are currently about 30,000-35,000 grizzly bears in Alaska, 21,660 in Canada, and 800-1000 in the lower 48 states. As of the early 1990s, the Yellowstone population was estimated at 200-350 (Mattson and Reid 1991). USFWS (1990) noted that a record 57 cubs were born in the Yellowstone ecosystem in 1990. Northern Continental Divide population was estimated at 440-680 in 1985, unknown number in Selway-Bitterroot (probably fewer than 10) (Matthews and Moseley 1990). Selkirk recovery zone includes an estimated 46 bears, 19 in the U.S. and 27 in Canada (USFWS 1999). Cabinet-Yaak recovery zone supports 30-40 bears (conservative estimate) (USFWS 1999). Between 1964 and 1991, there were 21 credible reports of grizzly bears in the North Cascades south of Canada (Almack et al. 1993). Short-term fitness (i.e., survival and reproduction rates) can be attained by maintaining an effective population size of at least 50 individuals (Frankel and Soule 1981). This corresponds to an inbreeding rate of 1% per generation, which is considered “tolerable” by animal breeders (Frankel and Soule 1981). For long-term fitness (i.e., the ability to adapt), Franklin (1980) posited that an effective population size of at least 500 is needed to prevent the loss of genetic diversity over time through the natural processes of inbreeding, drift, and mutation. (USFWS, 2011; NatureServe, 2015)

**Threats and Stressors**

**Stressor:** Habitat loss and fragmentation (NatureServe, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Historical decline due to habitat loss and fragmentation and killing by humans. Primary threats are habitat alienation, alteration, and loss; increased access to wilderness; and hunting (both legal and illegal). Increased access increases human-bear contacts, some of which result in destruction of bears. Alien species threaten food resources in some areas; in Montana, white pine blister rust has killed whitebark pines (seeds serve as food for bears) and knapweed have displaced native plants that serve as foods for bears and their prey. See Horejsi (1989) for a discussion of land-use threats (petroleum and natural gas development, grazing by domestic stock) and excessive bear mortality in southwestern Alberta. Basic problem in the Cabinet-Yaak/Selkirk ecosystem is reduced habitat availability due to land use by humans and increased human access into habitat; this results in increased bear mortality. Access management plans have addressed this problem to some degree but have not been fully implemented (USFWS 1999). Several large mines in Montana, if approved, may pose a threat (USFWS 1999). Forestry, mining, recreation, and road building also affect habitat in British Columbia where the two portions of this distinct population segment are connected (USFWS 1999). (NatureServe, 2015)

**Stressor:** Timber practices (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** "Timbering practices" were identified in 1975 as activities which may compromise grizzly bear habitat (40 FR 31734, July 28, 1975). The primary impacts to grizzly bears associated with extractive activities such as timber harvest, mining, and oil and gas development are increases in road densities, with subsequent increases in human access, grizzly bear/human encounters, and human-caused grizzly bear mortalities (McLellan and Shackleton 1988, 1989; Mace et al. 1996). Although seismic exploration associated with oil and gas development or mining may disturb denning grizzly bears (Harding and Nagy 1980; Reynolds et al. 1986), actual den abandonment is rarely observed, and there has been no documentation of such abandonment by grizzly bears in the lower 48 States (Hegg 2010; Kasworm 2010; Servheen 2010). (USFWS, 2011)

**Stressor:** Livestock use (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The 1975 listing identified "livestock use of surrounding national forests" as detrimental to grizzly bears " ... unless management measures favoring the species are enacted" (40 FR 31734, p. 31734). While grizzly bears frequently coexist with cattle without depredating them, when grizzly bears encounter domestic sheep, they are often attracted to them and depredate the sheep (Jonkel 1980; Knight and Judd 1983; Orme and Williams 1986; Anderson et al. 2002). If repeated depredations occur, managers either relocate the bear or remove it from the population, resulting in such domestic sheep areas becoming population sinks (Knight et al. 1988). As referenced in previous paragraphs, the implementation of the Guidelines led to the reduction of many livestock allotments with an emphasis on sheep allotments. Available

information regarding livestock allotments for each recovery zone is reported below, by area. (USFWS, 2011)

**Stressor:** Human development (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Human developed sites can impact bears through temporary or permanent habitat loss and displacement, unsecured bear attractants, increased length of time of human presence, and increased human disturbance to surrounding areas. Developed sites refer to sites developed or improved for human use or resource development. Examples include campgrounds, trailheads, lodges, summer homes, restaurants, visitor centers, oil and gas exploratory wells, production wells, active mining operations, and work camps. The primary concerns for grizzly bears related to developed sites are direct mortality from bear/human encounters, food conditioning, and habituation of bears to humans (Mattson et al. 1987). Habituation occurs when grizzly bears encounter humans or developed sites frequently, and without negative consequences, so that the bears no longer avoid humans and areas of human activity (FWS 1993). Habituation does not necessarily involve human-related food sources. Food conditioning occurs when grizzly bears receive human-related sources of food and thereafter seek out humans and human use areas as feeding sites (FWS 1993). Gunther (1994) noted that grizzly bear management in Yellowstone National Park has shifted from problems involving food-conditioned bears to problems involving habituated (but not food-conditioned) bears seeking natural foods near developed sites or along roadsides. Because of the issues associated with developed sites, unregulated residential development is a concern in and around grizzly bear recovery zones. Only a small subset of this development undergoes Section 7 consultation via a Federal nexus. The sale of private lands that were traditionally commercial forest lands to other private owners for real estate development has led to an increase in private residential development. Residential development that occurs on private lands by private entities does not undergo Federal consultation under Section 7 of the ESA unless there is a Federal nexus to this development. This means that these rural subdivisions are not required to consult with the FWS or mitigate for grizzly bear impacts under the ESA even if they are developing within grizzly bear recovery zones or surrounding occupied habitat. In the States of Washington, Wyoming, Montana, and Idaho, most residential development regulation is at the county government level. (USFWS, 2011)

**Stressor:** Human-grizzly bear encounters (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** As human population densities increase, the frequency of encounters between humans and grizzly bears also increases, which can result in more human-caused grizzly bear mortalities due to a perceived or real threat to human life or property (Mattson et al. 1996). This outcome happens because human population growth results in corresponding increases in both the number of people recreating in grizzly bear habitat and human site developments. (USFWS, 2011)

**Stressor:** Recreation (USFWS, 2011)

**Exposure:**

**Response:****Consequence:**

**Narrative:** In 1975, “trail construction in ... areas which were formerly inaccessible” and “increasing human use of Yellowstone and Glacier National Parks” were identified as possible threats to grizzlies (40 FR 31734, July 28, 1975). The FWS recognized that increasing recreational use of grizzly bear habitat could be detrimental if not properly managed. Based on current recreation and human population growth trends, the number of people recreating in grizzly bear habitat is expected to increase (USFS 2006a; Cordell et al. 2008; NPA Data Services 2008, 2009; USFS 2009). The primary concerns associated with recreational activities are the same as those with developed sites: displacement, direct mortality from bear/human encounters and habituation of bears to humans (Joslin and Youmans 1999; White et al. 1999; FWS 2002). Snowmobiling and Off Road Vehicle (ORV) use have the potential to negatively impact individual grizzly bears although population level impacts have not been documented. Many of these recreational activities, such as snowmobiling and ORV use, can be regulated through motorized access management and travel planning. The potential impacts of non-motorized recreational activities such as hiking, cross-country skiing, and hunting on grizzly bears are mitigated most effectively through educational outreach. Most grizzly bear/human conflicts with people recreating on National Forest lands are related to hunting (Servheen et al. 2004) (please see our discussion of this source of mortality under Factor C below). These surprise encounters and misidentifications frequently result in grizzly bear mortalities. Most conflicts between grizzly bears and people recreating in grizzly bear habitat can be avoided if proper educational materials are received and followed. (USFWS, 2011)

**Stressor:** Snowmobiling (USFWS, 2011)

**Exposure:****Response:****Consequence:**

**Narrative:** Snowmobiling has the potential to disturb bears while in their dens and after emergence from their dens in the spring. Because grizzly bears are easily awakened in the den (Schwartz et al. 2003b) and have been documented abandoning den sites after seismic disturbance (Reynolds et al. 1986), the potential impact from snowmobiling must be considered. The Service found no studies in the literature specifically addressing the effects of snowmobile use on any denning bear species and the information that is available is anecdotal in nature (FWS 2002). (USFWS, 2011)

**Stressor:** Den disturbances (USFWS, 2011)

**Exposure:****Response:****Consequence:**

**Narrative:** Disturbance in the den has the potential to result in energetic costs (increased activity and heart rate inside the den) and possibly den abandonment, ultimately causing a decline in physical condition of the individual or even cub mortality (Graves and Reams 2001). Although the potential for this type of disturbance while in the den certainly exists, Reynolds et al. (1986) found that grizzly bears denning within 1.4-1.6 km (0.9-1.0 mi) of active seismic exploration and detonations moved around inside their dens but did not leave them. Harding and Nagy (1980) documented two instances of den abandonment during fossil fuel extraction operations. One bear abandoned its den when a seismic vehicle drove directly over the den (Harding and Nagy 1980). The other bear abandoned its den when a gravel mining operation literally destroyed the

den (Harding and Nagy 1980). Reynolds et al. (1986) also examined the effects of tracked vehicles and tractors pulling sledges. In 1978, there was a route for tractors and tracked vehicles within 100 meters (m) (328 feet (ft)) of a den inhabited by a male. This male was not disturbed by the activity nor did he abandon his den at any point. Reynolds et al. (1986) documented only one instance of possible den abandonment due to seismic testing (i.e., detonations) within 200 m of a den (Reynolds et al. 1986). This bear was not marked but an empty den was reported by seismic crews. Swenson et al. (1997) monitored 13 different grizzly bears for at least 5 years each and documented 18 instances of den abandonment, 12 of which were related to human activities. Although many of these instances (n=4) were hunting related (i.e., gunshots fired within 100 m (328 ft) of the den), 2 occurred after “forestry activity at the den site,” 1 had moose and dog tracks within 10 m of a den, 1 had dog tracks at the den site, 1 had ski tracks within 80-90 m from a den, 1 had an excavation machine working within 75 m of a den, and 2 were categorized as “human related” without further details (Swenson et al. 1997). (USFWS, 2011)

**Stressor:** Human predation (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The original 1975 listing for grizzly bears equated overutilization with any type of human-caused mortality. Within this context, the Rule stated that “overutilization” was resulting in “... a continual loss of animals through indiscriminate illegal killing ... control operations ... and livestock depredations.” We now address human-caused mortality from illegal kills, management removals, livestock depredations, defense of life and property, mistaken identity, and accidental take as a type of “predation.” Since 1980, there have been 83 grizzly bear mortalities for recreational purposes (i.e., legal grizzly bear hunting) in the NCDE and in adjacent population units in Canada. In Montana, there was a legal grizzly bear hunting season until 1991. Between 1980 and 1991, there were 81 grizzly bear mortalities in the NCDE during the legal grizzly bear hunting season for recreational purposes. This hunting season in the lower 48 States was suspended permanently in 1991 (57 FR 37478, August 19, 1992). In total, there have been 29 grizzly bears in the lower 48 States that have died for scientific purposes between 1980 and 2008. These mortalities were accidental mortalities from research trapping and handling. In sum, a total of 112 grizzly bears from the lower 48 States have died since 1982 (when detailed mortality record keeping began) for scientific or recreational purposes. (USFWS, 2011)

**Stressor:** Disease (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Researchers have documented grizzly bears with brucellosis (type 4), clostridium, toxoplasmosis, canine distemper, canine parvovirus, canine hepatitis, leptospirosis, and rabies (Zarnke 1983; LeFranc et al. 1987; Zarnke and Evans 1989; Marsilio et al. 1997; Zarnke et al. 1997). The most common internal parasite noted in grizzly bears is *Trichinella* for which 62% of grizzly bears tested positive from 1969-1981 (Greer 1982). Disease screening of captured black and grizzly bears in the CYE, SE, and NCDE recovery zones during 2000 showed antibody levels consistent with exposure to several diseases, but no clinical sign of disease (Port et al. 2001). Effects of these levels of incidence are unknown but negative impacts to vital rates or bears showing symptoms of these diseases have not been documented. Despite this lack of observed data indicative of symptoms or population level impacts, monitoring will continue. Although

grizzly bears have been documented with a variety of bacteria and other pathogens, parasites, and disease, fatalities are uncommon (LeFranc et al. 1987) and population-level impacts on grizzly bears have not been documented (Jonkel and Cowan 1971; Mundy and Flook 1973; Rogers and Rogers 1976). Based on 30 years of monitoring in grizzly bear ecosystems, natural mortalities in the wild due to disease are rare (IGBST 2005) and it is likely that mortalities due to any of these bacteria or pathogens are negligible components of total mortality. Disease is likely to remain an insignificant factor in population dynamics into the foreseeable future. (USFWS, 2011)

**Stressor:** Natural predation and mortality (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Grizzly bears are occasionally killed by other bears. Adult grizzly bears kill cubs, subadults, or other adults (Stringham 1980; Dean et al. 1986; Hessing and Aumiller 1994; McLellan 1994; Schwartz et al. 2003b). This source of natural grizzly bear mortality seems to occur rarely (Stringham 1980) and there were only 30 known grizzly bear mortalities in the lower 48 States between 1980 and 2008 attributed to this type of natural predation: 14 in the GYA; 13 in the NCDE; and 3 in the CYE. Overall, these types of aggressive interactions among grizzly bears are rare and are likely to remain an insignificant factor in population dynamics into the foreseeable future. Other sources of natural mortality in grizzly bears include starvation and natural events such as avalanches or fires. For a complete discussion of this type of natural mortality in the GYA, please see p. 14920 of the Yellowstone Final Rule (Appendix A). In the lower 48 States outside of the GYA, natural mortalities accounted for 8.5% (24 / 284) of all known mortalities between 1999 and 2008. This included 15 natural mortalities in the NCDE; 9 in the CYE; and 0 that we are aware of in the SE (TABLE 2). This is comparable to the natural mortality rate of 11.4% in the GYA (Servheen et al. 2004). All nine of the natural mortalities in the CYE occurred between 1999 and 2002. While there have not been any known natural mortalities since 2002 in the CYE, in light of the natural mortality levels observed from 1999-2002, the FWS remains vigilant regarding this factor and continues to monitor natural mortality to determine if it becomes a threat to the population at some point in the future. Monitoring of this factor will continue, but natural predation and mortality do not appear to be limiting the population at this point. Our figures for natural mortality in the SE may be incomplete. In recent years, natural mortalities in the SE have gone largely undetected because there is no active research trapping program. Overall, the level of natural mortality observed in the lower 48 States is not unusual. Even without the protections of the ESA, natural mortality would not threaten grizzly bear populations. (USFWS, 2011)

**Stressor:** Human predation (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Humans have historically been the most effective predators of grizzly bears. Excessive human-caused mortality is the driving factor behind grizzly bear declines during the 19th and 20th centuries (Leopold 1967; Koford 1969; Servheen 1990, 1999; Mattson and Merrill 2002; Schwartz et al. 2003b), eventually leading to their listing as a threatened species in 1975. Grizzlies were seen as a threat to livestock and to humans and, therefore, an impediment to westward expansion. The Federal government, as well as many of the early settlers in grizzly

bear country, was dedicated to eradicating large predators. Grizzly bears were shot, poisoned, and killed wherever humans encountered them (Servheen 1999). By the time grizzlies were listed under the ESA in 1975, there were only a few hundred grizzly bears remaining in the lower 48 States in less than 2% of their former range (see FIGURE 1, inset) (FWS 1993). For a detailed discussion of human-caused mortality in the GYA, please refer to pp. 14920-14922 of the Yellowstone Final Rule and to Servheen et al. (2004; 2009). Outside of the GYA, from 1999-2008, a total of 284 known grizzly bear deaths occurred in the lower 48 States. An analysis of mortality sources from the previous decade reflects on-the-ground conditions and provides management priorities. Of these grizzly bear mortalities, 88% (250 / 284) were human-caused, 8.5% (24 / 284) were natural, and 3.5% (10 / 284) were from unknown causes. (USFWS, 2011)

**Stressor:** Genetic status (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The 1975 listing identified the genetic isolation of some grizzly bear populations as a potential threat (40 FR 31734, July 28, 1975). Declines in genetic diversity due are expected in isolated populations, but will occur gradually over decades when populations are large and have long generational times (Frankel and Soule 1981; Ralls et al. 1986; Allendorf et al. 1991; Burgman et al. 1993). Maintaining genetic diversity is important because it provides the raw genetic material with which organisms are able to respond to selective pressures over many generations (i.e., adapt). In general, the more diverse the genetic material, the more likely organisms will be able to adapt to changing environmental conditions successfully. Levels of genetic diversity in grizzly bear populations in the lower 48 States are a potential concern because of small population size in some populations (i.e., SE, CYE, NCASC) and limited genetic exchange with other grizzly bear populations (i.e., SE, NCASC, GYA). (USFWS, 2011)

**Stressor:** Stochasticity, connectivity, and genetic management (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** In addition to the challenges posed by small population size, grizzly bear populations in the lower 48 States are relatively isolated from each other and increasingly fragmented from populations in Canada (Proctor et al. in press). This habitat fragmentation informs our discussion and options for recovery. Because grizzly bears live at relatively low population densities and are vulnerable to excessive human-caused mortality, anthropogenic fragmentation of historically contiguous populations into isolated “remnant” populations is a management reality on the current ecological landscape (Forman and Alexander 1996; Proctor et al. in press; Lindenmayer and Fischer 2006). It is a widely accepted tenet in conservation biology that extinction risk is reduced even through minimal levels of connectivity (Soule 1987). At greatest risk of extinction are small isolated populations with less than 100 individuals like the NCASC. Such populations are more susceptible to extinction due to human-caused mortality and environmental processes such as poor food years, climate change, and habitat loss. While the SE and CYE populations also contain less than 100 individuals each, they are not entirely isolated from Canadian populations. Small populations benefit greatly from both demographic rescue (i.e., the immigration of female bears) and to a lesser degree genetic rescue (i.e., immigration of male bears). Although reconnection of these isolated populations is challenging (Forman and Alexander 1996; Lindenmayer and Fischer 2006), metapopulation theory directs that connectivity is the best long-

term conservation practice to increase the resiliency, redundancy, representation, and overall probability of persistence of remaining grizzly bear populations in the lower 48 States (Boyce 2000). (USFWS, 2011)

**Stressor:** Climate change (USFWS, 2011)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Climate change may generate a number of changes in grizzly bear habitat in the foreseeable future. Whether these changes will translate into population level impacts in grizzly bears in the lower 48 States is unknown but most grizzly bear biologists in the U.S. and Canada do not anticipate that habitat changes predicted under climate change scenarios will directly threaten grizzly bears (Servheen and Cross 2010). The most likely ways in which global climate change may affect grizzly bear habitat include: reduction in snowpack levels; shifts in denning times; shifts in the abundance and distribution of some natural food sources; and changes in fire regimes. These ecological changes may ultimately impact grizzly bears by affecting the timing and frequency of grizzly bear/human interactions and conflicts (Servheen and Cross 2010). (USFWS, 2011)

## ***Recovery***

### **Reclassification Criteria:**

Reclassification criteria are not available.

### **Delisting Criteria:**

1. Meeting the demographic recovery goals that include a minimum number of females with cubs seen annually, distribution of family groups throughout the recovery zones, and a limit on human-caused mortality. Recovery goals for the Yellowstone population include 15 females with cubs over a running 6-year average, 16 of 18 bear management units (BMU's) occupied by females with young from a running 6-year sum of verified sightings and evidence, and less than 4% human-caused mortality. For the North Continental Divide ecosystem, 10 females with cubs inside Glacier National Park (GNP) and 12 females with cubs outside GNP over a running 6-year average, 21 of 23 BMU's occupied by females with young from a running 6-year sum of verified sightings and evidence, and less than 4% human-caused mortality. For the Cabinet-Yaak Ecosystem (CYE), six females with cubs over a running 6-year average; 18 of 22 BMU's occupied by females with young from a running 6-year term, and less than 4% human-caused mortality. For the Selkirks ecosystem (SE), six females with cubs over a running 6-year average, 7 of 10 BMU's on the U.S. side occupied by females with young from a running 6-year sum, and less than 4% human-caused mortality (USFWS, 1993).

2. Demonstrating goals for the NCE and BE currently are being developed and will be appended to this plan when finalized. (USFWS, 1993)

Secure Core Habitat and Motorized Access Management Criterion (USFWS 2018).

Developed Recreation Site Criterion (USFWS 2018).

Livestock Allotment Criterion (USFWS 2018).

**Recovery Actions:**

- Minimize sources of human-bear conflict (USFWS, 1993)
- Limit habitat loss or degradation because of human actions such as road building, timber harvest, oil and gas exploration and development, mining, and recreation. (USFWS, 1993)
- Improve habitat and / or security where applicable. (USFWS, 1993)
- Understand the relationship between bear density and habitat value to better understand limiting factors. (USFWS, 1993)
- Develop techniques to successfully move bears into areas where the populations are in need of augmentation. (USFWS, 1993)
- Improve public relations and education to develop better support for and understanding of the species and to minimize adverse human actions. (USFWS, 1993)
- Continue grizzly bear and habitat research to ensure adequate scientific knowledge is available on which to base management decisions. (USFWS, 1993)

***Conservation Measures and Best Management Practices:***

- Revise the recovery plan for grizzly bears in the lower 48 States so that it reflects the best scientific and commercial information available. The revised recovery plan should include objective, measurable criteria which, when met, will result in a determination that the species be removed from the Federal List of Endangered and Threatened Wildlife. (USFWS, 2011)
- Delist the “error states” to correct the 1975 Listing which listed the grizzly bear throughout the lower 48 States, including areas it was absent from historically. (USFWS, 2011)
- Begin the EIS process to select appropriate conservation actions to recover the NCASC grizzly bear population and evaluate the need for grizzly bear augmentation in this ecosystem. (USFWS, 2011)
- When revising demographic recovery criteria for the NCDE, include mortalities in a 10-mile buffer extending into Canada, as is done with our other trans-border populations in the CYE, SE, and NCASC. (USFWS, 2011)
- Review the Swan Valley Conservation Agreement to reflect new ownership patterns after the Legacy Project. (USFWS, 2011)
- Continue to seek support and resources for a DNA-based population census in the CYE, SE, and NCASC recovery zones. (USFWS, 2011)
- Evaluate the scientific basis for each ecosystem’s definition of Secure Habitat and consider a standardized definition if appropriate. (USFWS, 2011)
- Complete the NCDE Conservation Strategy and implement the habitat standards in the strategy via forest plan amendments and park and Tribal management plan amendments. (USFWS, 2011)
- Implement standardized sanitation regulations on public and private lands in grizzly habitat as a basic conservation measure for multiple wildlife species across the landscape. (USFWS, 2011)
- Enhance conservation action in identified linkage zones between the Purcell Mountains in B.C. southward to the BE and the southward to the GYA. (USFWS, 2011)
- Encourage an annual requirement for bear identification tests for black bear hunters in all States. (USFWS, 2011)
- Continue to fund annual monitoring of population trend and habitat conditions in the NCDE. (USFWS, 2011)
- Obtain funding to enhance the research and monitoring program in the SE and NCASC. (USFWS, 2011)
- Target the most important linkage areas on private lands for conservation delivery through easements, securing attractants, and possible acquisition with willing landowners. (USFWS, 2011)

- Continue to augment the Cabinet Mountains with subadult female grizzly bears in order to increase reproductive output (i.e., demographic rescue). (USFWS, 2011)
- Deliver sanitation enhancement assistance to private residents in grizzly habitat particularly on the periphery of grizzly habitat where grizzly conflicts and mortalities are increasing as bears expand their range. Assistance in the form of bear-resistant garbage containers and electric fencing along with more people to work on increased outreach and education will reduce these conflict and mortality levels. (USFWS, 2011)
- Deliver effective linkage conservation in the Northern Rockies with private and public partners. To accommodate wildlife linkage across the many human-populated valleys, linkage opportunities must be conserved on public lands, private lands found in intervening valleys, and major transportation routes. (USFWS, 2011)
- Hire more law enforcement personnel to control current levels of illegal motorized access and prevent grizzly bear poaching on public lands. (USFWS, 2011)
- Establish a grizzly bear conservation fund to provide a secure funding source for ongoing and future management and monitoring actions. (USFWS, 2011)
- Establish a grizzly bear compensation fund that would continue to compensate livestock operators when grizzlies depredate on their livestock post-delisting. (USFWS, 2011)
- Develop a standardized definition of “suitable habitat” which can be applied in both occupied and unoccupied areas. It may be more credible to use radio-collar data in the definition of suitable habitat as more of those data become available. (USFWS, 2011)
- In accordance with the 1993 Recovery Plan, other areas throughout the historic range of the grizzly bear in the lower 48 States should be evaluated to determine their habitat suitability for grizzly bear recovery (FWS 1993). (USFWS, 2011)
- Continue to follow-up on credible sighting data outside existing range and continue surveys using cameras and DNA hair snares in areas where population expansion is likely. (USFWS, 2011)
- Develop standardized estimates of current grizzly bear range in each ecosystem using methods of Schwartz et al. 2002 and 2006b, or other scientific methodology. (USFWS, 2011)
- With cooperation from the Tribes in the NCDE, update road, habitat security, and livestock data on Tribal lands. (USFWS, 2011)
- Assess the impacts of climate change on vegetative food sources, the distribution and extent of important vegetation communities, and the ability of alpine plant communities and insect communities to continue to exist as these areas are important grizzly bear habitat use areas that will be subject to amplified climate change effects. (USFWS, 2011)
- Continue genetic monitoring to document range expansion and population exchange by obtaining DNA samples from all management and research captured bears. (USFWS, 2011)
- Monitor location and status of radio-collared animals in all ecosystems, using GPS collars when possible. (USFWS, 2011)
- To avoid potential conflicts between snowmobiles and denning grizzlies, National Forests should conduct analyses of suitable denning habitat and spring foraging areas that may overlap with snowmobile use then direct snowmobile use accordingly to minimize conflicts. (USFWS, 2011)
- Calculate the current number of developed sites on both private and public lands within each recovery ecosystem. (USFWS, 2011)
- Obtain habitat data for all public lands outside recovery zones (i.e., road densities, amount of Secure Habitat, number of developed sites, number and type of livestock allotments, etc.). (USFWS, 2011)
- Evaluate the effectiveness of MFWP’s black bear identification test in reducing the number of mistaken identity grizzly bear mortalities from black bear hunters in Montana. (USFWS, 2011)

- Identify areas where movement opportunities still exist in each of the mountain valleys between the Canadian border and the GYA using monitoring data, modeling, and informed decision-making. (USFWS, 2011)
- Monitor effectiveness of ongoing Cabinet Mountains augmentation efforts. (USFWS, 2011)
- Complete stable isotope and bioimpedance analyses annually to document food habits and nutritional status in relation to management status and geographic location. (USFWS, 2011)

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## SPECIES ACCOUNT: *Vulpes macrotis mutica* (San Joaquin kit fox)

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### *Species Taxonomic and Listing Information*

**Commonly-used Acronym:** SJKF

**Listing Status:** Endangered; March 11, 1967 (32 FR 4001). Listed under the Endangered Species Preservation Act in 1967 (USFWS 2010).

### **Physical Description**

The San Joaquin kit fox (*Vulpes macrotis mutica*), is the larger of two subspecies of the kit fox, the smallest canid species in North America. The San Joaquin kit fox, on average, stands about 30 centimeters (cm) (12 inches [in.]) high at the shoulder. The average weight of adult males is 2.3 kilograms (kg) (5 pounds [lbs.]), and of adult females is 2.1 kg (4.6 lbs.). It has a small slim body, large close-set ears, a narrow nose, and a long bushy tail that tapers at the tip. Depending on location and season, the fur coat of the kit fox varies in color and texture from buff to tan or yellowish-grey. The tail is distinctly black-tipped (USFWS 2010).

### **Taxonomy**

The San Joaquin kit fox, *V. m. mutica*, was first described in 1902. Eight subspecies were recognized historically. Today, only *V. m. macrotis* and *V. m. mutica* are recognized subspecies (USFWS 1998). Recent genetic information supports the designation of swift (*V. velox*) and kit fox (*V. macrotis*) as separate species, while supporting the categorization of the San Joaquin kit fox as a subspecies (USFWS 2010). Because all three fox species that occur in the San Joaquin Valley are primarily nocturnal, identification of these free-living and often fast-moving animals can be a challenge. The black-tipped tail and coat color differences usually distinguish kit foxes from red foxes (*V. vulpes*). At 4 to 5 kg (8 to 11 lbs.), the red fox also is much heavier than the kit fox. Gray foxes (*Urocyon cinereoargenteus*), however, are sometimes misidentified as kit foxes, especially in winter when the kit fox coat is thicker and has more gray. Both species have a black tail tip, but gray foxes also have a distinctive black stripe running along the top of the tail. Gray foxes are more robust than kit foxes; they are heavier, with an average body weight of about 3.6 kg (8 lbs.). However, San Joaquin kit foxes have longer ears, averaging 8.6 cm (3.4 in.) compared with 7.8 cm (3 in.) for gray foxes (USFWS 1998).

### **Historical Range**

The subspecies historically ranged in alkali scrub/shrub and arid grasslands throughout the level terrain of the San Joaquin Valley floor, from southern Kern County north to Tracy in San Joaquin County, on the western side of the valley, and up into more gradual slopes of the surrounding foothills and adjoining valleys of the interior Coast Range (USFWS 2010). On the eastern side of the valley, the historic range extended to La Grange, Stanislaus County. By 1930, it was estimated that the kit fox range had been reduced by more than half, with the largest portion of the range remaining in the southern and western parts of the valley (USFWS 1998).

### **Current Range**

San Joaquin kit fox inhabited a portion, but not all, of the areas of suitable habitat remaining in the San Joaquin Valley and lower foothills of the coastal ranges and the Sierra Nevada and Tehachapi mountains. The boundaries of the kit fox's range still extended from southern Kern County north to Contra Costa, Alameda, and San Joaquin counties on the western side of the

valley; and to the La Grange area, Stanislaus County, on the eastern side of the valley. The most northerly sighting was made at the Black Diamond Mines Regional Preserve near Antioch, Contra Costa County, in the early 1990s. The largest extant populations were known from western Kern County, on and around the Elk Hills area and Buena Vista Valley; and the nearby Carrizo Plain Natural Area, where relatively level terrain is separated by narrow rugged ranges (USFWS 1998; USFWS 2010).

**Distinct Population Segments Defined**

N/A; Distinct Population Segments are limited to species, and the San Joaquin kit fox is categorized as a subspecies (USFWS 2010).

**Critical Habitat Designated**

No;

***Life History*****Feeding Narrative**

Adult: The kit fox was thought to subsist primarily on kangaroo rats (*Dipodomys* spp.), and historically populations appear to be most robust where kangaroo rats persist. Studies have shown that kangaroo rat remains comprised 80 to 90 percent of fecal material at most collecting sites throughout the range of the kit fox. During several years of drought, seed resources for granivorous rodents such as kangaroo rats become scarce, resulting in declining abundance of these kit fox prey species. Local extirpation of kit fox communities has also been linked to the previous loss of kangaroo rat populations. High rainfall events also are known to reduce prey abundance dramatically. The kit fox diet currently varies geographically, seasonally, and annually. It includes nocturnal rodents such as kangaroo rats, white-footed mice and pocket mice (*Peromyscus* spp.), California ground squirrels (*Spermophilus beecheyi*), rabbits (*Sylvilagus* spp.) and hares (*Lepus* spp.), San Joaquin antelope squirrels (*Ammospermophilus nelsoni*), and ground-nesting birds. Insects appear to be important seasonal prey items for at least some populations. California ground squirrels were found to be the most common prey item in the Bethany Reservoir area of Alameda County. No kangaroo rats were detected at this site; ground squirrels have also been important food items in some areas where kangaroo rats appeared to be abundant, although the relative density of kangaroo rats in these areas is not known. In eastern Contra Costa County, a crash in the kit fox population was associated with extirpation of the California ground squirrel due to a ground squirrel eradication program. In the Bakersfield vicinity, urban kit foxes have access to anthropogenic food resources to supplement available natural prey, so food is generally abundant and kit fox abundance shows little inter-annual variation (USFWS 2010). In addition, vegetation occurs frequently in feces. Grass is the most commonly ingested plant material (USFWS 1998). In some locations, coyotes only infrequently consume the kit fox they kill, suggesting that coyote attacks are competitive interactions that can include prey consumption rather than a strict predator-prey interaction. The diets and habitats selected by coyotes and kit fox often overlap. Increases in coyote abundance may be a causal factor in past local kit fox declines (USFWS 2010).

**Reproduction Narrative**

Adult: During September and October, adult females begin to clean and enlarge natal or pupping dens. Mating occurs typically in late December or early January, but can occur as late as March, and young are typically born in February or March. Although some yearling female kit

fox will produce young, most do not reproduce until 2 years of age. The median gestation period is estimated to range from 48 to 52 days. Typically, females give birth to two to six pups. The young are born in large natal dens; dens are essential for the survival and reproduction, because the kit fox use them year-round for temperature regulation, shelter from adverse environmental conditions, reproduction, rearing young, and escape from predators. The female is rarely seen hunting during the time she is lactating. During this period, the male provides most of the food for her and the pups. The pups emerge above ground at slightly more than 1 month of age. After 4 to 5 months, usually in August or September, the family bonds begin to dissolve and the young begin dispersing. Occasionally, a juvenile female will remain with the adult female for several more months. Offspring of both sexes sometimes remain with their parents through the following year and help raise a subsequent litter. Reproductive success appears to be correlated with prey abundance. Starvation, especially of pups, was noted to be a likely limiting factor for kit fox populations (USFWS 1998; USFWS 2010). Kit fox exhibit a perennial monogamous social system with generally life-long pair bonds. Kit foxes form pair-mates throughout the year, and the pair-mates continue to associate throughout the year, not just during breeding and pup raising. Pair-bond duration is therefore generally for more than a year; pair-mates that survive to the next breeding season generally remain together. Loss of pairs-mates to mortality, typically due to predation, accounted for dissolution of most pair bonds – due to high mortality rates, few pairs would be expected to last more than three breeding seasons. Red foxes live an average of 8 years; they are more fecund than kit foxes. Kit foxes live an average of 2 years, but may live for 7 or more years, and produce fewer offspring (USFWS 2010). Kit fox in natural habitats generally suffer from high mortality rates due to interference competition from coyote. This social system is determined to increase fitness by enhancing survival and reproductive success in these nonmigratory, territory-holding animals. Remaining on a well-known territory with familiar den locations has been shown to decrease predation risk (USFWS 2010).

**Geographic or Habitat Restraints or Barriers**

Adult: Loss, degradation, and fragmentation of habitat. Studies have shown that kit fox presence is generally negatively associated with ruggedness; kit fox are apparently excluded from steeper terrain by combined factors that influence detection of and increase kit fox susceptibility to predators, especially coyotes, that use these areas and that constitute a significant source of kit fox mortality (USFWS 2010).

**Spatial Arrangements of the Population**

Adult: Clumped

**Environmental Specificity**

Adult: Moderate

**Tolerance Ranges/Thresholds**

Adult: Moderate

**Site Fidelity**

Adult: High

**Dependency on Other Individuals or Species for Habitat**

Adult: In areas with high clay content, San Joaquin kit foxes rely on burrows dug by other animals (USFWS 1998).

**Habitat Narrative**

Adult: Kit fox are an arid-land-adapted species and typically occur in desert-like habitats in North America. Such areas have been characterized by sparse or absent shrub cover, sparse ground cover, and short vegetative structure. Kit foxes have been associated with areas having open, level, sandy ground that is relatively stone-free to depths of about 1 to 1.4 meters (m) (3 to 4.5 feet [ft.]). San Joaquin kit fox are absent or scarce in areas where soils are shallow due to high water tables, impenetrable hardpans, or proximity to parent material, such as bedrock (USFWS 2010). Studies have shown that kit fox presence is generally negatively associated with ruggedness; kit fox are apparently excluded from steeper terrain by combined factors that influence detection of and increase kit fox susceptibility to predators, especially coyotes, that use these areas and that constitute a significant source of kit fox mortality. Current understanding of kit fox habitat indicates that habitat with slopes of less than 5 percent is optimal for the kit fox, and habitat having slopes of greater than 15 percent is unsuitable. At one site, kit foxes were found to be more abundant, and to live the longest when they were located in relatively flat or rolling terrain, suggesting that such terrain likely has the most potential for sustaining viable populations of the species. Highly suitable habitat, consisting of arid scrub and grassland habitats with relatively sparse vegetative cover and slopes under 5 percent, was found to be highly fragmented, with many patches either too small or too isolated to support viable kit fox populations (USFWS 2010). San Joaquin kit foxes use dens for temperature regulation, shelter from adverse environmental conditions, reproduction, and escape from predators. Though kit foxes are reputed to be poor diggers, the complexity and depth of their dens do not support this assessment. Kit foxes also modify and use dens constructed by other animals, such as ground squirrels, badgers, and coyotes, and human-made structures (culverts, abandoned pipelines, and banks in sumps or roadbeds). Den characteristics vary across the San Joaquin kit fox's geographic range. In the southernmost portion, dens with two entrances are most frequently found. Natal and pupping dens, in which pups are born and raised, tend to be larger and have more entrances. Entrances are usually from 20 to 25 cm (8 to 10 in.) in diameter, and normally are higher than wide. Ramp-shaped mounds of dirt from 1 to 2 m (3 to 6 ft.) long are deposited at some den entrances. Kit foxes often change dens, and numerous dens may be used throughout the year (USFWS 1998). Kit foxes establish home ranges that are extensive, but home range sizes vary among locations. Home range size is thought to be related to prey abundance. In the Bakersfield vicinity, kit fox selection of den sites appears to be associated with areas of open space, or areas having light or infrequent disturbance, such as canal rights-of-way and detention basins. Urban kit foxes have access to anthropogenic food sources, and kit foxes in this urban area have smaller home ranges than those in nonurban areas. Kit foxes are also found in the arid and alkaline foothill areas along the western edge and southern part of the San Joaquin Valley, with dominant plant species including saltbush (*Atriplex polycarpa*), iodine bush (*Allenrolfea occidentalis*), tumbleweed (*Amaranthus albus*), alkali heath (*Frankenia salina*), and pickleweed (*Salicornia subterminalis*) widely spaced. Areas in which iodine bush was predominant were known to be poorly drained areas that did not support kangaroo rats and were not apparently used by the kit fox. In the period since listing, studies in various areas of the state have examined kit fox use of and persistence in other habitat types, including grasslands and altered habitat, although information on preferred vegetative types has not changed (USFWS 2010). A study of seven radio-collared kit fox that were radiotracked for up to 14 months has indicated that kit fox are unable to occupy farmland on a long-term basis. Agricultural lands do not provide suitable habitat for the kit fox for a variety of reasons. Lands producing row crops are subjected to weekly inundation during irrigation, which impedes kit fox foraging and

precludes the establishment, maintenance, and use, of earthen dens. Prey abundance is relatively low in row crops, prey diversity is reduced, prey species composition changes, and favored prey species such as kangaroo rats disappear (USFWS 2010).

***Dispersal/Migration*****Motility/Mobility**

Adult: High

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory (NatureServe 2015)

**Dispersal**

Adult: Young disperse in August or September. Average dispersal was 1.9 to 19.3 kilometers (km) (1.2 to 12 miles [mi.]) in one study; another found that the average dispersal was 7.7 km (4.8 mi.) (USFWS 2010).

**Immigration/Emigration**

Adult: Immigration/emigration.

**Dispersal/Migration Narrative**

Adult: San Joaquin kit fox are nonmigratory but appear to disperse readily throughout the range. Kit fox are highly reliant on successful dispersal from population strongholds into suitable habitat to sustain subpopulations. Young typically disperse in August or September, when they are 4 or 5 months old. Successful dispersal appears to be a key factor for the recovery and survival of kit fox, in large part because kit fox populations are becoming more fragmented and are thought to be approaching a metapopulation structure wherein local subpopulations occupy patches of suitable habitat and use the intervening habitat only for movement from one patch to another. Successful dispersal among subpopulations is often thought to maintain genetic diversity, and to rescue declining populations and prevent extinction. However, dispersal does have associated costs that may negatively affect species survival in fragmented landscapes. For the kit fox, animals traveling to unfamiliar areas are more vulnerable to predation, and dispersing juveniles have been shown to suffer high mortality when traveling outside their natal territory. During a 6-year study at the Elk Hills Naval Petroleum Reserves in California (NPRC), pups dispersed an average of  $8 \pm 1.4$  km ( $5 \pm 0.9$  mi.), but the maximum reported distances can vary considerably. One individual traveled 40 km (25 mi.) from its whelping den (USFWS 1998; USFWS 2010). Home range size reflects prey abundance and varies with location and from year to year. Based on several studies, adult home ranges average around 400 to 2,340 hectares (ha) (988 to 5,782 acres [ac.]) (NatureServe 2015). Historically, there was high gene flow among San Joaquin kit fox populations, although additional studies are needed to determine levels of gene flow among subpopulations. Demographic research has suggested that some kit fox populations may be at considerable risk of loss of fitness due to inbreeding depression, even though they may be at greater risk of local extirpation due to either demographic or environmental stochasticity (USFWS 1998).

**Additional Life History Information**

Adult: For the kit fox, animals traveling to unfamiliar areas are more vulnerable to predation, and dispersing juveniles have been shown to suffer high mortality when traveling outside their natal territory (USFWS 1998).

### ***Population Information and Trends***

#### **Population Trends:**

Short-term trend: decline of 10 to 30 percent. Long-term trend: decline of 80 percent (NatureServe 2015).

#### **Species Trends:**

Declining (USFWS 2010)

#### **Resiliency:**

Moderate

#### **Representation:**

Moderate

#### **Redundancy:**

Moderate

#### **Number of Populations:**

16 historic and/or current subpopulations; composed of three core and 13 satellite areas: Western Kern County Core Area (inter-annual fluctuation, slow decline); Carrizo Plains Core Area (inter-annual fluctuation); Ciervo-Panoche Core Area (presumed declining); Alameda, Contra Costa, and San Joaquin counties (decline, no known breeding); Western Merced and Stanislaus counties (decline); Central Merced County (presumed extirpated); Western Madera County (presumed extirpated); Southwestern Fresno County (isolated); Southwestern Kings County (isolated); Southwestern Tulare County (isolated, Pixley National Wildlife Refuge [NWR] extirpated); Tulare County Foothills (unknown); Northwestern Kern County (unknown); Northeast Bakersfield (stable); Metropolitan Bakersfield (stable); Cuyama Valley (unknown, presumed extant); and Salina-Pajaro (Camp Roberts, potentially extirpated; Fort Hunter Liggett, extirpated) (USFWS 2010).

#### **Population Size:**

Unknown (USFWS 2010). 2,500 to 10,000 individuals (NatureServe 2015). Estimate of 7,000 was given in 1975 (USFWS 2010).

#### **Resistance to Disease:**

Low

#### **Adaptability:**

Moderate/high.

#### **Additional Population-level Information:**

Monitoring of kit fox subpopulations has indicated that the occupied range of the kit fox is contracting and increasingly fragmented, and that kit foxes have likely disappeared from areas

of extant habitat in the central and northern portions of their historic range. In many areas, kit foxes appear to have decreased in abundance on a range-wide basis. In some cases, resident family groupings appear to have disappeared from more isolated areas of extant habitat. Kit fox populations are larger in the Bakersfield, Western Kern County, and Carrizo Plains areas than in other portions of the range, but both the western Kern County and Carrizo populations appear to be subject to marked population fluctuations that put them at risk of population loss in less than 10 years in unfavorable environmental and demographic situations. Of all known subpopulations of the kit fox, the Bakersfield animals appear to sustain the most stable population numbers, although the size of this subpopulation is not clear. To date, no comprehensive range-wide surveys have been completed to determine the status of kit fox populations throughout its historic range (USFWS 2010).

**Population Narrative:**

There are 16 subpopulations of the San Joaquin kit fox, consisting of three core areas and 13 satellite areas. Among the three core areas (Western Kern County, Carrizo Plain, and Ciervo-Panoche), distribution/abundance appears to be declining (slow overall decline in Western Kern County; inter-annual fluctuations in Carrizo Plain; and presumed declining in Ciervo-Panoche) (USFWS 2010). In the 13 satellite populations, the current trend is declining in two areas (Alameda, Contra Costa, and San Joaquin County; and Western Merced and Stanislaus counties), presumed/probably/potentially extirpated in three areas (Central Merced County, Western Madera County, and Salinas-Pajaro), isolated in three areas (Southwestern Fresno County, Southwestern Kings County, and Southwestern Tulare County), unknown in three areas (Tulare County Foothills, Northwest Kern County, and Cuyama Valley), and stable in two areas (Northeast Bakersfield and Metropolitan Bakersfield) (NatureServe 2015; USFWS 2010). Currently, the entire range of the kit fox appears to be similar to what it was at the time of the 1998 Recovery Plan. The largest extant population are known to occur in western Kern County on and around the Elk Hills and Buena Vista Valley areas, and in the Carrizo Plain Natural Area, San Luis Obispo County. Though monitoring was not continuous in the central and northern portions of the range, populations were recorded in the late 1980s at San Luis Reservoir, Merced County, North Grasslands, and Kesterson NWR area on the valley floor, Merced County, and in the Los Vaqueros watershed, Contra Costa County. Smaller populations and isolated sightings included other parts of the San Joaquin Valley floor, including Madera County and eastern Stanislaus County (USFWS 2010). However, current population structure has become more fragmented, and at least some of the resident satellite subpopulations, such as those at Camp Roberts, Fort Hunter Liggett, Pixley NWR, and the San Luis NWR, have apparently been locally extirpated. The populations are in the Western Kern County Core Area; Carrizo Plains Core Area; Ciervo-Panoche Core Area; Alameda, Contra Costa, and San Joaquin counties; Western Merced and Stanislaus counties; Central Merced County; Western Madera County; Southwestern Fresno County; Southwestern Kings County; Southwestern Tulare County; Tulare County Foothills; Northwest Kern County; Northeast Bakersfield; and Metropolitan Bakersfield, and is thought to be extant from Cuyama Valley (San Luis Obispo and Santa Barbara counties) and extirpated from Salinas-Pajaro (San Luis Obispo, Monterey, and San Benito counties) (USFWS 2010). As of 1975, the remaining population was believed to include about 7,000 individuals. Subsequent survey data suggest that range-wide kit fox abundance has declined since then (USFWS 2010). The largest remaining population occurs in the Carrizo Plain; in 2000, that population was estimated at 251 to 610 individuals (NatureServe 2015; USFWS 2010). Monitoring of kit fox subpopulations has indicated that the occupied range of the kit fox is contracting and increasingly fragmented, and that kit foxes have likely disappeared from areas

of extant habitat in the central and northern portions of their historic range. In many areas, kit foxes appear to have decreased in abundance on a range-wide basis. In some cases, resident family groupings appear to have disappeared from more isolated areas of extant habitat. Kit fox populations are larger in the Bakersfield, Western Kern County, and Carrizo Plains areas than in other portions of the range, but both the western Kern County and Carrizo populations appear to be subject to marked population fluctuations that put them at risk of population loss in less than 10 years in unfavorable environmental and demographic situations. Of all known subpopulations of the kit fox, the Bakersfield animals appear to sustain the most stable population numbers, although the size of this subpopulation is not clear. To date, no comprehensive range-wide surveys have been completed to determine the status of kit fox populations throughout its historic range (USFWS 2010). Serological surveys of the San Joaquin kit fox and co-occurring carnivores, including the coyote and red fox, have provided evidence of kit fox exposure to pathogens. In serological tests for disease antibodies, high numbers of kit fox test positive for CDV and CPV, indicating that they have been exposed to these diseases. CDV and CPV could be sources of mortality in kit fox populations, but population-level effects have not been studied. Infectious canine hepatitis virus, CDV, CPV, *Leptospira interrogans*, and *Toxoplasma gondii* were found in varying percentages of adult kit fox; however, only one of eight juveniles tested was positive for antibodies (to *L. interrogans*) (USFWS 2010).

### ***Threats and Stressors***

**Stressor:** Conversion of land to agricultural land

**Exposure:** Conversion of land to agricultural land.

**Response:** Burial, displacement, human disturbance, fire suppression, and pest control.

**Consequence:** Mortalities, displacement, reduction of prey populations and denning sites, changes in the distribution and abundance of larger canids that compete with kit fox for resources, and reduction in carrying capacity.

**Narrative:** Conversion of natural lands to agriculture has continued since the kit fox was listed. By 1979, most of the San Joaquin Valley floor had been developed, with approximately 150,000 ha (370,000 ac.) out of a total of approximately 3.4 million ha (8.5 million ac.) remaining undeveloped. Land conversions contribute to declines in kit fox abundance through direct and indirect means: mortalities, displacement, reduction of prey populations and denning sites, changes in the distribution and abundance of larger canids that compete with kit fox for resources, and reductions in carrying capacity. Kit fox may be buried in their dens during land conversion activities, or permanently displaced from areas where structures are erected or the land is intensively irrigated. In addition to the direct loss of habitat for denning and foraging by kit fox, land conversion and associated human-intensive uses can bring additional stressors, including human disturbance, fire suppression, and pest control (USFWS 2010). The conversion of natural lands to agriculture continues to be a threat on private lands on the western side of the San Joaquin valley floor in areas where agriculture has been extended west to the base of the foothills since the 1960s. Past agricultural conversion has removed most areas of the valley floor as kit fox habitat. However, conversion of natural habitat to intensive agriculture continues to be the primary cause of habitat loss for the San Joaquin kit fox in the San Joaquin, Salinas, and associated valleys, and in adjacent foothill areas. Agricultural lands do not appear to be suitable habitat for long-term kit fox persistence due to practices including soil cultivation, frequent irrigation, and use of agricultural chemicals and pesticides, and due to altered prey and predator communities. Loss and modification of habitat due to agricultural use continues to be a primary threat to kit fox (USFWS 2010).

**Stressor:** Loss of habitat due to urbanization

**Exposure:** Population growth and urban development.

**Response:** Loss or degradation of habitat, and restriction of core habitat and movement corridors.

**Consequence:** Loss of habitat, fragmentation, and reduction in population numbers.

**Narrative:** The increasing human population of California, with the concomitant high demand for limited supplies of land, water, and other resources, has been identified as the primary underlying cause of habitat loss and degradation. Between 1970 and 2000, the human population of the San Joaquin Valley doubled in size; it is expected to more than double again by 2040. In the San Joaquin Valley, the continued increase in the human population has resulted in increased urban development. On the floor of the valley, urbanization occurs most often on previously cultivated lands, where natural habitat has been lost or degraded. However, urbanization is also occurring along all edges of the San Joaquin Valley in areas of extant natural habitat that is important to the kit fox. In these areas, cities that are undergoing substantial growth include but are not limited to: Livermore, Antioch, Tracy, and Los Baños, in the northwestern portion of the kit fox's range; and Paso Robles, Tulare, and Bakersfield in the southern portion of the range. Development along the San Joaquin Valley periphery and in adjacent valleys, such as the Salinas Valley, continues to restrict both core habitat and movement corridors for the kit fox (USFWS 2010).

**Stressor:** Habitat loss and modification due to oil extraction and mining activities

**Exposure:** Oil exploration, and spills.

**Response:** Human disturbance, loss of habitat and den sites, entombment, entrapment in sumps or oil spills, exposure to contaminants, changes to remaining habitat, changes in predator and prey community composition and abundance, disruptions to migration and foraging.

**Consequence:** Reduction in habitat, mortality, reduction in prey, toxicity due to oil, and reduction in population numbers.

**Narrative:** Currently, oil extraction and gravel mining may pose both direct and indirect risks to the San Joaquin kit fox. Direct risks to kit fox from oil-field development include human disturbance, loss of habitat and den sites, entombment, entrapment in sumps or oil spills, and exposure to contaminants. San Joaquin kit fox appear to tolerate human activities; they have frequently been observed around facilities and are known to use manmade structures (pipe, culverts, and foundations) as dens, although with some mortality. Indirect effects of oil field development on kit fox include changes to remaining habitat, and changes in predator and prey community composition and abundance. Oil spills may create short-term disruptions of primary travel routes and foraging areas for kit fox. Short-term effects of oil spills have included a 67 percent decrease in abundance of Heermann's kangaroo rats (*Dipodomys heermanni*) between spill areas and control areas. Similarly, oil field disturbances in western Kern County have been found to result in shifts in the small mammal community from the primarily granivorous (seed-eating) species (kangaroo rats) that are a staple prey of kit fox, to species adapted to disturbed areas (murid, or old world rodents). The effect of an altered prey community on the energetics of the kit fox is not currently known, but early studies suggest that such altered prey composition may result in lower kit fox density (USFWS 2010). Currently, the southern half of the San Joaquin Valley continues to be an area of expansion and development activity for extraction of petroleum products. Recent and continuing oil and gas leases are being offered within the range of the kit fox in Kern, Kings, Fresno, San Benito, and Monterey counties, where they have the potential to affect kit fox habitat and dispersal corridors. In addition, in the Carrizo National Monument,

Vintage Production LLC, a subsidiary of Occidental Petroleum, recently submitted a permit request to the Bureau of Land Management (BLM) to explore for oil on 12,000 ha (30,000 ac.) of subsurface mineral holdings in the heart of the Monument's valley floor grasslands. Although exploration could set the stage for negotiations to purchase the oil rights, it is also possible that exploration will result in development of oil resources in high-value kit fox habitat (USFWS 2010).

**Stressor:** Habitat loss, modification, and fragmentation due to construction of solar facilities

**Exposure:** Development of solar facilities.

**Response:** Fragmentation of habitat, significant restriction of the kit fox's range, and barriers to linkages/dispersal.

**Consequence:** Loss of habitat, reduction in population numbers, and reduction in dispersal.

**Narrative:** A number of large-scale solar development projects that would threaten kit fox population clusters are currently proposed for construction in kit fox habitat. In the Carrizo Core Area, two solar firms propose to install solar panels on 34 square kilometers (13 square miles) of land on the valley floor of the Carrizo Plain, San Luis Obispo County, just north of the Carrizo Plain National Monument. Although this area of the Carrizo has a fair amount of dryland farming and is less likely to be optimal kit fox habitat than land in the National Monument, these projects will create barriers to the linkage between the Carrizo Plain Core Area, the Western Kern core area, and core and satellite areas to the north and west, thereby impeding kit fox dispersal and increasing habitat fragmentation. In the Ciervo-Panoche core area, two large, utility-scale, solar farms that will cover approximately 4,500 ha (11,000 ac.) of valley floor habitat in the Panoche and Little Panoche Valleys (essentially all flatland habitat), are being proposed. Consultation between project proponents and state and federal wildlife agencies has not yet been completed, but preliminary maps of the proposed projects suggest that most suitable habitat in the area would be developed, leading to a significant restriction of the kit fox's range. One 65-ha (160-ac.) solar project is proposed for the Cuyama Valley. Although loss of currently occupied habitat may be minimal because the land is currently in row crop production, the proposed solar project would limit opportunities for future restoration. U.S. Fish and Wildlife Service (USFWS) expects that additional solar projects will be proposed on lands important to the kit fox at the southern extent of its range (USFWS 2010).

**Stressor:** Habitat loss, modification, and fragmentation due to construction of infrastructure

**Exposure:** Construction of roads, canals, reservoirs, water banks, sound walls, and similar facilities.

**Response:** Vehicle strikes, modification of land-use patterns, and barriers to movement.

**Consequence:** Reduction in habitat, mortality, and reduction in population numbers.

**Narrative:** Construction of infrastructure projects continues to result in the direct loss and indirect modification of remaining kit fox habitat throughout the range of the kit fox. Paved roads, canals, reservoirs, water banks, sound walls, and similar facilities present both permanent loss of habitat and potential barriers to kit fox movement that fragment habitat. Linear infrastructure features that accompany development, such as roads, freeways, and canals, have the potential to disrupt or stop the movement of a variety of mammals, including kit fox and their prey. This fragments the remaining suitable habitat into patches, where patch size affects the ability of the patch to support larger species and species that are less tolerant of human disturbance. Natural recovery following such declines can be difficult if community conditions have been altered. Overall, the effects of roadways on kit fox movement vary depending on the location, size, and volume of vehicle use. However, in urban areas such as Bakersfield, the effect of higher volume roads on kit fox dispersal is not clear, but does result in at least some mortality,

thereby presenting at least a partial barrier to connectivity of kit fox. Four-lane highways with median barriers generally present impermeable barriers to movement of the kit fox compared to rural roadways. Road construction in the San Joaquin Valley has resulted in the loss of kit fox habitat since listing. Rough calculations of the acreage of land lost to road development indicate that by 2003, more than 3,800 ha (7,000 ac.) of land had been transferred to California Department of Transportation jurisdiction, including 1,500 ha (3,670 ac.) of land in Kern County, 240 ha (590 ac.) in Kings County, 431 ha (1,065 ac.) in Merced County, and 820 ha (2,020 ac.) in Fresno County (USFWS 2010).

**Stressor:** Habitat alteration due to fires

**Exposure:** Wildfire

**Response:** Changes in habitat, changes in foraging, and vulnerability to predation.

**Consequence:** Mortalities, displacement, and changes in the distribution.

**Narrative:** Wildfires have the potential to alter kit fox habitat, and could either negatively or positively affect kit fox persistence. Wildfires may increase under drought conditions or with increasing human populations and habitat change. In addition, prescribed burns may be used to control shrub growth. Fires may directly endanger individual kit fox, although the magnitude of this threat is expected to be relatively low in typical kit fox habitat, which is characterized by sparse vegetation. The threat to individual fox is expected to be higher in grassland habitats or where exotic grasses, or shrub overgrowth, carry fire into native habitat. However, kit fox that must relocate their areas of foraging within their home range in response to fires become more vulnerable to predation as they relocate. Wildfires are known to occur within the range of the kit fox. In 1998, a major wildfire burned through the Lokern Natural Area, destroying shrublands. Smaller repeated fires also occur on the landscape, resulting in expanded areas of grassland habitat due to the failure of saltbush scrub to regenerate. Wildfires commonly occur on the western hills of Kern and Kings counties, and into the Tumey, Ciervo, and Panoche Hills. The BLM uses prescribed fire on 160 to 810 ha (400 to 2,000 ac.) every 3 to 5 years in the Carrizo Plain. Military Reserves, such as Fort Hunter-Liggett, also use prescribed fire to control vegetation so that military operations do not ignite wildfires (USFWS 2010).

**Stressor:** Habitat alteration due to changes in vegetation structure from growth of nonnative vegetation, and altered grazing regimes

**Exposure:** Overgrazing and reduction or cessation of grazing.

**Response:** Coyote predation and reduced prey base.

**Consequence:** Mortality, starvation, reduction in population numbers, and reduction in habitat.

**Narrative:** In the period since the kit fox was listed, grazing practices that result in either overgrazed areas or in relatively high vegetative structure have been proposed as potential threats to kit fox by either reducing their prey base or increasing their vulnerability to predation. Kit fox are more vulnerable to coyotes in dense vegetation. Arid grassland habitat with low vegetative structure, common patches of bare ground, and abundant kangaroo rats is recognized as optimal habitat for the kit fox. In contrast, lands that develop dense stands of vegetation higher than approximately 46 cm (18 in.) are expected to result in increased predation risk for the kit fox. Nonnative grasses have become the dominant herbaceous component in many California habitats. In such grasslands, reduction or cessation of grazing has been demonstrated to result in conditions unsuitable for the kit fox under some conditions (e.g., where precipitation and soil conditions allow dense vegetative growth). In addition to nonnative grasslands, parcels of vacant or retired lands often harbor dense growths of weedy species (e.g., mustards [*Brassica nigra* and *Sisymbrium irio*], five-hook bassia [*Bassia hyssopifolia*], and silverscale [*Atriplex*

argentea]) that render habitat unsuitable for kit fox. Altered vegetative structure can also affect the availability of the kit fox's prey base, particularly for kangaroo rat species. Grazing effects on kangaroo rats appear to be mixed, and USFWS expects that grazing may either negatively or positively affect kangaroo rats, depending on the particular site conditions, grazing level, annual weather regime, and the particular species involved. Although kangaroo rats depend on open areas for burrow construction, they also consume seeds, and research on grazing effects suggests potential benefits to kangaroo rats of a mix of ungrazed and grazed habitats (USFWS 2010).

**Stressor:** Disease

**Exposure:** Rabies, canine parvovirus, and canine distemper virus.

**Response:** See narrative.

**Consequence:** See narrative.

**Narrative:** Wildlife diseases (rabies, canine parvovirus [CPV], canine distemper virus [CDV], etc.) could cause substantial mortality or contribute to reduced fertility in female kit foxes. Diseases may threaten long-term viability of small populations of wildlife. Although high numbers of kit fox test positive for CDV and CPV, indicating that they have been exposed to these diseases, past studies have not observed clinical indications of these diseases nor found evidence that disease was an important mortality factor where it was studied. Disease and predation may have both contributed to the catastrophic decline in the isolated population of San Joaquin kit fox at Camp Roberts, in San Luis Obispo County. Kit fox captures decreased from 103 in 1988 to 20 in 1991, and decreased further to only three in 1997. During this same period captures of striped skunks (*Mephitis mephitis*) also generally decreased, but the proportion of skunks that were found to be rabid increased. This correlation led biologists to propose that rabies was a factor in the kit fox decline (USFWS 2010).

**Stressor:** Predation and competition

**Exposure:** Predation and competition from coyotes, red foxes, and domestic dogs.

**Response:** Predation, competition, and reduction in prey.

**Consequence:** Mortality, out-competed, reduction in useable habitat, and reduction in population numbers.

**Narrative:** Predation of kit fox by large canid predators, including the coyote (*Canis latrans*) and nonnative red fox (*Vulpes vulpes*), appears to be a major and increasing threat to the viability of kit fox populations. In most areas of the kit fox's range, coyotes are the primary cause of kit fox mortality, and survival rates of kit fox decrease significantly as coyote-caused mortality increases. Canid predators have increased both in distribution and abundance with the increased land conversion, presence of water sources, and related human activities in the San Joaquin Valley. Abundant coyote populations currently appear to be excluding kit fox from some protected kit fox habitat (USFWS 2010). In addition to direct mortality, coyotes and red fox also negatively affect kit fox by competing for prey resources, and by competing with kit fox for habitat and/or denning resources. Increased presence of wild and domestic canids that pose an increasing threat to kit fox may be due to human-associated changes in the natural environment. The diets and habitats selected by coyotes and kit fox often overlap. Coyotes apparently threaten kit fox on lands that are protected to provide natural habitat. In recent years, coyotes have increased in density at some conservation lands that have been protected for the kit fox and other listed upland species (USFWS 2010). Nonnative red fox are known to kill kit fox, displace kit fox from dens, and compete with them for habitat and prey resources. Nonnative red fox are close to kit fox both morphologically and taxonomically, which could result in more intense competitive interactions, including predation. Red foxes also live longer than kit foxes, with a lifespan of 8

years compared to an average of 2 years in kit fox. The predation threat posed by domestic canids is thought to be small, but has not been quantified (USFWS 2010).

**Stressor:** Inadequacy of existing regulatory mechanisms

**Exposure:** See narrative.

**Response:** See narrative.

**Consequence:** See narrative.

**Narrative:** The Endangered Species Act (ESA) is the primary federal law that provides protection for this species. The California ESA provides protection against take of the species, but the definition of take is more limited than that provided under ESA and does not protect the kit fox from significant modification of habitat. Other federal and state regulatory mechanisms provide discretionary protections for the species based on current management direction, but do not guarantee protection for the species absent its status under ESA. There are several state and federal laws and regulations that are pertinent to federally listed species, each of which may contribute in varying degrees to the conservation of federally listed and nonlisted species. These laws, most of which have been enacted in the past 30 to 40 years, have greatly reduced or eliminated the threat of wholesale habitat destruction, although the extent to which they prevent the conversion of natural lands to agriculture is less clear. Therefore, we continue to believe other laws and regulations have limited ability to protect the species in the absence of the ESA (USFWS 2010).

**Stressor:** Rodenticides and pesticides

**Exposure:** Consumption of rodenticides and pesticides, or exposure throughout their environments.

**Response:** Decline in rodent populations, and consumption of rodenticides and pesticides.

**Consequence:** Mortality, illness.

**Narrative:** At the time of listing, early generation poisons—such as compound 1080 and strychnine—were used as pesticides for predator and rodent control and were considered a threat to kit fox. Pesticides, and specifically rodenticides, pose a threat to kit fox through direct or secondary poisoning. For example, kit fox may be killed if they ingest rodenticide in a bait application, or if they consume rodents that have consumed bait. Kit fox may also be threatened by loss of prey if rodent prey populations decline due to rodent control programs, or if availability of insect prey is substantially reduced by insecticide treatments, especially if insect prey declines occur when overall prey resources are limited. There also is the potential that availability of den sites may be impacted by rodent control programs, because kit fox can depend on ground squirrels to create potential burrows in areas with hardpan soil layers. Although kit foxes have been excluded from large portions of agricultural lands, they currently use agricultural lands that border natural lands. In 1997, the California Department of Pesticide Regulation listed approximately 400 pesticides for which at least one use occurred within 1.6 km (1 mi.) of kit fox habitat, warranting further evaluation of potential effects to the kit fox. Pesticides used in close proximity to kit fox habitat include the following: Malathion, aldicarb, carbaryl, chlorpyrifos, lindane, parathion, and the anticoagulant rodenticides brodifacoum, chlorophacinone, and diphacinone. Currently, both first-generation anticoagulant rodenticides (FGARs) and second-generation anticoagulant rodenticides (SGARs) may be used as rodent control agents within the range of the kit fox, although the appropriate use of individual anticoagulants differs depending on the terms of their registration. FGARs include warfarin, chlorophacinone, and diphacinone, while brodifacoum, bromadiolone, difethialone, and difenacoum are considered SGARs. Both FGARs and SGARs interfere with blood clotting, leading to death from hemorrhaging. FGARs

require several days of consecutive feedings to deliver a lethal dose to the target species, while SGARs can deliver a lethal dose in only one night of feeding (USFWS 2010).

**Stressor:** Selenium and other contaminants

**Exposure:** Selenium toxicity.

**Response:** Reduction in prey abundance, and bioaccumulation.

**Consequence:** Reduction in food source, and selenium toxicity.

**Narrative:** Selenium toxicity may pose a threat to the kit fox in some areas on the western side of the San Joaquin Valley where federal water is delivered to the San Luis Unit; and where local conditions result in elevated concentrations of selenium in soil and surface water, or in near-surface groundwater. In these areas, naturally occurring selenium has been concentrated in surface waters due to drainage from agricultural areas. These localities can include retired or fallowed seleniferous farm land, open ditches that convey subsurface drain water, and drain water reuse projects. Selenium has the potential to bio-accumulate in aquatic organisms, such as zooplankton and benthic invertebrates, and may then biomagnify as it reaches top level predators, including birds, mammals, and fish. Cover-cropping systems proposed for reuse areas include crops, such as grain crops and pasture lands, that may support substantial prey resources although some areas may be grazed, which would reduce prey abundance. The selenium applied to these reuse areas via agricultural drain water can enter the food chain through uptake by plants and soil invertebrates, where it may be bio-accumulated by the seed- and invertebrate-eating organisms that comprise typical kit fox prey (USFWS 2010).

**Stressor:** Prey availability

**Exposure:** Lack of prey availability, and use of rodenticides/insecticides.

**Response:** Prey scarcity.

**Consequence:** Decreased reproductive success, and extirpation.

**Narrative:** Kit foxes have been strongly linked ecologically to kangaroo rats, with kit fox densities and population stability highest in areas with abundant kangaroo rats. Abundance of prey species, particularly abundance of kangaroo rats, has been linked with successful recruitment of young kit foxes and increases in kit fox population numbers. Conversely, prey scarcity has been a primary factor contributing to decreased reproductive success during droughts, or to extirpation of kit fox in specific localities. Studies suggested that kangaroo rats were a preferred food for the kit fox throughout the range, and that kit fox densities were lower in areas like those near Bakersfield where plant associations changed and abundant ground squirrels replaced kangaroo rats. In addition to rodents, insects can be important prey for the San Joaquin kit fox, especially during periods of low prey availability. In the northern portion of the kit fox's range, insects, especially grasshoppers and crickets, currently provide the primary prey for kit foxes during the summer months, particularly July and August. Insecticides that target grasshoppers and crickets (Orthoptera spp.) may suppress kit fox populations, reduce juvenile survivorship, or inhibit successful dispersal (USFWS 2010).

**Stressor:** Inbreeding depression, genetic drift, and stochastic extinction.

**Exposure:** See narrative.

**Response:** See narrative.

**Consequence:** Extirpation

**Narrative:** Small populations may be subject to inbreeding depression and genetic drift, and also to chance extinction from stochastic environmental and demographic incidents. Demographic research has suggested that kit foxes may be susceptible to inbreeding depression, and that they

are threatened by local extirpation due to stochastic events. It appears that at least several of these small and isolated resident subpopulations have recently “winked out” (become locally extinct), including subpopulations at the Fort Hunter Liggett military reserve, and at San Luis and Pixley (USFWS 2010).

**Stressor:** Vehicle strikes

**Exposure:** Kit foxes being hit by cars.

**Response:** See narrative.

**Consequence:** Mortality, injury, and population decline.

**Narrative:** Vehicle strikes are a consistent, but small source of kit fox mortality on natural lands. Although impacts of roads on kit fox ecology are generally thought to be low, mortality due to vehicle strikes may significantly affect small populations. Although vehicle strikes may not have population-level effects in natural lands where traffic volume is low, they appear to be a more substantial source of mortality in human-altered landscapes, including urban environments. In urban settings such as Bakersfield, vehicle strikes can be the largest source of kit fox mortality, and may impact urban kit fox populations (USFWS 2010).

**Stressor:** Accidental shooting

**Exposure:** Intentional or incidental shooting.

**Response:** See narrative.

**Consequence:** Reduction in population numbers; mortality and injury.

**Narrative:** In the past, state regulations, such as restrictions on night hunting and spotlighting, were promulgated to reduce the potential for intentional and incidental shooting of kit fox. Although threats have been reduced, it appears that kit fox are still subject to accidental and illegal shooting throughout most of their range. Kit fox may potentially be mistaken for other wild canids, especially coyotes (*Canis latrans*), but naïve hunters could also potentially mistake kit fox for gray fox or red fox (*Vulpes vulpes*). Kit fox superficially resemble juvenile coyotes, suggesting that kit fox may be particularly vulnerable to misidentification at particular times of the year. Both the coyote and the gray fox are nongame species that may be taken in any number. Although the coyote may be taken all year, hunting gray fox is restricted to a season that runs from November 24 through February. A closure on night hunting is in effect in those portions of the species' range in Monterey and San Benito counties lying east of Highway 101, but legal in the rest of the range. Coyote hunting by people using predator calls, and by sheepherders, has been reported in lands surrounding the former NPR-1. Documented kit fox mortality due to shooting occurs occasionally on both public and private lands, including protected lands. In addition, kit fox harassment in association with hunting has been reported (USFWS 2010).

**Stressor:** Off-road vehicle (ORV) use

**Exposure:** Use of ORVs.

**Response:** Disturbance of soil, reduction of herbaceous vegetation, destruction of burrow systems of prey species such as the kangaroo-rat, and damage to kit fox dens.

**Consequence:** See narrative.

**Narrative:** Use of ORVs poses an unquantified threat to the San Joaquin kit fox, primarily through the potential for off-road travel to disturb soil, reduce or destroy herbaceous vegetation, destroy burrow systems of prey species such as the kangaroo-rat, and damage kit fox dens. Off-road travel also increases access to areas that are otherwise remote and little used. Off-road travel is expected to increase impacts to animals on large expanses of natural lands, including both publicly and privately held lands. The increase in ORV use in this area appears to be an increasing

threat to the kit fox in otherwise suitable habitat. Although effects on habitat have not been quantified in large portions of the western Kern County area, in specific areas the recent increased use of ORVs has substantially degraded soil and vegetation conditions on lands targeted for conservation (USFWS 2010).

**Stressor:** Climate change

**Exposure:** Change in climate.

**Response:** See narrative.

**Consequence:** See narrative.

**Narrative:** Climate researchers list three clear, observable connections between climate and terrestrial ecosystems, such as those inhabited by the kit fox: seasonal timing of life-cycle events (phenology), responses of plant growth, and biogeographic distributions of plant and animal species. Current climate change predictions for terrestrial areas in the northern hemisphere indicate warmer air temperatures, more intense precipitation events, and increased summer continental drying. Kit fox subpopulations are threatened by both droughts and high rainfall events. Kit fox subpopulations, including the relatively large subpopulations at the NPRC and Carrizo Plains areas, demonstrate large fluctuations in abundance in response to weather-mediated prey levels, which increases the potential for these groups to be extirpated. Weather conditions usually vary over larger landscape scales, leading to the general expectation that drought-mediated decreases in kit fox abundance, or local extirpation of some groups, should not affect persistence of the species as long as healthy core kit fox populations are not limited to one portion of the range. However, the loss and fragmentation of habitat documented herein has reduced the likelihood that lost sites will be recolonized. Because increased drying and droughts, and substantial precipitation events, are expected to negatively affect the native prey species on which the kit fox depends, USFWS expects climate change to pose a substantial threat to the species by further exacerbating interannual fluctuations in kit fox reproductive success and abundance (USFWS 2010).

**Stressor:** Research-related activities

**Exposure:** Research activities.

**Response:** See narrative.

**Consequence:** See narrative.

**Narrative:** A limited amount of mortality has been documented to occur due to research activities. During monitoring of 542 radio-collared kit fox at the NPRC between 1980 and 1991, seven suffered minor injuries, while one suffered a lethal injury when its front paw became trapped in the collar. Newly collared adults lost body mass compared to uncollared adults, consistent with collaring effects observed in other species, but the long-term effects of this difference have not been determined. In general, these research-related effects on kit fox appear to have few population-level consequences, but could potentially be important to dynamics of a small subpopulation (USFWS 2010).

## ***Recovery***

### **Reclassification Criteria:**

Reclassification of the kit fox to threatened status may occur when the following criteria are met:

Secure and protect specified recovery areas from incompatible uses: a. The three core populations: Carrizo Natural Area, western Kern County, and Ciervo-Panoche Area; and b. Three satellite areas (USFWS 1998; USFWS 2010).

Management plans that include survival of the kit fox as an objective are approved and implemented for all protected areas identified as important to continued survival of the kit fox (USFWS 1998; USFWS 2010).

Population monitoring in the specified recovery areas shows: a. Stable or increasing populations in the three core areas through one precipitation cycle;\* and b. Population interchange between one or more core populations and the three satellite populations (USFWS 1998; USFWS 2010).

\*A precipitation cycle is defined as “a period when annual rainfall includes average to 35 percent above average through greater than 35 percent below-average and back to average or greater. The direction of change (average to above or below average) is unimportant in this criterion.” A stable population is one in which population size remains statistically the same during the average phase of a precipitation cycle (anticipated to be about 20 years). Increasing population size means that the population has increased over the previous or baseline year, measured during the specified portion of a precipitation cycle. Range-wide population monitoring programs would have to be established to measure progress in meeting recovery criteria (USFWS 1998; USFWS 2010).

**Delisting Criteria:**

These criteria provide for delisting of the kit fox. The Recovery Plan states that delisting criteria include meeting all of the reclassification/downlisting criteria. It also specifies a protection level for the kit fox that provides an extinction probability of 5 percent for 300 years for the entire population of the San Joaquin kit fox (USFWS 1998; USFWS 2010).

In addition to the satellite areas protected under reclassification/downlisting criteria, secure and protect from incompatible uses several additional satellite populations (number dependent on the results of research), encompassing as much as possible of the environmental and geographic variation of the historic geographic range (USFWS 1998; USFWS 2010).

Management plans that include survival of the kit fox as an objective are approved and implemented for all protected areas identified as important to continued survival of the kit fox (USFWS 1998; USFWS 2010).

Population monitoring in the specified recovery areas shows stable or increasing populations in the three core areas and three or more of the satellite areas during one precipitation cycle\* (USFWS 1998; USFWS 2010).

**Recovery Actions:**

- Develop and implement a regional cooperative program and participation plan (USFWS 1998).
- Protect and secure existing populations (USFWS 1998).
- Determine distributions and population status (USFWS 1998).
- Conduct important research and monitoring (USFWS 1998).

- Maintain and establish linkages in existing natural lands and between islands of habitat on the valley floor and natural lands around the fringe of the valley (USFWS 1998).
- Apply adaptive management in protected areas (USFWS 1998).
- The 1998 Recovery Plan identified core and satellite areas where subpopulations of kit fox occur. However, baseline mapping and quantification of the extant habitat remaining in each core and satellite area at the time of Recovery Planning has not yet been completed. Mapping efforts that quantify the acreage of suitable/native habitat and altered or degraded habitat in core, satellite, and linkage areas at 1) the time of the 1998 Recovery Plan; and 2) the current time will assist USFWS and other conservation entities in prioritizing conservation strategies and in determining progress in meeting recovery goals for protection of core and satellite areas. The locations, acreage, and quality (or characteristics) of protected habitat could also be compiled and mapped (USFWS 2010).
- Studies that assist in determining the population-level effects of contaminants, including FGARs and SGARs, on kit fox or surrogate species are needed. Studies that test correlations between rodenticide use and kit fox population parameters, measure sublethal effects on behavior, or quantify rodenticide/pesticide effects on availability of prey in relation to the energetic needs of the kit fox would provide information useful to recovery actions (USFWS 2010).
- Focus land acquisitions on the establishment of large blocks of land (at least 10,000 ac. in size) on the San Joaquin Valley floor and western fringes. Such large parcels are critical to supporting sustainable populations of kit fox for long-term conservation, and should be linked with protected broad dispersal corridors. These acquisitions are most likely to aid kit fox recovery if they build on existing protected lands to achieve larger expanses of protected land, if acquired lands possess the vegetative structure and native prey base that are associated with thriving kit fox populations, and if acquired lands are not isolated from extant populations of either the kit fox or its prey species. Large holdings of native habitat are also expected to be less suitable for coyotes and red fox that are responsible for high levels of kit fox mortality. Lands no longer suitable for agriculture, such as those targeted for land retirement, may be restored and conserved through fee title acquisition, conservation easement acquisition, or conservation banking arrangements from willing sellers or participants. However, on suboptimal habitat, conservation planning should recognize the lag times inherent in restoration of the ecological community needed to support the kit fox. Linkages will be most effective in contributing to kit fox recovery where they link to habitat that retains the characteristics needed to sustain resident populations (USFWS 2010).
- A range-wide census of kit fox should be conducted using a methodology that ensures statistically significant data collected for all areas. Collaboration with the U.S. Geological Service on methods that use occupancy models may be a promising approach, but additional consideration is needed. Some biologists have suggested that more northerly satellite areas and/or linkages have become population sinks for the kit fox, and this possibility merits further study to determine what factors contribute to population status in these areas, and how these factors may be altered to promote range-wide recovery. The amount of gene flow between subpopulations of the kit fox should be confirmed using appropriate methods, adequate sample size, and inclusion of subpopulations of interest, including isolated groupings in the valley center and subpopulations occurring along the western side of the valley (USFWS 2010).
- Consultations on the location of solar facilities may wish to consider lands that are drainage-impaired and that may not constitute suitable habitat for the kit fox due to level of

groundwater present, condition of site vegetation, presence and density of preferred prey species, and isolation from other suitable habitat. These lands may be a potential alternative to development of solar facilities in areas north of the Carrizo Plain and Panoche Valley (USFWS 2010).

***Conservation Measures and Best Management Practices:***

- The following measures are presented in the USFWS' Standardized Recommendations for Protection of the Endangered San Joaquin Kit Fox Prior to or During Ground Disturbance (USFWS 2011). This document contains additional recommendations for small projects, recommended exclusions zones, and destruction of dens that are not discussed below. Habitat subject to permanent and temporary construction disturbances and other types of ongoing project-related disturbance activities should be minimized by adhering to the following activities. Project designs should limit or cluster permanent project features to the smallest area possible, while still permitting achievement of project goals. To minimize temporary disturbances, all project-related vehicle traffic should be restricted to established roads, construction areas, and other designated areas. These areas should also be included in preconstruction surveys and, to the extent possible, should be established in locations disturbed by previous activities to prevent further impacts.
- 1. Project-related vehicles should observe a daytime speed limit of 20 miles per hour (mph) throughout the site in all project areas, except on county roads and state and federal highways; this is particularly important at night when kit foxes are most active. Night-time construction should be minimized to the extent possible. However, if it does occur, then the speed limit should be reduced to 10-mph. Off-road traffic outside of designated project areas should be prohibited (USFWS 2011).
- 2. To prevent inadvertent entrapment of kit foxes or other animals during the construction phase of a project, all excavated, steep-walled holes or trenches more than 2 ft. deep should be covered at the close of each working day by plywood or similar materials. If the trenches cannot be closed, one or more escape ramps constructed of earthen-fill or wooden planks shall be installed. Before such holes or trenches are filled, they should be thoroughly inspected for trapped animals. If at any time a trapped or injured kit fox is discovered, USFWS and the California Department of Fish and Wildlife (CDFW) shall be contacted as noted under measure 13, referenced below (USFWS 2011).
- 3. Kit foxes are attracted to den-like structures such as pipes, and may enter stored pipes and become trapped or injured. All construction pipes, culverts, or similar structures with a diameter of 4 in. or greater that are stored at a construction site for one or more overnight periods should be thoroughly inspected for kit foxes before the pipe is subsequently buried, capped, or otherwise used or moved in any way. If a kit fox is discovered inside a pipe, that section of pipe should not be moved until USFWS has been consulted. If necessary, and under the direct supervision of the biologist, the pipe may be moved only once to remove it from the path of construction activity, until the fox has escaped (USFWS 2011).
- 4. All food-related trash items such as wrappers, cans, bottles, and food scraps should be disposed of in securely closed containers and removed at least once a week from a construction or project site (USFWS 2011).
- 5. No firearms shall be allowed on the project site (USFWS 2011).
- 6. To prevent harassment, mortality of kit foxes, or destruction of dens, no pets, such as dogs or cats, should be permitted on the project site (USFWS 2011).
- 7. Use of rodenticides and herbicides in project areas should be restricted. This is necessary to prevent primary or secondary poisoning of kit foxes and the depletion of prey populations on which they depend. All uses of such compounds should observe label and other restrictions mandated by the U.S. Environmental Protection Agency, California Department of Food and Agriculture, and other

state and federal legislation, as well as additional project-related restrictions deemed necessary by USFWS. If rodent control must be conducted, zinc phosphide should be used because of a proven lower risk to kit fox (USFWS 2011).

- 8. A representative shall be appointed by the project proponent who will be the contact source for any employee or contractor who might inadvertently kill or injure a kit fox, or who finds a dead, injured, or entrapped kit fox. The representative will be identified during the employee education program, and their name and telephone number shall be provided to USFWS (USFWS 2011).
- 9. An employee education program should be conducted for any project that has anticipated impacts to kit fox or other endangered species. The program should consist of a brief presentation by persons knowledgeable in kit fox biology and legislative protection to explain endangered species concerns to contractors, their employees, and military and/or agency personnel involved in the project. The program should include the following: a description of the San Joaquin kit fox and its habitat needs; a report of the occurrence of kit fox in the project area; an explanation of the status of the species and its protection under ESA; and a list of measures being taken to reduce impacts to the species during project construction and implementation. A fact sheet conveying this information should be prepared for distribution to the previously referenced people and anyone else who may enter the project site (USFWS 2011).
- 10. Upon completion of the project, all areas subject to temporary ground disturbances, including storage and staging areas, temporary roads, and pipeline corridors, should be re-contoured if necessary, and revegetated to promote restoration of the area to pre-project conditions. An area subject to "temporary" disturbance means any area that is disturbed during the project, but which after project completion will not be subject to further disturbance and will have the potential to be revegetated. Appropriate methods and plant species used to revegetate such areas should be determined on a site-specific basis in consultation with USFWS, CDFW, and revegetation experts (USFWS 2011).
- 11. In the case of trapped animals, escape ramps or structures should be installed immediately to allow the animal(s) to escape, or USFWS should be contacted for guidance (USFWS 2011).
- 12. Any contractor, employee, or military or agency personnel who are responsible for inadvertently killing or injuring a San Joaquin kit fox shall immediately report the incident to their representative. This representative shall contact the CDFW immediately in the case of a dead, injured, or entrapped kit fox. The CDFW contact for immediate assistance is State Dispatch at (916) 445-0045. They will contact the local warden or Mr. Paul Hoffman, the wildlife biologist, at (530) 934-9309. USFWS should be contacted at the numbers below (USFWS 2011).
- 13. The Sacramento Fish and Wildlife Office and CDFW shall be notified in writing within 3 working days of the accidental death or injury to a San Joaquin kit fox during project-related activities. Notification must include the date, time, and location of the incident or of the finding of a dead or injured animal, and any other pertinent information (USFWS 2011). The USFWS contact is the Chief of the Division of Endangered Species, at the addresses and telephone numbers below. The CDFW contact is Mr. Paul Hoffman at 1701 Nimbus Road, Suite A, Rancho Cordova, California 95670, (530) 934-9309.
- 14. New sightings of kit fox shall be reported to the California Natural Diversity Database. A copy of the reporting form and a topographic map clearly marked to show where the kit fox was observed should also be provided to USFWS at the address below (USFWS 2011). Any project-related information required by USFWS, or questions concerning the above conditions or their implementation, may be directed in writing to USFWS at: Endangered Species Division 2800 Cottage Way, Suite W2605 Sacramento, California 95825-1846 (916) 414-6620 or (916) 414-6600

***Additional Threshold Information:***

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## SPECIES ACCOUNT: *Vulpes vulpes necator* (Sierra Nevada red fox)

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### *Species Taxonomic and Listing Information*

**Commonly-used Acronym:** SNRF

**Listing Status:** Proposed Endangered

### **Physical Description**

The red fox is a relatively small canid with an elongated snout, large ears, slender legs and body, and a bushy tail with a white tip (Larivière and Pashitschniak-Arts 1996, p. 2; Aubry 1997, p. 55; Perrine 2005, p. 1; Perrine et al. 2010, p. 5). Red foxes typically have primarily red fur, but can also occur in a “cross phase” (primarily grayish-brown, with darker lines along the back and shoulders) or “black phase” (also called the silver phase; primarily black with occasional silver guard hairs) (Aubry 1997, p. 55; Perrine et al. 2010, p. 5). The Sierra Nevada red fox and two other montane subspecies (i.e., Cascades and Rocky Mountain red foxes) are characterized by specialized adaptations to cold areas (Sacks et al. 2010a, p. 1524). Such adaptations include a particularly thick and deep winter coat (Grinnell et al. 1937, p. 377) and small toe pads (4 millimeters (mm) (0.2 inches (in)) across or less) that are completely covered in winter by dense fur to facilitate movement over snow (Grinnell et al. 1937, pp. 378, 393; Sacks 2014a, p. 30). The Sierra Nevada red fox and other montane subspecies also tend to be smaller than other red foxes (Perrine et al. 2010, p. 5) (USFWS, 2015).

### **Taxonomy**

Sierra Nevada red fox is classified in the mammalian order Carnivora, family Canidae, and is one of 10, 11, or 13 subspecies of red fox recognized in North America by various sources (Hall 1981, p. 938; Larivière and Pashitschniak-Arts 1996, pp. 1–2; Aubry 1997, p. 55; Sacks et al. 2010a, pp. 1523, 1535; ITIS 2014, p. 1). The current known distribution of genetic variation across the range of the Sierra Nevada red fox places a disproportionate significance on both the Southern Cascades and Sierra Nevada segments for the maintenance of genetic diversity in the subspecies. In addition, different mtDNA haplotypes separate the Sierra Nevada red foxes that reside in the Southern Cascades from those that reside in the Sierra Nevada, indicating a lack of gene flow (USFWS, 2015). Based on genetic data, Sacks et al. (2010) expanded the scope of this subspecies. They indicated that the Sacramento Valley red fox population (named as a new subspecies *V. v. patwini*) is native to California and closely related to the Sierra Nevada red fox (*V. v. necator*) (NatureServe, 2015).

### **Historical Range**

In 1937, Grinnell et al. (1937, pp. 381–382) defined the range of the Sierra Nevada red fox in California as three separate areas: (1) The area of Mt. Shasta, primarily in the Cascades but extending slightly into the Trinity Mountains; (2) in the California Cascades around Lassen Peak; and (3) along the upper elevations of the Sierra Nevada Mountain Range from Tulare to Sierra Counties. A study by Sacks et al. (2010a, p. 1536) extended the historical range into the Cascade Mountains of Oregon to the Columbia River (USFWS, 2015).

### **Current Range**

The Sierra Nevada red fox’s range has been confirmed (via a combination of genetics and photographic evidence) to extend into the Oregon Cascades as far north as Mt. Hood,

significantly extending the subspecies' range beyond its historically known range in California. Sierra Nevada red fox are thus known from a total of seven sighting areas, located in the vicinity of (north to south) Mt. Hood, Mt. Washington, Dutchman Flat, Willamette Pass, and Crater Lake in Oregon; and Lassen and Sonora Pass in California (USFWS, 2015).

**Distinct Population Segments Defined**

Yes; Sierra Nevada

**Critical Habitat Designated**

No;

***Life History*****Feeding Narrative**

Adult: Sierra Nevada red fox appear to be opportunistic predators and foragers, with a diet primarily composed of small rodents, but also including deer carrion (*Odocoileus hemionus*) (particularly in winter and spring) and manzanita berries (*Arctostaphylos nevadensis*) (particularly in fall) (Perrine et al. 2010, pp. 24, 30, 32–33). Sierra Nevada red fox are most active at dusk and at night (Perrine 2005, p. 114), when many rodents are most active. High-elevation lagomorphs, such as snowshoe hare (*Lepus americanus*) and pika (*Ochotona princeps*), also are diet components of the subspecies, although they were not an important food source in the Lassen sighting area, possibly due to scarcity in the region (Perrine 2005, pp. 29–30) (USFWS, 2015).

**Reproduction Narrative**

Adult: Sierra Nevada red fox use natural openings in rock piles at the base of cliffs and slopes as denning sites (Grinnell et al. 1937, p. 394). They may also dig earthen dens similar to Cascade red foxes (although this has not been directly documented) (Aubry 1997, p. 58; Perrine 2005, p. 153). Sierra Nevada red fox litters are reported by Grinnell et al. (1937, p. 394) to average six pups with a range of three to nine; however, recent evidence suggests that litter sizes of two to three are more typical, and that reproductive output is generally low in montane foxes (Perrine 2005, pp. 152–153). The average lifespan, age-specific mortality rates, sex ratios, and demographic structure of Sierra Nevada red fox populations are not known; one study within a portion of the Lassen sighting area found that three Sierra Nevada red fox lived at least 5.5 years (CDFW 2015, p. 1), and another study within a portion of the Sonora Pass sighting area found the average annual adult survival rate to be 82 percent, which is relatively high for red foxes (Quinn and Sacks 2014, pp. 10, 14–15, 24) (USFWS, 2015). Mates usually in winter; 4-5 young are born in early spring, weaned in about 8 weeks (Biosystems Analysis 1989) (NatureServe, 2015).

**Geographic or Habitat Restraints or Barriers**

Adult: None known, typically occurs > 4,200 ft. elevation (USFWS, 2015)

**Environmental Specificity**

Adult: Moderate (inferred from USFWS, 2015)

**Habitat Narrative**

Adult: Sierra Nevada red fox use multiple habitat types in the alpine and subalpine zones (near and above treeline) (California Department of Fish and Game (CDFG) 1987, p. 3). In addition to meadows and rocky areas (U.S. Department of Agriculture (USDA) 2009, p. 506), Sierra Nevada red fox use high-elevation conifer habitat of various types (Perrine 2005, pp. 63–64). Nearest the treeline in the Lassen sighting area, where habitat use has been best documented, the subspecies frequents subalpine conifer habitat dominated by whitebark pine (*Pinus albicaulis*) and mountain hemlock (*Tsuga mertensiana*) (Perrine 2005, pp. 6, 63–64; California Department of Fish and Wildlife (CDFW) undated, p. 3; Verner and Purcell undated, p. 3). Winter sightings have occurred as low as 1,410 m (4,626 ft.) in the Lassen sighting area (Perrine 2005, pp. 2, 162), and 1,280 m (4,200 ft.) in Oregon (Aubry et al. 2015, p. 1). There are no clear barriers to dispersal (USFWS, 2015).

### ***Dispersal/Migration***

#### **Motility/Mobility**

Adult: High (USFWS, 2015)

#### **Migratory vs Non-migratory vs Seasonal Movements**

Adult: Seasonal movement (USFWS, 2015)

#### **Dispersal**

Adult: High (USFWS, 2015)

### **Dispersal/Migration Narrative**

Adult: Sierra Nevada red fox in Oregon, and at the Lassen sighting area in California, have also been found to descend during winter months into high-elevation conifer areas below the subalpine zone (Perrine 2005, pp. 63–64; Aubry et al. 2015, p. 1). While on these lower winter ranges, the subspecies has shown a preference for mature closed canopy conifer forests, despite the rarity of this forest structural category (less than 7 percent) in the area studied (Perrine 2005, pp. 67, 74, 90). Perrine (2005, pp. 2, 159) found within a portion of the Lassen sighting area that adult Sierra Nevada red fox established summer home ranges averaging 2,564 hectares (ha) (6,336 acres (ac)), with individual home ranges ranging from 262 ha (647 ac) to 6,981 ha (17,250 ac) (Perrine 2005, pp. 2, 159). Winter home ranges were larger, averaging 3,255 ha (8,042 ac) and ranging from 326 to 6,685 ha (806 to 16,519 ac) (Perrine 2005, p. 159). Quinn and Sacks (2014, pp. 2, 9, 11) found within a portion of the Sonora Pass sighting area that minimum home range estimates averaged 910 ha (2,249 ac), and were maintained both winter and summer (USFWS, 2015).

### ***Population Information and Trends***

#### **Population Trends:**

Oregon: unknown (USFWS, 2015); likely > 50% (NatureServe, 2015)

#### **Resiliency:**

Moderate (inferred from USFWS, 2015; see current range/distribution)

#### **Redundancy:**

Low (inferred from USFWS, 2015)

**Number of Populations:**

7 (USFWS, 2015; see current range/distribution)

**Population Size:**

Oregon: unknown; California: 71 adults (USFWS, 2015)

**Resistance to Disease:**

Low to salmon poisoning disease (SPD) and Elokomin fluke fever (EFF) (USFWS, 2015)

**Population Narrative:**

Information (both historical and current) is not available regarding the abundance or trends of Sierra Nevada red fox populations in Oregon. The estimated population size in the Lassen area is 42 adults, and 29 adults in the Sonora Pass area. Foxes are highly susceptible to SPD (USFWS, 2015). Over the long term, distribution and abundance have declined, but the precise degree of decline is uncertain - estimated > 50% (NatureServe, 2015).

***Threats and Stressors***

**Stressor:** Competition with coyotes (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Both coyotes and Sierra Nevada red foxes are opportunistic predators with considerable overlap in food consumed (Perrine 2005, pp. 36–37). Coyotes occur throughout the current range of the Sierra Nevada red fox, but typically at lower elevations during winter and early spring when snowpacks are high. If snowpacks are reduced in area due to climate change, coyotes would likely encroach into high-elevation areas during early spring when Sierra Nevada red fox are establishing territories and raising pups. Even in the absence of direct predation, the tendency of coyotes to chase off red foxes generally, and to compete with Sierra Nevada red fox for prey, may interfere with the ability of the subspecies to successfully raise offspring (Service 2015, pp. 48–51) (USFWS, 2015).

**Stressor:** Wildfire and fire suppression (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Wildfires may impact Sierra Nevada red fox by modifying suitable habitat that the subspecies relies on for multiple aspects of its life history (e.g., reducing denning habitat, reducing or eliminating habitat conditions that support an adequate prey base). In general, wildfires in western States, including California and Oregon, have been more frequent, larger, and more intense in the past 50 years, and particularly in the past 15 years (Independent Scientific Advisory Board (ISAB) 2007, pp. 22–23). Long-term habitat changes caused by wildfires acting in concert with increased temperatures and altered moisture regimes could possibly result in tree mortality or long-term removal of forested habitat that the subspecies relies on. Fire suppression can change suitable habitat conditions for the Sierra Nevada red fox to denser stands of trees with fewer open meadow or shrub areas, thereby potentially reducing the prey base for the subspecies. Fire suppression could also lead to direct effects on Sierra Nevada red

fox by allowing greater fuel buildup, thereby producing larger and hotter wildfires. However, there are no reports of direct mortality to red foxes from wildfires, and wildfires can improve habitat for red foxes by removing competing vegetation and encouraging production of grasses and shrubs favored by small mammals (Tesky 1995, p. 7), which the Sierra Nevada red fox depends upon as a prey base (USFWS, 2015).

**Stressor:** Climate change (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Over the past 50 years, warming temperatures have led to a greater proportion of precipitation falling as rain rather than snow, earlier snowmelt, and a decrease in snowpack throughout the western United States (Kapnick and Hall 2010, pp. 3446, 3448; Halofsky et al. 2011, p. 21). The consequent lengthening of summer drought and associated increases in mean annual temperature have, in recent decades, caused increased tree mortality rates in mature conifer forests in the range of the SNRF (van Mantgem et al. 2009, pp. 522–523). In addition to increased tree mortality, water deficit from climate change is also expected to decrease seedling establishment and tree growth in many currently forested areas, thereby altering tree species distributions (Littell et al. 2013, p. 112). Montane scrub communities, which require less water, may tend to increase, thereby decreasing and isolating areas of appropriate habitat for the subspecies. Thus, this type of vegetation change/shift could lead to greater competition and predation from coyotes (which are better adapted to drier and warmer conditions). Potential shifts in future vegetation type may lead to range shifts for the Sierra Nevada red fox in some localities, although information is not available to indicate precisely where nor how rapidly this may occur (USFWS, 2015).

**Stressor:** Trapping/hunting (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The Sierra Nevada red fox has historically been hunted and trapped for its thickly furred pelt, which was the most valuable of any terrestrial animal in California (Grinnell et al. 1937, pp. 396–397). Due to regulatory protections, hunting and trapping do not constitute a current or likely future stressor to Sierra Nevada populations in California or at the Crater Lake sighting area in Oregon, as there is no legal hunting or fur trapping for Sierra Nevada red fox in California or at Crater Lake National Park where the sightings in that area are known. In the counties where the other four Oregon sighting areas occur, low numbers of red foxes are harvested, some of which may be Sierra Nevada red fox. The best available data indicate that relatively few red fox (some of which may be Sierra Nevada red fox) are removed from an unknown number of populations as a result of fur trapping in Oregon. The potential impacts of live-trapping and handling for research purposes on Sierra Nevada red fox is considered discountable (USFWS, 2015).

**Stressor:** Disease (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Numerous pathogens are known to cause severe disease in canids. Those that have the highest potential to have population-level impacts on Sierra Nevada red fox are sarcoptic mange, canine distemper, and rabies (Perrine 2010, pp. 17, 28), as well as SPD and EFF. Future impacts of such diseases on any given population are difficult to predict, but the low population densities of the subspecies (Perrine et al. 2010, p. 9) should make transmission within a population or sighting area less likely except within family groups. SPD and EFF are known to occur within the subspecies' range and could potentially result in bacterial infections that are typically fatal to canids (USFWS, 2015).

**Stressor:** Predation (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Sierra Nevada red fox could be preyed on by domestic dogs at recreational areas (such as ski lodges or national parks) within their sighting areas, and in the course of being hunted with dogs, in any of the Oregon sighting areas other than at Crater Lake. To date, one documented case of Sierra Nevada red fox predation by a dog exists (i.e., a radio-collared female Sierra Nevada red fox was found dead in October 2002, as a result of a dog attack within 175 m (574 ft.) of a ski chalet in the Lassen sighting area (Perrine 2005, p. 141)). Although no direct documentation of coyote predation on Sierra Nevada red fox is available, coyotes will chase and occasionally kill other North American red fox subspecies, and are considered important competitors of red fox generally (Perrine 2005, pp. 36, 55; Perrine et al. 2010, p. 17). However, as climate change progresses, climatologists predict that snowpacks are expected to diminish in the future (Kapnick and Hall 2010, pp. 3446, 3448; Halofsky et al. 2011, p. 21). Thus, higher elevations with deep snowpack that currently deter coyotes may become more favorable to them, potentially increasing the likelihood of coyote predation in the future (USFWS, 2015).

**Stressor:** Hybridization (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Hybridization of Sierra Nevada red fox with other nonnative red fox could result in outbreeding depression or genetic swamping (Quinn and Sacks 2014, pp. 16–17). Researchers documented interbreeding between female Sierra Nevada red fox and two male nonnative red foxes, resulting in seven hybrid pups in 2013, and an additional four hybrid pups in 2014 (Sacks et al. 2015, p. 3). These hybrids were the only clear indication of successful reproduction in the study area between 2011 and 2014. In comparison, only eight full-blooded Sierra Nevada red fox were identified in the area during those years (Sacks et al. 2015, p. 3). Second, two Sierra Nevada red fox individuals at the Mt. Hood sighting area show evidence (via genetic testing of mtDNA) of past hybridization with nonnative red foxes, although the timing and extent of that hybridization remains unknown (Akins and Sacks 2015, p. 1) (USFWS, 2015).

**Stressor:** Vehicles (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Collision with vehicles is a known source of mortality for the Sierra Nevada red fox currently and is expected to continue into the future, given the presence of roads within the

range of the subspecies. Snowmobiles are another potential source for collisions and noise disturbance in all sighting areas with the exception potentially of the Lassen sighting area and a small area in the northwest portion of the Crater Lake sighting area, given the high level of recreational activity within or adjacent to those sighting areas (USFWS, 2015).

**Stressor:** Small isolated populations (USFWS, 2015)

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Small, isolated populations are more susceptible to impacts overall, and relatively more vulnerable to extinction due to genetic problems, demographic and environmental fluctuations, and natural catastrophes (Primack 1993, p. 255). Sierra Nevada red fox populations may be small or isolated from one another to the degree that such negative effects may be realized in the subspecies. The best available information suggests that Sierra Nevada red fox distribution within California (i.e., Lassen and Sonora Pass sighting areas) has contracted in the recent past. The primary risk of such impacts is in the future (within 50 years), although evidence of low reproductive success based on studies in portions of both populations suggest this could constitute a current impact of inbreeding depression, but to an unknown degree (USFWS, 2015).

### ***Recovery***

**Reclassification Criteria:**

Not available - this species does not have a recovery plan.

**Delisting Criteria:**

Not available - this species does not have a recovery plan.

**Recovery Actions:**

- Not available - this species does not have a recovery plan.

### ***Conservation Measures and Best Management Practices:***

- Because the Sierra Nevada red fox has only been documented to date to occur on Forest Service and NPS lands, primary conservation actions currently fall to those land management agencies, as well as the States. Various conservation and management efforts have been occurring since approximately 1974, including: (1) Significant subspecies-specific protections in California from hunting and trapping as a California-stated listed species in 1980; (2) minimized impacts from various stressors by the Forest Service as a result of its sensitive species designation in California (since 1998) and Oregon (since 2015); and (3) National Park Service protections at the Lassen and Crater Lake sighting areas associated with their requirement to “preserve fundamental physical and biological processes, as well as individual species, features, and plant and animal communities” (USDI NPS 2006, p. 36) (USFWS, 2015).
- Coordination with Federal and State partners in the future is anticipated by the Service if it is collectively determined that translocation of Sierra Nevada red fox individuals to different populations are prudent to aid in the conservation of the subspecies (USFWS, 2015).

### **References**

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Proposed Rule. 80 Federal Register 195. October 8, 2015. Pages 60989 - 61028

USFWS. 2015. Endangered and Threatened Wildlife and Plants

USFWS. 2015. 12-Month Finding on a Petition To List Sierra Nevada Red Fox as an Endangered or Threatened Species

## SPECIES ACCOUNT: *Zapus hudsonius luteus* (New Mexico meadow jumping mouse)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Endangered; July 10, 2014 (79 FR 33119).

### **Physical Description**

The New Mexico meadow jumping mouse is grayish-brown on the back, yellowish-brown on the sides, and white underneath. The species is approximately 18.7 to 25.5 centimeters (cm) (7.4 to 10 inches [in.]) in total length, with elongated feet (30.6 millimeters [1.2 in.]) and an extremely long, bicolored tail (13 cm [5.1 in.]) (USFWS 2015).

### **Taxonomy**

The New Mexico meadow jumping mouse was described in 1911 as *Zapus luteus*. In 1954, *Z. luteus* was synonymized with the western jumping mouse (*Z. princeps*) on the basis of skull and pelage (the hairy coat of a mammal) characteristics, and was renamed *Z. princeps luteus*. In 1981, Hafner et al. genetically analyzed southwestern *Zapus* and other representatives of the genus and reclassified the New Mexico meadow jumping mouse as the subspecies *Z. h. luteus*. Recent microsatellite and mitochondrial DNA genetic and morphological studies conclusively found that the New Mexico meadow jumping mouse is a distinct, well-diverged, monophyletic group (originating from a common ancestor) (USFWS 2013).

### **Historical Range**

The historical distribution likely included riparian wetlands along streams in the Sangre de Cristo and San Juan Mountains from southern Colorado to central New Mexico, including the Jemez and Sacramento Mountains and the Rio Grande Valley from Espanola to Bosque del Apache National Wildlife Refuge, and into parts of the White Mountains in eastern Arizona (79 FR 33119).

### **Current Range**

The current range includes portions of New Mexico, Arizona, and Colorado. The majority of local extirpations have occurred since the late 1980s to early 1990s; recent surveys have indicated that about 70 formerly occupied locations are now considered to be extirpated. Since 2005, there have been 29 documented remaining populations spread across eight conservation areas (two in Colorado, 15 in New Mexico, and one in Arizona (USFWS 2014). Recent surveys throughout New Mexico determined that populations persist at six locations in the Jemez Mountains, two locations in the Sangre de Cristo Mountains, and two isolated locations in the Sacramento Mountains. The species might be found in mountainous portions of Catron County in west-central New Mexico, given the relict populations in the White Mountains in nearby Arizona. The subspecies has also been reported from western Las Animas County, southern Colorado. The known elevational range in Arizona is 1,983 to 2,876 meters (6,500 to 9,430 feet) (NatureServe 2015).

### **Distinct Population Segments Defined**

No

**Critical Habitat Designated**

Yes; 3/16/2016.

**Legal Description**

On March 16, 2016, the U.S. Fish and Wildlife Service (Service) designated critical habitat for the New Mexico meadow jumping mouse (*Zapus hudsonius luteus*) under the Endangered Species Act of 1973, as amended (Act). In total, an area of approximately 5,657 hectares (13,973 acres) along 272.4 kilometers (169.3 miles) of flowing streams, ditches, and canals as critical habitat was designated in eight units within Colfax, Mora, Otero, Sandoval, and Socorro Counties in New Mexico; Las Animas, Archuleta, and La Plata Counties in Colorado; and Greenlee and Apache Counties in Arizona.

**Critical Habitat Designation**

The Service has designated approximately 5,657 hectares (13,973 acres) along 272.4 kilometers (169.3 miles) of flowing streams, ditches, and canals in eight units as critical habitat for the jumping mouse in the States of Colorado, New Mexico, and Arizona. Units 3, 4, and 5 have subunits, resulting in a total of 21 subunits and units designated. The critical habitat areas we describe below constitute our current best assessment of areas that meet the definition of critical habitat for the jumping mouse.

Unit 1—Sugarite Canyon Unit 1 consists of 344 ha (849 ac) along 13.0 km (8.1 mi) of streams on private lands and areas owned by the States of Colorado and New Mexico. The Colorado stream areas are found within Las Animas County, Colorado, and the New Mexico stream areas are found within Colfax County, New Mexico. The unit begins 0.6 km (0.4 mi) north of the headwaters of Lake Dorothey, Colorado, along the East Fork and 1.1 km (0.7 mi) north of the headwaters of Lake Dorothey along the West Fork of Schwacheim Creek and follows the drainage downstream, to include a 2.0-km (1.25-mi) segment of Chicorica Creek that is a tributary flowing into the headwaters of Lake Maloya and a 0.8-km (0.5-mi) segment of Segerstrom Creek, which is a tributary flowing into the western edge of Lake Maloya, New Mexico. The unit continues through Lake Maloya and includes about 1.8 km (1.1 mi) of the small western tributary Soda Pocket Creek, which flows into and includes lower Chicorica Creek below Lake Maloya Dam downstream to the terminus of the area at Lake Alice Dam within Sugarite Canyon State Park. Based upon captures of the jumping mouse since 2005 (Frey 2006d, pp. 19– 21, 67; Frey and Kopp 2013, entire; Colorado Parks and Wildlife 2013a, p. 1) approximately 2.8 ha (7 ac) within Unit 1 are considered occupied at the time of listing and contain suitable habitat. The occupied areas occur within Sugarite Canyon State Park in New Mexico along Sugarite Canyon at five locations: (1) Chicorica Creek 0.6 km (0.4 mi) below Lake Maloya Dam; (2) Segerstrom Creek just above the western confluence with Lake Maloya; (3) the headwaters of Lake Alice; and (4) Soda Pocket Creek and Campground along the two streams (2 separate locations) that cross the open meadow on Barlett Mesa near the campfire program area and behind campsite number 16 (Frey 2006d, pp. 19–21, 67; Frey and Kopp 2013, entire; Colorado Parks and Wildlife 2013a, p. 1). In 2011, the Track Fire burned nearly the entire watershed of Sugarite Canyon, significantly impacting the population at Sugarite Canyon State Park (Frey and Kopp 2013, entire; Service 2013c, entire). We consider this area within the geographical area occupied by the jumping mouse at the time of listing. The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, recreation, grazing, water use and management, floods, the reduction in the distribution and abundance of beaver ponds, and coalbed methane development. The occupied

areas are centered around the five capture locations plus an additional 0.8-km (0.5- mi) segment upstream and downstream of each of these areas where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Unit 1 are found both upstream and downstream of the occupied areas, and are considered essential to the conservation of the jumping mouse.

**Unit 2—Coyote Creek** Unit 2 consists of 239 ha (591 ac) along 11.8 km (7.4 mi) of Coyote Creek on private lands and an area owned by the State of New Mexico within Mora County. The unit begins at the confluence of Little Blue Creek and Coyote Creek and extends downstream to about the terminus just south of the Village of Guadalupita. Based upon captures of the jumping mouse since 2006 (Frey 2006d, pp. 24, 70; Frey 2012, p. 6), approximately 1.7 ha (4.3 ac) within Unit 2 are considered occupied at the time of listing and contain suitable habitat. The occupied areas occur within Coyote Creek State Park and several miles north of the park along Highway 434 in New Mexico at two locations along Coyote Creek including: (1) An area that contains extensive beaver ponds, dams, and canals and is located between the only vehicle bridge within the southwestern part of Coyote Creek State Park and the southern boundary of the park; and (2) within another area that contains extensive beaver activity about 1.9 km (1.2 mi) south of the confluence of Little Blue Creek and Coyote Creek. The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, recreation, grazing, water use and management, floods, the reduction in the distribution and abundance of beaver ponds, and development. The occupied areas are centered around the two capture locations plus an additional 0.8-km (0.5-mi) segment upstream and downstream of these areas where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Unit 2 are found both upstream and downstream of the occupied areas, and are considered essential to the conservation of the jumping mouse.

**Unit 3—Jemez Mountains** Unit 3 consists of 1,118 ha (2,761 ac) along 55.5 km (34.5 mi) of streams within three subunits on private lands and areas owned by the Forest Service and the State of New Mexico within Sandoval County, New Mexico. Areas designated as critical habitat for the jumping mouse in this unit incorporate the only habitat known to be occupied by the species since 2005 within the Jemez Mountains with the capability to support the breeding and reproduction of the species. **Subunit 3A—San Antonio:** Subunit 3A consists of 234 ha (579 ac) along 11.5 km (7.1 mi) of San Antonio Creek on private lands and areas owned by the Forest Service. This subunit begins along the northern part of San Antonio Creek where it exits the boundary of the Valles Caldera National Preserve and follows the creek through mostly Forest Service lands where it meets private land immediately downstream of the San Antonio Campground. Based upon the capture of one jumping mouse since 2005 (Frey 2005a, pp. 15, 24, 58), approximately 0.4 ha (1 ac) within Subunit 3A are considered occupied at the time of listing and contain suitable habitat. The occupied area is located along San Antonio Creek within a wet meadow near the southwestern part of San Antonio Campground (Frey 2005a, pp. 15, 24, 58). The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, recreation, grazing, floods, and the reduction in the distribution and abundance of beaver ponds. The occupied area is centered around the one capture location plus an additional 0.8-km (0.5-mi) segment upstream and downstream of this area where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Subunit 3A are found both upstream and downstream of the occupied area, and are considered essential to the

conservation of the jumping mouse (as described under the heading Unit Descriptions, above).

**Subunit 3B—Rio Cebolla:** Subunit 3B consists of 429 ha (1,060 ac) along 20.7 km (12.9 mi) of the Rio Cebolla on private lands and areas owned by the Forest Service and the State of New Mexico. This subunit extends from an old beaver dam about 0.6 km (0.4 mi) north of Hay Canyon downstream about where it meets the Rio de las Vacas. Based upon captures of the jumping mouse since 2005 (Frey 2005a, pp. 23– 28, 37–38; Frey 2007b, p. 11), approximately 10.7 ha (26.4 ac) within Subunit 3B are considered occupied at the time of listing and contain suitable habitat. The occupied areas occurs on State of New Mexico and Forest Service lands in New Mexico at six locations along the Rio Cebolla: (1) Near the western edge of the northwestern pond along the access road within the New Mexico Department of Game and Fish’s Seven Springs Hatchery; (2) within Fenton Lake State Park at the upper end of Fenton Lake Marsh above Highway 126 and the New Mexico Highway 126 bridge; (3) within Fenton Lake State Park Day Use Area at the mouth of a small tributary that enters the southwest side of Fenton Lake; (4) within Lake Fork Canyon inside a livestock enclosure above the bridge on Forest Road 376; (5) within a network of channels, beaver ponds, and wet meadows about 0.9 km (0.6 mi) southwest of Forest Road 376 bridge; and (6) about 2.7 km (1.7 mi) north of the confluence of the Rio Cebolla and the Rio de las Vacas (Frey 2005a, pp. 23–28, 37–38; Frey 2007b, p. 11). The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, recreation, grazing, floods, the reduction in the distribution and abundance of beaver ponds, development, and highway reconstruction. The occupied areas are centered around the six capture locations plus an additional 0.8- km (0.5-mi) segment upstream and downstream of these areas where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Subunit 3B are found both upstream and downstream of the occupied areas, and are considered essential to the conservation of the jumping mouse (as described under the heading Unit Descriptions, above).

**Subunit 3C—Rio de las Vacas:** Subunit 3C consists of 454 ha (1,122 ac) along 23.3 km (14.5 mi) of the Rio de las Vacas on private lands and areas owned by the Forest Service. This subunit starts about 0.8 km (0.5 mi) north of Forest Road 94 adjacent to Burned Canyon and extends downstream to the confluence with Subunit 3B. Although much of the habitat was historically occupied with individuals detected as recently as 1989 (Morrison 1985; 1992, p. 311; Frey 2005a, p. 7), no New Mexico meadow jumping mice were captured during surveys in 2005 (Frey 2005a, p. 18). The entire subunit is considered unoccupied at the time of listing. This subunit has perennial flowing water with saturated soils and a high potential of being restored to suitable habitat. It has the potential for natural recolonization of jumping mice populations through individuals that naturally disperse. This subunit would provide connectivity to Subunit 3B and allow for possible expansion of jumping mice from that currently occupied subunit, which is contiguous with Subunit 3C, into historically occupied habitat along the Rio de las Vacas drainage. We found this entire stream section would provide further connectivity to the adjacently occupied habitat within Subunit 3B and increase the length and size of the suitable habitat. All of the areas within Subunit 3C are considered essential to the conservation of the jumping mouse.

**Unit 4—Sacramento Mountains** Unit 4 consists of 777 ha (1,920 ac) along 36.2 km (22.5 mi) of streams within five subunits on private lands and areas owned by the Forest Service within Otero County, New Mexico. Areas designated as critical habitat for the jumping mouse in this unit incorporate the only habitat known to be occupied by the species since 2005 within the Sacramento Mountains with the capability to support the breeding and reproduction of the species.

**Subunit 4A—Silver Springs:** Subunit 4A consists of 105 ha (260 ac) along 5.2 km (3.2 mi) of Silver Springs Creek on private lands and areas owned by the Forest Service. This subunit

begins about 0.3 km (0.2 mi) north of the intersection of Forest Road 162 and New Mexico Highway 244 and follows Silver Springs Creek downstream to the boundary of Forest Service and Mescalero Apache lands. Based upon the capture of one jumping mouse since 2005 (Frey 2005a, p. 31), approximately 5.4 ha (13.3 ac) within Subunit 4A are considered occupied at the time of listing. The occupied area is located on Forest Service lands in New Mexico within a grazing exclosure containing welldeveloped riparian habitat about 7.4 km (4.6 mi) north of Cloudcroft along middle Silver Springs Creek, at Junction of Turkey Pen Canyon and Forest Road 405 (Frey 2005a, pp. 31, 38). The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, grazing, floods, and the reduction in the distribution and abundance of beaver ponds. The occupied area is centered around one capture location plus an additional 0.8-km (0.5-mi) segment upstream and downstream of this area where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Subunit 4A are found both upstream and downstream of the occupied area, and are considered essential to the conservation of the jumping mouse.

**Subunit 4B—Upper Penasco:** Subunit 4B consists of 136 ha (335 ac) along 6.4 km (4.0 mi) of the Rio Penasco on private lands and areas owned by the Forest Service. This subunit begins at the junction of Forest Service Road 164 and New Mexico Highway 6563 and follows the Rio Penasco drainage downstream to about 2.4 km (1.5 mi) below Bluff Spring at the boundary of private and Forest Service lands. Although much of the habitat was historically occupied with individuals detected as recently as 1988 (Morrison 1989, pp. 7–10, Frey 2005a, pp. 30–31), no New Mexico meadow jumping mice were captured during surveys in 2005 (Frey 2005a, pp. 19–20, 32–34). The entire subunit is considered unoccupied at the time of listing. This subunit contains perennial flowing water with saturated soils and has a high potential of being restored to suitable habitat. It would augment the current size and connectivity of suitable habitat to increase the distribution of the jumping mouse in the Sacramento Mountains and provide population redundancy and resiliency. All of the areas within Subunit 4B are considered essential to the conservation of the jumping mouse (as described under the heading Unit Descriptions, above).

**Subunit 4C—Middle Penasco:** Subunit 4C consists of 264 ha (652 ac) along 11.4 km (7.1 mi) of the Rio Penasco on private lands and areas owned by the Forest Service. This subunit begins at the junction of Wills Canyon and Forest Service Road 169 and follows the Rio Penasco drainage downstream to the junction of Forest Road 212. Based upon the capture of two jumping mice in 2012, following the cessation of grazing for 2 years (Forest Service 2012a, entire; 2012c, entire; Forest Service 2012h, pp. 2–4; Service 2012d; U.S. Army Corps of Engineers 2012, entire; 2012a, entire), approximately 0.3 ha (0.75 ac) within Subunit 4C are considered occupied at the time of listing. The occupied area is located on Forest Service lands in New Mexico within a wetland at the junction of Cox Canyon and the Rio Penasco (Forest Service 2012h, pp. 2–4). The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, recreation, grazing, floods, and the reduction in the distribution and abundance of beaver ponds. The occupied area is centered around one capture location plus an additional 0.8-km (0.5-mi) segment upstream and downstream of this area where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Subunit 4C are found both upstream and downstream of the occupied area, and are considered essential to the conservation of the jumping mouse (as described under the heading Unit Descriptions, above).

**Subunit 4D—Wills Canyon:** Subunit 4D consists of 111 ha (275 ac) along 5.5 km (3.4 mi) of streams on private lands and areas owned by the Forest Service. This subunit begins at upper Mauldin Spring, the head of the Wills Canyon, and follows the drainage downstream along Forest Service Road 169 to the boundary of Forest Service and private lands in the vicinity of Bear

Spring. Based upon the capture of jumping mice in 2012 and 2013 (Forest Service 2012a, entire; 2012h, pp. 2–5; 2013a, entire; Service 2012d, pp. 2, 8), approximately 0.8 ha (1.9 ac) within Subunit 4D are considered occupied at the time of listing. The occupied area is located on Forest Service lands in New Mexico within the grazing exclosures at Mauldin Spring in Wills Canyon (Forest Service 2012a, entire; 2012h, pp. 2–5; 2013a, entire; Service 2012d, pp. 2, 8). The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: severe wildland fires, grazing, floods, and the reduction in the distribution and abundance of beaver ponds. The occupied area is centered around the capture locations plus an additional 0.8-km (0.5-mi) segment upstream and downstream of this area where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Subunit 4D are found both upstream and downstream of the occupied area, and are considered essential to the conservation of the jumping mouse (as described under the heading Unit Descriptions, above). Subunit 4E—Agua Chiquita Canyon: Subunit 4E consists of 161 ha (398 ac) along 7.7 km (4.8 mi) of Agua Chiquita Creek on areas owned by the Forest Service. This subunit begins about 0.8 km (0.5 mi) upstream of the livestock exclosure around Barrel and Sand Springs along Agua Chiquita Creek and follows the canyon downstream along Forest Service Road 64 to Crisp, a Forest Service riparian pasture. Based upon multiple captures of jumping mice since 2005 (Frey 2005a, p. 34; Forest Service 2010, entire; Service 2012d, pp. 1–2), approximately 4.9 ha (12.0 ac) within Subunit 4E are considered occupied at the time of listing. The occupied areas are located on Forest Service lands in New Mexico within two of four fenced livestock exclosures, which includes the exclosure surrounding Sand and Barrel Springs and the most downstream section of the second in the series of four exclosures (Frey 2005a, p. 34; Forest Service 2010, entire; Service 2012d, pp. 1–2). The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, recreation, grazing, floods, and the reduction in the distribution and abundance of beaver ponds. The occupied areas are centered around the two capture locations plus an additional 0.8-km (0.5-mi) segment upstream and downstream of these areas where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Subunit 4E are found both upstream and downstream of the occupied areas, and are considered essential to the conservation of the jumping mouse.

Unit 5—White Mountains Unit 5 consists of 2,448 ha (6,046 ac) along 116.6 km (72.4 mi) of streams within eight subunits on private lands and areas owned by the Forest Service and the State of Arizona within Greenlee and Apache Counties, Arizona. Areas designated as critical habitat for the jumping mouse in this unit incorporate the only habitat known to be occupied by the species since 2005 within the White Mountains with the capability to support the breeding and reproduction of the species. Subunit 5A—Little Colorado: Subunit 5A consists of 478 ha (1,181 ac) along 22.6 km (14.0 mi) of the Little Colorado River on private lands and areas owned by the Forest Service. This subunit encompasses the East and West Forks of the Little Colorado River. The East Fork Segment begins 0.8 km (0.5 mi) upstream of the Phelps Research Natural Area and follows the drainage downstream about 3.2 km (2.0 mi) to the confluence of Lee Valley Creek and then runs upstream about 1.6 km (1.0 mi) to the dam of Lee Valley Reservoir. The subunit continues from the confluence of Lee Valley Creek and the East Fork, downstream to the confluence of the West Fork of the Little Colorado River, continuing to about 8.9 km (5.5 mi) upstream along the drainage to about 0.8 km (0.5 mi) past Sheep's Crossing. Based upon multiple captures of jumping mice since 2008 (Frey 2011, pp. 29, 87; AGFD 2012a, p. 3), approximately 0.6 ha (1.5 ac) within Subunit 5A are considered occupied at the time of listing. The occupied area is

located on Forest Service lands in Arizona within a livestock enclosure along a short 0.4-km (0.25-mi) stream reach that is 1.8 km (1.1 mi) south of Greer, below Montlure Camp (Frey 2011, pp. 29, 87; AGFD 2012a, p. 3). In 2011, the Wallow Fire burned much of this area, and surveys during 2012 continued to detect New Mexico meadow jumping mice (AGFD 2012a, p. 3). The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, recreation, grazing, floods, the reduction in the distribution and abundance of beaver ponds, and development. The occupied areas are centered around the capture locations plus an additional 0.8-km (0.5-mi) segment upstream and downstream of this area where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Subunit 5A are found both upstream and downstream of the occupied area, and are considered essential to the conservation of the jumping mouse (as described under the heading Unit Descriptions, above).

Subunit 5B—Nutrioso: Subunit 5B consists of 413 ha (1,021 ac) along 20.4 km (12.7 mi) of Nutrioso Creek on private lands and areas owned by the Forest Service. This subunit begins at the confluence of Paddy Creek about 4.8 km (3 mi) south of the town of Nutrioso and follows the drainage downstream about 16 km (10 mi) to Nelson Reservoir. Based upon multiple captures of jumping mice since 2008 (Frey 2011, pp. 29, 35, 89, 95; AGFD 2012a, p. 3), approximately 1.9 ha (4.9 ac) within Subunit 5B are considered occupied at the time of listing. The occupied area is located on Forest Service lands in Arizona along a short 1.3-km (0.8-mi) stream reach 3.9 km (2.4 mi) south of the town of Nutrioso. In 2011, the Wallow Fire burned much of this area, and surveys during 2012 continued to detect New Mexico meadow jumping mice (AGFD 2012a, p. 3). The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, grazing, floods, the reduction in the distribution and abundance of beaver ponds, highway reconstruction, and development. The occupied area is centered around the capture locations plus an additional 0.8-km (0.5-mi) segment upstream and downstream of this area where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Subunit 5B are found both upstream and downstream of the occupied area, and are considered essential to the conservation of the jumping mouse (as described under the heading Unit Descriptions, above).

Subunit 5C—San Francisco: Subunit 5C consists of 252 ha (622 ac) along 11.8 km (7.3 mi) of the San Francisco River and its tributary Turkey (=Talwiwi) Creek on private lands and areas owned by the Forest Service. This subunit begins about 0.6 km (0.4 mi) west of Forest Road 8854 along the San Francisco River and follows the drainage downstream about 10.5 km (6.5 mi), including a 1.3-km (0.8-mi) segment of Turkey (=Talwiwi) Creek that is south of Arizona Highway 180, then continues downstream to the headwaters of Luna Lake. Based upon multiple captures of jumping mice since 2008 (Frey 2011, pp. 29, 97, 100), approximately 0.9 ha (2.3 ac) within Subunit 5C are considered occupied at the time of listing. There are two occupied areas within this unit located on Forest Service lands in Arizona including: (1) A small livestock enclosure along a 0.2-km (0.1-mi) stream reach of upper Turkey Creek at the junction of Highway 80 and Forest Road 289; and (2) two fenced livestock enclosures along a 0.4-km (0.2-mi) stream reach at the junction of the San Francisco River and Forest Road 8854 (Frey 2011, p. 97). In 2011, the Wallow Fire burned much of this area, and surveys during 2012 did not detect New Mexico meadow jumping mice (AGFD 2012, entire, 2012a, p. 2). However, until multiple years of surveys determine that the population has been extirpated, we consider this area within the geographical area occupied by the jumping mouse at the time of listing. The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, grazing, floods, the reduction in the distribution and abundance of beaver ponds, highway reconstruction, and development. The

occupied areas are centered around the two capture locations plus an additional 0.8-km (0.5- mi) segment upstream and downstream of these areas where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Subunit 5C are found both upstream and downstream of the occupied areas, and are considered essential to the conservation of the jumping mouse (as described under the heading Unit Descriptions, above).

**Subunit 5D—East Fork Black:** Subunit 5D consists of 421 ha (1,040 ac) along 20.3 km (12.6 mi) of the East Fork of the Black River areas owned by the Forest Service. This subunit begins 0.8 km (0.5 mi) north of the intersection of Three Forks Road and Route 285 and follows the drainage downstream about 20.3 km (12.6 mi), where it abuts Subunit 5E. Based upon multiple captures of jumping mice since 2008 (Frey 2011, p. 97; AGFD 2012, entire, 2012a, p. 2), approximately 6.9 ha (16.9 ac) within Subunit 5D are considered occupied at the time of listing. The occupied area is located on Forest Service lands in Arizona along the headwaters of the East Fork Black River near the intersection of Three Forks Road and Route 285 (Frey 2011, p. 29, 35, 40, 104; AGFD 2012, entire, 2012a, p. 2). In 2011, the Wallow Fire burned much of this area, and surveys during 2012 continued to detect New Mexico meadow jumping mice (AGFD 2012a, p. 2). The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, grazing, floods, the reduction in the distribution and abundance of beaver ponds, and highway reconstruction. The occupied area is centered around the capture location plus an additional 0.8-km (0.5-mi) segment upstream and downstream of this area where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Subunit 5D are found both upstream and downstream of the occupied area, and are considered essential to the conservation of the jumping mouse (as described under the heading Unit Descriptions, above).

**Subunit 5E—West Fork Black:** Subunit 5E consists of 481 ha (1,188 ac) along 23.0 km (14.3 mi) of the West Fork of the Black River on private lands and areas owned by the Forest Service and the State of Arizona. The subunit begins at the confluence of the West Fork of the Black River and Burro Creek and follows the drainage downstream where it abuts Subunit 5D. Based upon multiple captures of jumping mice since 2007 (Underwood, 2007, entire; Frey 2011, pp. 29, 40, 104; AGFD 2012, p. 2), approximately 13.7 ha (33.9 ac) within Subunit 5E are considered occupied at the time of listing. The occupied areas occur on Forest Service lands in Arizona at four locations: (1) Along the upper West Fork Black River just north of Forest Road 116; (2) immediately adjacent to the campground along the middle Fork of the Black River; (3) at the junction of Forest Road 68 and the middle Fork of the Black River; and (4) near the junction of the lower Fork of the Black River and Home Creek (Underwood 2007, entire; Frey 2011, pp. 29, 40, 104; AGFD 2012, p. 2012a, pp. 2–3). In 2011, the Wallow Fire burned much of this area, and surveys during 2012 continued to detect New Mexico meadow jumping mice at the lower and middle sections of the West Fork Black River (AGFD 2012a, pp. 2–3). Although New Mexico meadow jumping mice were not detected at the upper West Fork Black River location, until multiple years of surveys determine that the population has been extirpated, we consider this area within the geographical area occupied by the jumping mouse at the time of listing. The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, grazing, floods, the reduction in the distribution and abundance of beaver ponds, and highway reconstruction. The occupied areas are centered around the four capture locations plus an additional 0.8-km (0.5- mi) segment upstream and downstream of these areas where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Subunit 5E are found both upstream and downstream of the occupied areas, and are considered essential to the conservation of the jumping mouse (as described under the heading Unit Descriptions, above).

**Subunit 5F—Boggy**

and Centerfire: Subunit 5F consists of 197 ha (485 ac) along 8.9 km (5.5 mi) of Boggy Creek and Centerfire Creek on areas owned by the Forest Service. The east segment of the subunit begins 0.8 km (0.5 mi) north of the intersection of Route 25 and Boggy Creek and follows the drainage downstream to the confluence with Centerfire Creek. The west segment begins 0.8 km (0.5 mi) north of the intersection of Route 25 and Centerfire Creek, and follows the drainage downstream to the confluence with Boggy Creek, then continues downstream to the confluence with the Black River. Based upon multiple captures of jumping mice since 2008 (Frey 2011, pp. 29, 104–105; AGFD 2012, pp. 3–4; 2012a, p. 3), approximately 3.0 ha (7.5 ac) within Subunit 5F are considered occupied at the time of listing. The occupied areas are located on Forest Service lands in Arizona within fenced livestock exclosures at the junction of Forest Road 25 and Boggy Creek; and within a fenced livestock exclosure at the junction of Forest Road 25 and Centerfire Creek (Frey 2011, pp. 29, 104–105; AGFD 2012, pp. 3–4; 2012a, p. 3). In 2011, the Wallow Fire burned much of this area, and surveys during 2012 continued to detect New Mexico meadow jumping mice (AGFD 2012a, p. 3). The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, grazing, floods, and the reduction in the distribution and abundance of beaver ponds. The occupied areas are centered around the capture locations plus an additional 0.8-km (0.5-mi) segment upstream and downstream of these areas where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Subunit 5F are found both upstream and downstream of the occupied areas, and are considered essential to the conservation of the jumping mouse (as described under the heading Unit Descriptions, above).

Subunit 5G—Corduoy: Subunit 5G consists of 104 ha (256 ac) along 4.8 km (3.0 mi) of Corduroy Creek on lands owned by the Forest Service. The subunit begins at the headwaters about 0.8 km (0.5 mi) south of the intersection of County Road 24 and County Road 8184A and follows the drainage downstream to the confluence with Fish Creek. Based upon multiple captures of jumping mice since 2009 (Frey 2011, pp. 104–105; AGFD 2012, entire, 2012a, p. 4), approximately 0.4 ha (1.1 ac) within Subunit 5G are considered occupied at the time of listing. The occupied area is located on Forest Service lands in Arizona within fenced livestock exclosures at the junction of Forest Road 8184A and Corduroy Creek (Frey 2011, pp. 104–105; AGFD 2012, entire, 2012a, p. 4). In 2011, the Wallow Fire burned much of this area, and surveys during 2012 continued to detect New Mexico meadow jumping mice (AGFD 2012a, p. 4). The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, grazing, floods, and the reduction in the distribution and abundance of beaver ponds. The occupied area is centered around the capture location plus an additional 0.8-km (0.5-mi) segment upstream and downstream of this area where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Subunit 5G are found both upstream and downstream of the occupied area, and are considered essential to the conservation of the jumping mouse (as described under the heading Unit Descriptions, above).

Subunit 5H—Campbell Blue: Subunit 5H consists of 102 ha (253 ac) along 4.8 km (3.0 mi) of Campbell Blue Creek on private lands and areas owned by the Forest Service. The subunit begins at the confluence with Cat Creek along Forest Road 281 and extends downstream to the confluence with Turkey Creek. Based upon multiple captures of jumping mice since 2008 (Frey 2011, pp. 29, 101), approximately 0.008 ha (0.02 ac) within Subunit 5H are considered occupied at the time of listing. The occupied area is located on Forest Service lands in Arizona within a livestock exclosure 13 km (8 mi) north of the community of Blue (Frey 2011, pp. 29, 101). In 2011, the Wallow Fire burned much of this area, and surveys during 2012 did not detect New Mexico meadow jumping mice (AGFD 2012, entire, 2012a, p. 2). However, until multiple years of surveys determine that the population has been

extirpated, we consider this area within the geographical area occupied by the jumping mouse at the time of listing. The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Severe wildland fires, grazing, floods, and the reduction in the distribution and abundance of beaver ponds. The occupied area is centered around the capture location plus an additional 0.8-km (0.5-mi) segment upstream and downstream of this area where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Subunit 5H are found both upstream and downstream of the occupied area, and are considered essential to the conservation of the jumping mouse.

**Unit 6—Bosque del Apache National Wildlife Refuge (NWR)** Unit 6 consists of 403 ha (995 ac) along 21.1 km (13.1 mi) of ditches and canals on the Service's Bosque del Apache NWR, Socorro County, New Mexico. This unit includes parts of a complex ditch system with associated irrigation of NWR management units, making habitat within this area unique. This unit begins in the northern part of the NWR and generally follows the Riverside Canal to the southern end. The NWR is the only locality within the middle Rio Grande considered still in existence (Frey and Wright 2012; Service 2014a, entire). Based upon multiple captures of the jumping mouse since 2009 (Frey and Wright 2012, entire; Service 2014a, entire), approximately 4.1 ha (10.1 ac) within Unit 6 are considered occupied at the time of listing. The occupied area is located on NWR lands in New Mexico along a 2.7-km (1.7-mi) segment of the Riverside Canal (Frey and Wright 2012, entire; Service 2014a, entire). The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Water use and management; severe wildland fires; and thinning, mowing, or removing tamarisk (also known as saltcedar, *Tamarix ramosissima*), decadent stands of willow that are greater than 3 years old or 1.5 m (4.9 ft) tall. The occupied area is centered around the capture locations plus an additional 0.8-km (0.5-mi) segment upstream and downstream of this area where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Unit 6 are found both upstream and downstream of the occupied area, and are considered essential to the conservation of the jumping mouse.

**Unit 7—Florida** Unit 7 consists of 253 ha (626 ac) along 13.6 km (8.4 mi) of the Florida River on private lands and an area owned by the Bureau of Land Management, La Plata County, Colorado. The unit begins at the irrigation diversion structure (Florida Ditch main headgate) of the Florida Water Conservancy District about 0.8 km (0.5 mi) northeast of the intersection of La Plata County Road 234 and 237 and follows the drainage downstream to about 0.16 km (0.1 mi) north of Ranchos Florida Road. Based upon the capture of two jumping mice since 2007 (Museum of Southwestern Biology 2007; 2007a; Frey 2008c, pp. 42–45, 56; 2011a, pp. 19, 33), approximately 0.15 ha (0.37 ac) within Unit 7 are considered occupied at the time of listing. The occupied area is located on private lands in Colorado 0.9 km (0.6 mi) north of Highway 160 along the Florida River (Museum of Southwestern Biology 2007; 2007a; Frey 2008c, pp. 42–45, 56; 2011a, pp. 19, 33). The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Floods, water use and management, development, and coalbed methane. The occupied area is centered around the capture location plus an additional 0.8-km (0.5-mi) segment upstream and downstream of this area where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Unit 7 are found both upstream and downstream of the occupied area, and are considered essential to the conservation of the jumping mouse.

Unit 8—Sambrito Creek Unit 8 consists of 75 ha (185 ac) along 4.6 km (2.9 mi) of Sambrito Creek on private lands and areas owned by the State of Colorado within Navajo State Park, near Arboles, Archuleta County, Colorado. There are two segments within this unit. One segment begins at Archuleta County Road 977, following Sambrito Creek downstream to the headwaters of Navajo Reservoir. The second segment starts about 0.3 km (0.2 mi) west of the intersection of Colorado Road 977 and 988 and follows the drainage about 3.9 km (2.1 mi) through the Sambrito Wetlands Area downstream about to the headwaters of Navajo Reservoir. Based upon multiple captures of jumping mice since 2012 (Colorado Parks and Wildlife 2012, entire, 2013, entire; Ecosphere 2014, entire), approximately 0.9 ha (2.3 ac) within Unit 8 are considered occupied at the time of listing. The occupied area is located on State of Colorado lands immediately south of Archuleta County Road 977 along the unnamed drainage through the Sambrito Wetlands Areas about 1.8 km (1.1 mi) due west of Sambrito Creek (Colorado Parks and Wildlife 2012, entire). The features essential to the conservation of this subspecies may require special management considerations or protection to reduce the following threats: Floods, grazing, water use and management, the reduction in the distribution and abundance of beaver ponds, development, recreation, and coalbed methane. The occupied area is centered around the capture location that is about 0.5 km (0.3 mi) south of Archuleta County Road 977 plus an additional 0.8-km (0.5-mi) segment upstream and downstream of this area where the physical and biological features of critical habitat are found. The remaining unoccupied areas within Unit 8 are found both upstream and downstream of the occupied area, and are considered essential to the conservation of the jumping mouse.

#### **Primary Constituent Elements/Physical or Biological Features**

Critical habitat units are designated for Colfax, Mora, Otero, Sandoval, and Socorro Counties in New Mexico; Las Animas, Archuleta, and La Plata Counties in Colorado; and Greenlee and Apache Counties in Arizona. Within these areas, the primary constituent elements of the physical or biological features essential to the conservation of the New Mexico meadow jumping mouse consist of the following:

- (i) Riparian communities along rivers and streams, springs and wetlands, or canals and ditches that contain: (A) Persistent emergent herbaceous wetlands especially characterized by presence of primarily forbs and sedges (*Carex* spp. or *Schoenoplectus pungens*); or (B) Scrub-shrub riparian areas that are dominated by willows (*Salix* spp.) or alders (*Alnus* spp.) with an understory of primarily forbs and sedges; and
- (ii) Flowing water that provides saturated soils throughout the New Mexico meadow jumping mouse's active season that supports tall (average stubble height of herbaceous vegetation of at least 61 centimeters (24 inches)) and dense herbaceous riparian vegetation composed primarily of sedges (*Carex* spp. or *Schoenoplectus pungens*) and forbs, including, but not limited to, one or more of the following associated species: Spikerush (*Eleocharis macrostachya*), beaked sedge (*Carex rostrata*), rushes (*Juncus* spp. and *Scirpus* spp.), and numerous species of grasses such as bluegrass (*Poa* spp.), slender wheatgrass (*Elymus trachycaulus*), brome (*Bromus* spp.), foxtail barley (*Hordeum jubatum*), or Japanese brome (*Bromus japonicas*), and forbs such as water hemlock (*Circuta douglasii*), field mint (*Mentha arvensis*), asters (*Aster* spp.), or cutleaf coneflower (*Rudbeckia laciniata*); and

(iii) Sufficient areas of 9 to 24 kilometers (5.6 to 15 miles) along a stream, ditch, or canal that contain suitable or restorable habitat to support movements of individual New Mexico meadow jumping mice; and

(iv) Adjacent floodplain and upland areas extending approximately 100 meters (330 feet) outward from the boundary between the active water channel and the floodplain (as defined by the bankfull stage of streams) or from the top edge of the ditch or canal.

### **Special Management Considerations or Protections**

Critical habitat does not include manmade structures (such as buildings, fire lookout stations, runways, roads, and other paved areas) and the land on which they are located existing within the legal boundaries on April 15, 2016.

The features essential to the conservation of this species may require special management considerations or protection to reduce the following threats: Excessive grazing pressure, water use and management, highway reconstruction, commercial and residential development, severe wildland fires, unregulated recreation, and the reduction in the distribution and abundance of beaver ponds. These activities have the potential to affect the PCEs if they are conducted within or adjacent to units designated as critical habitat.

Management activities that could ameliorate these threats include, but are not limited to: (1) Maintaining occupied jumping mouse sites with active management to continue the protection of these areas from livestock grazing; (2) restoring, enhancing, and managing additional habitat through fencing of riparian areas, especially the Santa Fe, Lincoln, and Apache-Sitgreaves National Forests (this will facilitate restoration of the required vegetative components and support the expansion of populations of the jumping mouse into areas that were historically occupied by the species, but where natural expansion is currently unlikely because no suitable habitat remains); (3) restoring habitat on Bosque del Apache NWR or other areas by carefully managing mowing (e.g., not mowing during the active season) and removing willows older than 5 years to maintain early seral habitat conditions along irrigation canals and ditches; and (4) developing and implementing a beaver management or restoration plan for occupied and historic jumping mouse localities where appropriate.

### ***Life History***

#### **Feeding Narrative**

Adult: The New Mexico meadow jumping mouse is an omnivore, with a diet that consists primarily of insects, and seeds of grasses and forbs; with seeds of sedges, bulrush, and cattail infrequently eaten. These food resources are of limited distribution, because they occur in dense herbaceous vegetation of sedges and forbs along flowing streams. The New Mexico meadow jumping mouse may compete with meadow voles (*Microtus pennsylvanicus*) for available food resources (USFWS 2013). The bioenergetics requirements of this subspecies are low; they are nocturnal and hibernate for 9 months of the year (USFWS 2015). Prior to hibernation, jumping mice must accumulate sufficient fat reserves to sustain them through hibernation. The last several weeks prior to hibernation are spent rapidly building up fat reserves to survive, because jumping mice do not appear to cache food for the winter and as many as 67 percent may perish during hibernation (USFWS 2013).

**Reproduction Narrative**

Adult: The breeding season for the New Mexico jumping mouse begins in July or August and is followed by an approximately 17- to 21-day gestation period (USFWS 2013; USFWS 2015). Females give birth to one litter annually, with a clutch size between two and seven young. New Mexico meadow jumping mice must have rich, abundant food sources during the summer so they can accumulate sufficient fat reserves to survive their long hibernation period. In addition, they require intact upland areas adjacent to riparian wetland areas to build nests in dry soils, or use burrows to give birth to young in the summer and hibernate over the winter (79 FR 33119). Young are fully developed and weaned at 4 weeks after birth (USFWS 2015).

**Spatial Arrangements of the Population**

Adult: Clumped within a narrow habitat range. Individuals are solitary (USFWS 2013).

**Environmental Specificity**

Adult: Narrow/specialist

**Tolerance Ranges/Thresholds**

Adult: Low

**Site Fidelity**

Adult: High

**Dependency on Other Individuals or Species for Habitat**

Adult: No

**Habitat Narrative**

Adult: Habitat includes sedge-forb-willow zones along permanent streams (Jemez and Sacramento mountains); large, wet meadows in river floodplains (Rio Grande Valley); and narrow riparian zones along irrigation ditches (Bosque del Apache NWR). In many areas, moist riparian zones with tall, dense sedges provide suitable habitat; the presence of beavers is useful in maintaining habitat. Nests generally are in dry soils (NatureServe 2015). Populations are clumped within an extremely narrow habitat range, but individuals in the population are solitary. Key resources needed for habitat include permanent flowing water, saturated soils, vegetation dominated by sedges or forbs, and adjacent upland grassland habitat (USFWS 2013).

***Dispersal/Migration*****Motility/Mobility**

Adult: Low; limited vagility (ability to move) (USFWS 2013).

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Low

**Immigration/Emigration**

Adult: Unlikely due to habitat fragmentation.

**Dependency on Other Individuals or Species for Dispersal**

Adult: No

**Dispersal/Migration Narrative**

Adult: The motility of the New Mexico meadow jumping mouse is low due to limited vagility of the subspecies (USFWS 2013). They are nonmigratory and have a low capability for dispersal and/or immigration/emigration. Dispersal is dependent on habitat continuity between populations.

***Population Information and Trends*****Population Trends:**

Drastic decline (NatureServe 2015).

**Species Trends:**

The short-term trend is not known, but is presumed to be stable or slowly declining. The long term trend is greater than 70 percent decline (NatureServe 2015).

**Resiliency:**

Low

**Representation:**

Low

**Redundancy:**

Low

**Population Growth Rate:**

Stable to slowly declining (NatureServe 2015).

**Number of Populations:**

Since 2005, there have been 29 documented remaining populations spread across eight conservation areas (USFWS 2014).

**Population Size:**

250 to 2,500 individuals (NatureServe 2015).

**Resistance to Disease:**

Moderate

**Adaptability:**

Low

**Population Narrative:**

There has been a drastic decline in the number of occupied localities across the range of the species in New Mexico and Arizona. Of the original 98 known historical localities, there are currently 10 known extant localities in New Mexico, one locality in Arizona, and an additional

eight localities that have not been surveyed since the early to mid-1990s. The short-term species level trend is not known, but is presumed to be stable or slowly declining. The long term species level trend is greater than 70 percent decline (NatureServe 2015). Since 2005, there have been 29 documented remaining populations spread across eight conservation areas (USFWS 2014), with a population size estimated to be between 250 and 2,500 individuals (NatureServe 2015).

### ***Threats and Stressors***

**Stressor:** Habitat loss

**Exposure:** Destruction and degradation of wetlands and riparian zones due to grazing, lack of water due to drought, scouring floods, and wildfires.

**Response:** Poor habitat connectivity and lack of suitable habitat.

**Consequence:** Cumulative habitat loss has resulted in the extirpation of historical populations, reduced the size of existing populations, and isolated existing small populations.

**Narrative:** Reduction of the amount of suitable habitat due to destruction and degradation of wetlands and riparian habitat leads to poor habitat connectivity and habitat loss. This reduces the carrying capacity and eliminates populations (79 FR 33119).

**Stressor:** Vulnerability of extinction

**Exposure:** Highly fragmented distribution and limited dispersal ability.

**Response:** Increased risk of further extirpation.

**Consequence:** Continuing declines and small populations decrease the ability of this subspecies to recolonize and to survive stochastic events.

**Narrative:** Due to habitat destruction and fragmentation, the range of this subspecies has been considerably reduced. Based on the small population size, the New Mexico meadow jumping mouse is vulnerable to continuing declines (79 FR 33119).

### ***Recovery***

#### **Reclassification Criteria:**

Develop reclassification criteria as part of a Recovery Plan.

#### **Delisting Criteria:**

Develop delisting criteria as part of a Recovery Plan.

#### **Recovery Actions:**

- Establish partnerships to design and install effective barriers or exclosures, or change livestock management techniques (e.g., fencing, reconfiguration of grazing units, offsite water development, or changing the timing or duration of livestock use) to limit ungulate grazing and protect riparian habitats from damage.
- Work cooperatively with stakeholders to maintain the required microhabitat components, or modify or limit actions (e.g., bridge and road realignment projects, water use and management, stream restoration, and vegetation management) that preclude their development and restoration, to stabilize and expand current jumping mouse populations.
- Identify priority areas to reduce fuels to minimize the risk of severe wildland fire, and identify techniques for post-fire stabilization in areas that burn.
- Modify off-road vehicle use, and manage dispersed recreation through fencing, signage, education, and timing of use.

- Facilitate the natural expansion of jumping mouse habitat through the management and restoration of beaver. In New Mexico, beaver can no longer be relocated or transplanted without written consent from all property owners, land management agencies, or other affected parties (e.g., irrigation districts) within an 8-kilometer (5-mile) radius of the proposed release site or connective waters.
- Complete an emergency contingency and salvage plan to capture jumping mice and bring individuals into captivity in the event of severe wildlife fire, post-fire flooding, or severe drought.
- Establish a monitoring protocol to determine presence/absence or estimate the abundance of jumping mouse populations.
- Investigate the genetic diversity of populations to identify and address where long-term management strategies may be needed to enhance their genetic integrity.
- Formally evaluate whether assisted translocation or a captive breeding program for jumping mice would be beneficial as a recovery option.
- Conduct research on the critical aspects of jumping mouse life history (e.g., reproduction, abundance, survival, and movement behavior).

***Conservation Measures and Best Management Practices:***

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***Additional Threshold Information:***

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## SPECIES ACCOUNT: *Ursus maritimus* (Polar bear)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Threatened; 05/15/2008; Alaska Region (R7) (USFWS, 2016)

### **Physical Description**

Polar bears are carnivorous, and a top predator of the Arctic marine ecosystem. The polar bear is usually considered a marine mammal since its primary habitat is the sea-ice (Amstrup 2003), and it is evolutionarily adapted to life on sea-ice. Polar bears are the largest of the living bear species (DeMaster and Stirling 1981; Stirling and Derocher 1990). They are characterized by large body size, a stocky form, and fur color that varies from white to yellow. They are sexually dimorphic; females weigh 181 - 317 kilograms (kg) (400 - 700 pounds (lbs)), and males up to 654 kg (1,440 lbs).

### **Current Range**

Polar bears are widely distributed throughout the Arctic where the sea is ice-covered for large portions of the year (Figure A-23). The number of polar bears is estimated to be 20,000 - 25,000 with 19 recognized management subpopulations or "stocks" (Obbard et al. 2010). The International Union for Conservation of Nature and Natural Resources, Species Survival Commission (IUCN/SSC) Polar Bear Specialist Group (PBSG) ranked nine stocks as data deficient, six as stable, three as declining, and one as increasing (IUCN/PBSG 2015). The status designation of "data deficient" for nine stocks indicates that the estimate of the worldwide polar bear population was made with known uncertainty. The population estimate for the Southern Beaufort stock is 907 whereas the population within the Chukchi Sea population is unknown (IUCN/PBSG 2015).

### **Critical Habitat Designated**

Yes;

### **Legal Description**

On December 7, 2010, the U.S. Fish and Wildlife Service (Service), designated critical habitat for polar bear (*Ursus maritimus*) populations in the United States under the Endangered Species Act of 1973, as amended (Act). In total, approximately 484,734 square kilometers (km<sup>2</sup>) (187,157 square miles (mi<sup>2</sup>)) fall within the boundaries of the critical habitat designation.

### **Critical Habitat Designation**

The three units designated as critical habitat are: (1) Sea-ice Habitat; (2) Terrestrial Denning Habitat; and (3) Barrier Island Habitat.

**Unit 1: Sea-Ice Habitat** Unit 1 consists of approximately 464,924 km<sup>2</sup> (179,508 mi<sup>2</sup>) of the seaice habitat ranging from the mean high tide line to the 300-m (984.2-ft) depth contour. Because we are limited by 50 CFR 424.12(h) to designating critical habitat only on lands and waters under U.S. jurisdiction, Unit 1 does not extend beyond the U.S. 370.7 nautical-km (200 nm) EEZ to the north, the International Date Line to the west, or the United States–Canada border to the east. To delineate the southern boundary, we used the southern extent of the Chukchi-Bering Seas population as determined by telemetry data (Garner et al. 1990, p. 223), because the 300-m (984.2-ft) depth contour extends beyond the southern extent of the polar bear population. The

vast majority (92 percent) of Unit 1 is located within Federal waters. Unit 1 contains PCE number 1, which is required for feeding, breeding, denning, and movements that are essential for the conservation of polar bear populations in the United States. Special management considerations and protection may be needed to minimize the risk of crude oil spills associated with oil and gas development and production, oil and gas tankers, and the risks associated with commercial shipping within this region and along the Northern Sea Route.

**Unit 2: Terrestrial Denning Habitat** Unit 2 consists of an estimated 14,652 km<sup>2</sup> (5,657 mi<sup>2</sup>) of land, located along the northern coast of Alaska, with the appropriate denning macrohabitat and microhabitat characteristics (Durner et al. 2001, p. 118), as described under “Terrestrial Denning Habitat Criteria” above. The area designated as critical habitat contains approximately 95 percent of the known historical den sites from the southern Beaufort Sea population (Durner et al. 2009b, p. 3). The inland extent of denning distinctly varied between two longitudinal zones, with 95 percent of the polar bear dens between the Kavik River and the United States-Canada border occurring within 32 km (20 mi) of the mainland coast, and 95 percent of the dens between the Kavik River and Barrow occurring within 8 km (5 mi) of the mainland coast. We did not identify denning habitat for the Chukchi-Bering Seas population in western Alaska because coastal areas in western Alaska do not contain the “access via sea-ice” component of the terrestrial denning habitat PCE. Historically most of these polar bears den on Wrangel Island and Chukotka Peninsula, Russia. Typically polar bears follow the northerly retreat of the sea ice and are precluded from denning on the western coast of Alaska due to extreme open-water fetch and late ice freeze-up. Increases in the length of the open-water season along with declines in the sea ice extent will likely exacerbate this phenomenon. Twenty percent, 74 percent, and 6 percent of Unit 2 is located within State of Alaska land, Federal lands, and Native-owned lands, respectively. In addition, 53.3 percent of the land included within Unit 2 occurs within the boundaries of the Arctic National Wildlife Refuge. Unit 2 contains the necessary topographic, macrohabitat, and microhabitat features identified in PCE 2 that are essential for the conservation of polar bears in the United States. Special management considerations and protection may be needed to minimize the risk of human disturbances and crude oil spills associated with oil and gas development and production, and the risk associated with commercial shipping.

**Unit 3: Barrier Island Habitat** Unit 3 consists of an estimated 10,576 km<sup>2</sup> (4,083 mi<sup>2</sup>) of barrier island habitat. Barrier island habitat includes the barrier islands themselves and associated spits, and the water, ice, and any other terrestrial habitat within 1.6 km (1 mi) of the islands. Approximately sixty-four percent of Unit 3 consists of State of Alaska owned land and jurisdictional waters; 18.1 percent consists of Alaska Native owned land, and 17.6 percent consists of Federal Government owned land. Unit 3 contains PCE number 3, which is essential for the conservation of polar bear populations in the United States. Coastal barrier islands and spits off the Alaska coast provide areas free from human disturbance and are important for denning, resting, and movements along the coast to access maternal den and optimal feeding habitat. Special management considerations and protection may be needed to minimize the risk of human disturbances, shipping, and crude oil spills associated with oil and gas development and production, oil and gas tankers, and other marine vessels.

#### **Primary Constituent Elements/Physical or Biological Features**

Critical habitat areas are in the State of Alaska, and adjacent territorial and U.S. waters, as described below. The primary constituent elements of critical habitat for the polar bear in the United States are:

(i) Sea-ice habitat used for feeding, breeding, denning, and movements, which is sea ice over waters 300 m (984.2 ft) or less in depth that occurs over the continental shelf with adequate prey resources (primarily ringed and bearded seals) to support polar bears.

(ii) Terrestrial denning habitat, which includes topographic features, such as coastal bluffs and river banks, with the following suitable macrohabitat characteristics: (A) Steep, stable slopes (range 15.5– 50.0°), with heights ranging from 1.3 to 34 m (4.3 to 111.6 ft), and with water or relatively level ground below the slope and relatively flat terrain above the slope; (B) Unobstructed, undisturbed access between den sites and the coast; (C) Sea ice in proximity to terrestrial denning habitat prior to the onset of denning during the fall to provide access to terrestrial den sites; and (D) The absence of disturbance from humans and human activities that might attract other polar bears.

(iii) Barrier island habitat used for denning, refuge from human disturbance, and movements along the coast to access maternal den and optimal feeding habitat, which includes all barrier islands along the Alaska coast and their associated spits, within the range of the polar bear in the United States, and the water, ice, and terrestrial habitat within 1.6 km (1 mi) of these islands (no-disturbance zone).

### **Special Management Considerations or Protections**

Critical habitat does not include manmade structures (e.g., houses, gravel roads, generator plants, sewage treatment plants, hotels, docks, seawalls, pipelines) and the land on which they are located existing within the boundaries of designated critical habitat on the effective date of this rule.

Potential impacts that could harm the identified essential physical and biological features include reductions in the extent of arctic sea ice due to climate change; oil and gas exploration, development, and production; human disturbance; and commercial shipping. Pollution from various potential sources, including oil spills from vessels, or discharges from oil and gas drilling and production, could render areas containing the identified physical and biological features unsuitable for use by polar bears, effectively negating the conservation value of these features. Because of the vulnerabilities to pollution sources, these features may require special management considerations or protection through such measures as placing conditions on Federal permits or authorizations to stimulate special operational restraints, mitigative measures, or technological changes. Special management considerations and protection may be needed to minimize the risk of crude oil spills and human disturbance associated with oil and gas development and production, oil and gas tankers, and potential commercial shipping along the Northern Sea Route to polar bears and the habitat features essential to their conservation.

### ***Life History***

#### **Feeding Narrative**

Adult: Ringed seals (*Phoca hispida*) are polar bear's primary food source, and, to a lesser extent, bearded seals (*Erignathus barbatus*), but bears they may occasionally consume other marine mammals such as walruses (*Odobenus rosmarus*), narwhal (*Monodon monoceros*), and belugas (*Delphinapterus leucas*) (Kiliaan and Stirling 1978; Smith 1980; Smith 1985; Lowry et al. 1987). On average, an adult polar bear needs approximately 2 kg (4.4 lbs) of seal fat per day to survive

(Best 1985). Sufficient nutrition is critical and may be obtained and stored as fat when prey is abundant (Smith and Sjare 1990). Bowhead whale carcasses have been available as a food source on the North Slope since the early 1970s and may affect local polar bear distributions. Record numbers of polar bears were observed in 2012 in the vicinity of the bowhead whale carcass “bonepile” on Barter Island; the USFWS observed a minimum, maximum, and average of 24, 80, and 52 bears, respectively (USFWS 2012). Barter Island (near Kaktovik) has had the highest recorded concentration of polar bears on shore ( $17.0 \pm 6.0$  polar bears/100 km) followed by Barrow ( $2.2 \pm 1.8$ ) and Cross Island ( $2.0 \pm 1.8$ ) (Schliebe et al. 2008). The high number of bears on/near Barter Island compared to other areas is thought to be due in part to the proximity to the ice edge and high ringed seal densities (Schliebe et al. 2008); the whale harvest at Kaktovik is lower than that at Barrow or Cross Island. The use of whale carcasses as a food source likely varies among individuals and years. Stable isotope analysis of polar bears in 2003 and 2004 suggested that bowhead whale carcasses comprised 11%-26% (95% CI) of the diets of sampled polar bears in 2003, and 0-14% (95% CI) in 2004 (Bentzen et al. 2007). Because polar bears depend on sea-ice to hunt seals, and temporal and spatial availability of sea-ice will likely decline, polar bear use of whale carcasses may increase.

### Reproduction Narrative

Adult: Polar bears are characterized by a late age of sexual maturity, small litter sizes, and extended parental investment in raising young, factors that combine to contribute to a very low reproductive rate (Schliebe et al. 2006). Females may give birth for the first time at age four to six depending on local conditions such as seal abundance (Schliebe et al. 2006), and litters per female varies from 0.25 to 0.45 per adult female (Schliebe et al. 2006). Likewise, litter size and production rate vary geographically with hunting pressure, environmental factors and other population perturbations. Two-cub litters are most common (Schliebe et al. 2006). Body weights of mothers and their cubs decreased markedly in the mid-1970s in the Beaufort Sea following a decline in ringed (*Phoca hispida*) and bearded (*Erignathus barbatus*) seal pup production (Stirling et al. 1977; Kingsley 1979; DeMaster et al. 1980; Amstrup et al. 1986). Declines in reproductive parameters varied by region and year with the severity of ice conditions and corresponding reduction in numbers and productivity of seals (Amstrup et al. 1986). Most stocks use terrestrial habitat partially or exclusively for maternity denning; therefore, females must adjust their movements to access land at the appropriate time (Stirling 1988; Derocher et al. 2004). Most pregnant female polar bears excavate dens in the fall-early winter period (Harington 1968; Lentfer and Hensel 1980; Ramsay and Stirling 1990). The only known exceptions are in Western and Southern Hudson Bay where polar bears excavate earthen dens and later reposition into adjacent snow drifts (Jonkel et al. 1972, Richardson et al. 2005), and in the southern Beaufort Sea where a portion of the population dens in snow caves on sea-ice (Schliebe et al. 2006). Polar bears give birth in the dens during midwinter (Kostyan 1954; Harington 1968). Family groups emerge from dens in March and April when cubs are approximately three months old (Schliebe et al. 2006).

### Environmental Specificity

Adult: Narrow. Specialist or community with key requirements common. (Natureserve, 2015)

### Habitat Narrative

Adult: Polar bears are closely tied to arctic pack ice. They prefer areas with ice that is periodically active, such as at the interface of landfast ice and drifting pack ice along the arctic coasts or near polynyas. Polar bears show a preference for sea ice located over and near the

continental shelf, likely due to higher biological productivity in these areas and greater accessibility to prey in near-shore shear zones and polynyas (areas of open sea surrounded by ice) compared to deep-water regions in the central polar basin; they are most abundant near the shore in shallow-water areas, and also in other areas where currents and ocean upwelling increase marine productivity and serve to keep the ice cover from becoming too consolidated in winter (see USFWS 2008 for specific sources of this information). Sometimes polar bears wander inland as much as 150 km from the coast. In the Bering and Chukchi Seas, Alaska, where sea ice melts in summer, bears migrate up to 1,000 km to remain with the southern ice boundary (Garner et al. 1990, 1994 in Amstrup 2003); in Hudson Bay, James Bay and parts of the Canadian Arctic, bears may be forced onto land for up to several months when sea ice melts in summer (Jonkel et al. 1976, Lunn et al. 1997 in Amstrup 2003). During ice-free period along western Hudson Bay, adult males occupy the coast while family groups and pregnant females occur farther inland. Pregnant females remain on or near land in dens through winter while males and non-breeders winter on sea ice. On land, range of subadults overlaps that of adult males (Derocher and Stirling 1990). Female denning habitat may be found in mountain, fjord, or even relatively flat tundra areas, but generally it is near the coast and contains microhabitats which catch and collect snow in fall and early winter (Amstrup 2003). While most denning occurs on coastlines, bears may also den on drifting pack ice and on land-fast ice adjacent to shore (Amstrup and Gardner 1994). Females typically dig maternity dens in a hillside snowbank (in southwestern Hudson Bay, however, pregnant females commonly overwinter in earth dens 20-100 km from the coast). Dens often are built within 8 km of coast and rarely more than 48 km offshore (though sometimes in active offshore pack ice as much as 550 km north of Alaskan coast). Polar bears exhibit a general fidelity to denning areas and even after months of long-distance passive transportation on sea ice females often return to specific den habitats (Amstrup 2003). Near shore Tidal flat/shoreline; Tundra (NatureServe, 2015)

### ***Dispersal/Migration***

### **Motility/Mobility**

Adult: High

### **Dispersal/Migration Narrative**

Adult: Over most of their range, polar bears remain on the sea-ice year-round or spend only short periods on land. Sea-ice provides a platform for hunting and feeding, seeking mates and breeding, denning, resting, and long-distance movements. Areas near ice edges, leads, or polynyas where ocean depth is minimal are the most productive hunting grounds (Durner et al. 2004). However, some polar bear populations occur in seasonally ice-free environs and use land habitats for varying portions of the year. In the Chukchi Sea and Beaufort Sea areas of Alaska and northwestern Canada, for example, less than 10% of the polar bear locations obtained via radio telemetry were on land (Amstrup 2000; Amstrup, USGS, unpublished data); the majority of land locations were maternal dens during the winter. A similar pattern was found in East Greenland (Wiig et al. 2003). In the absence of ice during the summer season, some populations of polar bears in eastern Canada and Hudson Bay remain on land for extended periods of time until ice forms providing a platform for them to move to sea. Similarly, in the Barents Sea, a portion of the population spends greater amounts of time on land. Although polar bears are generally limited to areas where the sea is ice-covered for much of the year, they are not evenly distributed throughout their range on sea-ice. They show a preference for certain sea-ice characteristics, concentrations, and specific sea-ice features (Stirling et al. 1993; Mauritzen et al.

2001; Durner et al. 2004). Sea-ice habitat quality varies temporally as well as geographically (Amstrup et al. 2000). Polar bears show a preference for sea-ice located over and near the continental shelf (Derocher et al. 2004; Durner et al. 2004), likely due to higher biological productivity in these areas (Dunton et al. 2005) and greater accessibility to prey in nearshore shear zones and polynyas (areas of open sea surrounded by ice) compared to deep-water regions in the central polar basin (Stirling 1997). Bears are most abundant near the shore in shallow-water areas, and also in other areas where currents and ocean upwelling increase marine productivity and serve to keep the ice cover from becoming too consolidated in winter (Amstrup and Demaster 1988; Amstrup et al. 2000). Polar bear distribution in most areas varies seasonally with the seasonal extent of sea-ice cover and availability of prey. In Alaska in the winter, sea-ice may extend 400 km (248 mi) south of the Bering Strait, and polar bears will extend their range to the southernmost proximity of the ice (Ray 1971). Sea-ice disappears from the Bering Sea and is greatly reduced in the Chukchi Sea in the summer, and polar bears occupying these areas move as much as 1,000 km (621 mi) to stay with the pack ice (Garner et al. 1990; Garner et al. 1994). Significant northerly and southerly movements of polar bears appear to depend on seasonal melting and refreezing of ice (Amstrup 2000). In other areas, for example, when the sea-ice melts in Hudson Bay, James Bay, Davis Strait, Baffin Bay, and some portions of the Barents Sea, polar bears remain on land for up to four or five months while they wait for winter and new ice to form (Schweinsburg 1979; Prevett and Kolenosky 1982; Schweinsburg and Lee 1982; Ferguson et al. 1997). In areas where sea-ice cover is seasonally dynamic, a large multi-year home range, of which only a portion may be used in any one season or year, is an important part of the polar bear life history strategy. In other regions, where ice is less dynamic, home ranges are smaller and less variable (Ferguson et al. 2001). Data from telemetry studies of adult female polar bears show that they do not wander aimlessly on the ice, nor are they carried passively with the ocean currents as previously thought (Pedersen 1945 cited in Amstrup 2003). Results show strong fidelity to activity areas that are used over multiple years (Ferguson et al. 1997). All areas within an activity area are not used each year.

### ***Population Information and Trends***

#### **Number of Populations:**

6 - 20 (NatureServe, 2015)

#### **Population Size:**

10,000 - 100,000 individuals (NatureServe, 2015)

#### **Population Narrative:**

Total population is 20,000-25,000 (see USFWS 2008). Based on movement patterns and spatial segregation with limited interchange, the IUCN/SSC Polar Bear Specialist Group and USFWS (2008) recognized 19 relatively discrete (but variously overlapping) populations or management units (Stirling 1991, Amstrup 2003), which correspond with major occupied ecogeographic units. (NatureServe, 2015)

### ***Threats and Stressors***

**Stressor:** Climate Change

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Loss of sea-ice habitat due to climate change is identified as the primary threat to polar bears (Schliebe et al. 2006; 73 FR 28212; Obbard et al. 2010). Warming-induced habitat degradation and loss are negatively affecting some polar bear stocks, and unabated global warming will ultimately reduce the worldwide polar bear population (Obbard et al. 2010). Arctic summer sea-ice reached its lowest average extent in 2012 and has declined 13% per decade since 1979 (NSIDC 2012; Figure A-29). The loss rate of ice thickness is increasing (Haas et al. 2010), and trends in arctic sea-ice extent are declining (-12.2% and -13.5% per decade, respectively; Comiso 2012). Declines in sea-ice are more pronounced in summer than winter (NSIDC 2011a, b). Positive feedback systems (i.e., sea-ice albedo) and naturally-occurring events such as warm water intrusion into the arctic and changing atmospheric wind patterns can cause fragmentation of sea-ice, reduction in the extent and area of sea-ice in all seasons, retraction of sea-ice away from productive continental shelf areas throughout the polar basin, reduction of the amount of heavier and more stable multi-year ice, and declining thickness and quality of shore-fast ice (Parkinson et al. 1999; Rothrock et al. 1999; Comiso 2003; Fowler et al. 2004; Lindsay and Zhang 2005, Holland et al. 2006; Comiso 2006; Stroeve et al. 2008). These climatic phenomena may affect seal abundances, the polar bear's main food source (Kingsley 1979; DeMaster et al. 1980; Amstrup et al. 1986; Stirling 2002). Patterns of increased temperatures, earlier spring thaw, later fall freeze-up, increased rain-on-snow events (which can cause dens to collapse), and potential reductions in snowfall are also occurring. As stated above, the polar bear depends on sea-ice for its survival, and loss of sea-ice due to climate change is its largest threat worldwide. However, threats to polar bears will likely occur at different rates and times across their range, and uncertainty regarding their prediction makes management difficult (Obbard et al. 2010). Natural sources of mortality among polar bears are not well understood (Amstrup 2003). Polar bears are longlived (up to 30 years in captivity); have no natural predators, except other polar bears; and do not appear prone to death by diseases or parasites (Amstrup 2003). Accidents and injuries incurred in the dynamic and harsh sea-ice environment, injuries incurred while fighting other bears, starvation (usually during extreme youth or old age), freezing (also more common during extreme youth or old age), and drowning are all known natural causes of polar bear mortality (Amstrup 2003). Cannibalism by adult males on cubs and other adult bears is also known to occur; however, it is not thought that this is a common or significant cause of mortality. After natural causes and old age, the most significant source of polar bear mortality is from humans hunting polar bears (Amstrup 2003).

**Stressor:** Subsistence Hunting

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The largest human-caused loss of polar bears is from subsistence hunting of the species, but for most subpopulations where subsistence hunting of polar bears occurs, it is a regulated and/or monitored activity (Obbard et al. 2010). Polar bears historically have been, and continue to be, an important renewable resource for coastal communities throughout the Arctic (Amstrup and DeMaster 1988; Schliebe et al. 2006). Prior to the 1950s, most hunting was by indigenous people for subsistence purposes. Increased sport hunting in the 1950s and 1960s resulted in population declines (Prestrud and Sterling 1994). International concern about the status of polar bears resulted in biologists from the five polar bear range nations forming the PBSG within the IUCN. The PBSG was largely responsible for the development and ratification of the 1973 International Agreement on the Conservation of Polar Bears (1973 Polar Bear

Agreement) (Prestrud and Sterling 1994). The 1973 Polar Bear Agreement and the actions of the member nations are credited with the recovery of polar bears following the previous period of overexploitation. The various polar bear subpopulations face different levels of subsistence harvest pressure; some level of hunting is permitted in the U.S., Canada, Greenland, and recently, in the Russian Federation as well. Five populations (including four that are hunted) have no estimate of potential risk from overharvest, since adequate demographic information necessary to conduct a population viability analysis and risk assessment are not available. The Chukchi Sea, Baffin Bay, Kane Basin, and Western Hudson Bay populations may be overharvested (Aars et al. 2006). In other populations, including East Greenland and Davis Strait, substantial harvest occurs annually in the absence of scientifically derived population estimates (Aars et al. 2006). Considerable debate has occurred regarding the recent changes in population estimates based on indigenous or local knowledge and subsequent quota increases for some populations in Nunavut (Aars et al. 2006). Increased polar bear observations along the coast may be attributed to changes in bear distribution due to lack of suitable ice habitat rather than to increased population size (Stirling and Parkinson 2006). Additional data are needed to reconcile these differing interpretations. Amstrup et al. (2007) used a Bayesian network model to forecast the range-wide status of polar bears during the 21st century, factoring in a number of stressors, including intentional take or harvest. The authors conducted a sensitivity analysis to determine the importance and influence of the stressors on the population forecast. Their analysis indicated that intentional take was the 4th-ranked potential stressor, and could exacerbate the effects of habitat loss in the future. The relatively high ranking for this stressor indicates that effective management of hunting and evaluation of sustainable harvest levels will continue to be important to minimize effects for populations experiencing increased stress

**Stressor:** Other threats

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Other sources of polar bear mortality related to human activities, though few and very rare, include research activities and defense of life kills by non-Natives (Brower et al. 2002). Accumulation of persistent organic pollutants in polar bear tissue, tourism, human-bear conflict, and increased development in the Arctic are also sources of concern (Obbard et al. 2010). Because uncertainty exists regarding how human activities interact to ultimately affect the world-wide polar bear population, conservation and management of polar bears at the world-wide population level is challenging.

## ***Recovery***

### **Delisting Criteria:**

The worldwide probability of persistence is at least 95% over 100 years (USFWS 2016).

The probability of persistence in each recovery unit (ecoregion) is at least 90% over 100 years (USFWS 2016).

The mean adult female survival rate (at a density corresponding to maximum net productivity level and in the absence of direct human-caused removals) in each recovery unit is at least 93–96%, both currently and as projected over the next 100 years (USFWS 2016).

The ratio of yearlings to adult females (at a density corresponding to maximum net productivity level) in each recovery unit is at least 0.1–0.3, both currently and as projected over the next 100 years (USFWS 2016).

The carrying capacity, distribution, and connectivity in each recovery unit, both currently and as projected over the next 100 years, are such that the probability of persistence over 100 years is at least 90% (USFWS 2016).

Total direct human-caused removals in each recovery unit do not exceed a rate  $h$  (relative to the population size in the recovery unit) that maintains the population above its maximum net productivity level relative to carrying capacity (USFWS 2016).

In each recovery unit, either (a) the average annual ice-free period is expected not to exceed 4 months over the next 100 years based on model projections using the best available climate science, or (b) the average annual ice-free period is expected to stabilize at longer than 4 months over the next 100 years based on model predictions using the best available climate science, and there is evidence that polar bears in that recovery unit can meet ESA Demographic Criteria 1, 2, and 3 under that longer ice-free period (USFWS 2016).

For each recovery unit, the total level of direct, lethal removals of polar bears by humans, in conjunction with other factors, does not reduce the probability of persistence below 90% over 100 years (USFWS 2016).

**Recovery Actions:**

- Limit global atmospheric levels of greenhouse gases to levels appropriate for supporting polar bear recovery and conservation, primarily by reducing greenhouse gas emissions (USFWS 2016).
- Support international conservation efforts through the Range States relationships (USFWS 2016).
- Manage human-bear conflicts (USFWS 2016).
- Collaboratively manage subsistence harvest (USFWS 2016).
- Protect denning habitat (USFWS 2016).
- Minimize risks of contamination from spills (USFWS 2016).
- Conduct strategic monitoring and research (USFWS 2016).

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## SPECIES ACCOUNT: *Trichechus manatus* (West Indian Manatee)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Threatened; 04/05/2017 Southeast Region (R4) (82 FR 16668).

### **Physical Description**

West Indian manatees are massive fusiform-shaped animals with skin that is uniformly dark grey, wrinkled, sparsely haired, and rubber-like. Manatees possess paddle-like forelimbs, no hind limbs, and a spatulate, horizontally flattened tail. Females have two axillary mammae, one at the posterior base of each forelimb (Fig. 2). Their bones are massive and heavy with no marrow cavities in the ribs or long bones of the forearms (Odell 1982). Adults average about 3.0 m (9.8 ft) in length and 1,000 kg (2,200 lbs) in weight, but may reach lengths of up to 4.6 m (15 ft) (Gunter 1941) and weigh as much as 1,620 kg (3,570 lbs) (Rathbun et al. 1990). Newborns average 1.2 to 1.4 m (4 to 4.5 ft) in length and about 30 kg (66 lbs) (Odell 1981). The nostrils, located on the upper snout, open and close by means of muscular valves as the animals surface and dive (Husar 1977; Hartman 1979). A muscular flexible upper lip is used with the forelimbs to manipulate food into the mouth (Odell 1982). Bristles are located on the upper and lower lip pads. Molars designed to crush vegetation form continuously at the back of the jaw and move forward as older ones wear down (Domning and Hayek 1986). The eyes are very small, close with sphincter action, and are equipped with inner membranes that can be drawn across the eyeball for protection. Externally, the ears are minute with no pinnae (USFWS, 2001).

### **Taxonomy**

Domning and Hayek (1986) identified separate subspecies of the West Indian manatee in Florida (*Trichechus manatus latirostris*) and the Caribbean (*Trichechus manatus manatus*), based on cranial measurements. The distinctive morphological features are generally thought to be the result of and reflective of population isolation, where certain anatomical features are favored by adaptation. These subspecies will continue to be recognized and used unless future analyses prove otherwise (USFWS, 2001).

### **Historical Range**

Range encompasses rivers, estuaries, and coastal areas of subtropical and tropical areas of northern South America, West Indies/Caribbean region (but apparently never very abundant in the Greater Antilles, except perhaps Cuba, Lefebvre 1989), Gulf of Mexico (now mainly western and southwestern portions), and southeastern North America (mainly Florida). U.S. populations occur primarily in Florida (e.g., see Van Meter 1987), where they are effectively isolated from other populations by the cooler waters of the northern Gulf of Mexico and the deeper waters of the Straits of Florida (Domning and Hayek 1986). In the southeastern United States, manatees are more or less restricted to the vicinity of warm-water sites in peninsular Florida during the winter, although a few may remain year-round in Cumberland Sound, southeastern Georgia, where factory warm-water outfalls allow survival of colder winter months (Reeves et al. 1992). Occasional manatees occur in summer from Texas to North Carolina (e.g., see Schwartz 1995, *Brimleyana* 22:53-60, for North Carolina records). Those in Texas may be wanderers from Mexican population, but DNA analysis of an individual captured linked it to the Florida population (T. Ettel, pers. comm.). Manatees range along most of the Gulf coast of Florida but infrequently occur north of the Suwannee River and between the Chassahowitzka River and Tampa Bay. They inhabit the Atlantic coast of Florida from the Georgia coast to Biscayne Bay

and the Florida Keys, including the St. Johns River, the Indian River lagoon system, and various other waterways (O'Shea and Ludlow 1992). In Florida, the most well-used wintering areas are at Crystal River, Homosassa River, Tampa Bay, Ft. Myers, Port Everglades, Riviera Beach, near Titusville, and Blue Spring (O'Shea and Ludlow 1992) (NatureServe, 2015).

**Current Range**

Present range limits are similar to those known historically, but the distribution is fragmented due to areas of local extirpation (O'Shea and Ludlow 1992). Area of occupancy and abundance are apparently greatly reduced in Central and South America compared to the historical situation. Small numbers exist in the Greater Antilles but the species has not been documented in the Lesser Antilles south of the Virgin Islands since the 1700s. Sightings are rare in the Bahamas. Manatees remain relatively abundant in Belize (compared to elsewhere in Central America) and in Guyana, and they are still reasonably abundant in some areas of Mexico and on both coasts of Florida (Lefebvre et al. 1989). In Puerto Rico, manatees are most often observed in coastal areas from San Juan eastward to the east coast, (and including Vieques Island) and then south and west, past Jobos Bay, to the west coast, and about as far to the northwest as Rincon; they are concentrated in several areas, including Ceiba, Vieques Island, Jobos Bay, and Boquerón Bay, and are less abundant along the north coast, between Rincón and Dorado (USFWS 2007). Manatees are very rare (transient) in the U.S. Virgin Islands (USFWS 2007). See Fairbairn and Haynes (1982) and Hurst (1986) for information on status and distribution in Jamaica. See Lefebvre et al. (1989) for a fairly detailed overview of country by country status (USFWS, 2015).

**Distinct Population Segments Defined**

No

**Critical Habitat Designated**

Yes; 9/24/1976.

**Legal Description**

On September 24, 1976, the Director, U.S. Fish and Wildlife Service (hereinafter, the "Director" and the "Service" respectively) hereby issues a Rulemaking pursuant to Section 7 of the Endangered- Species Act of 1973 (16 U.S.C. 1531-1543; 87 Stat. 884; hereinafter, the "Act") which determines Critical Habitat for the American Crocodile (*Crocodylus acutus*), California Condor (*Gymnogyps californianus*), Indiana Bat (*Myotis sodalis*), and Florida Manatee (*Trichechus manatus*).

**Critical Habitat Designation**

The following areas (exclusive of those existing manmade structures or settlements which are not necessary to the normal needs or survival of the species) are Critical Habitat for the Species indicated. Pursuant to Section 7 of the Act, all Federal agencies must insure that actions authorized, funded, or carried out by them do not result in the destruction or adverse modification of these areas:

Florida. Crystal River and its headwaters known as King's Bay, Citrus County: the Little Manatee River downstream from the U.S. Highway 301 bridge. Hillsborough County: the Manatee River downstream from the Lake Manatee Dam, Manatee County; the Myakka River downstream from Myakka River State Park. Sarasota and Charlotte Counties: the Peace River downstream from the

Florida State Highway 760 bridge, De Soto and Charlotte Counties: Charlotte Harbor north of the Charlotte-Lee county line, Charlotte County: Caloosahatchee River downstream from the Florida State Highway 31 bridge, Lee County; all U.S. territorial waters adjoining the coast and islands of Lee County; all U.S. territorial waters adjoining the coast and Islands and all connected bays, estuaries, and rivers from Gordon's Pass, near Naples. Collier County, southward to and including Whitewater Bay, Monroe County; all waters of Card, Barnes, Blackwater, Little Blackwater, Manatee, and Buttonwood sounds between Key Large, Monroe County, and the mainland of Dade County: Biscayne Bay, and all adjoining and connected lakes, rivers, canals, and waterways from the southern tip of Key Biscayne northward to and including Maule Lake, Dade County; all of Lake Worth, from its northernmost point immediately south of the intersection of U.S. Highway 1 and Florida State Highway A1A -southward to its southernmost point immediately north of the town of Boynton Beach, Palm Beach County: the Loxahatchee River and its headwaters, Martin and West Palm Beach Counties; that section of the Intracoastal waterway from the town of Sewalls Point, Martin County to Jupiter Inlet, Palm Beach County: the entire inland section of water known as the Indian River, from its northernmost point immediately south of the intersection of U.S. Highway 1 and Florida State Highway 3. Volusia County, southward to its southernmost point near the town of Seawalls Point, Martin County, and the entire inland section of water known as the Banana River and all waterways between Indian and Banana rivers, Brevard County: the St. Johns River including Lake George, and including Blue Springs and Silver Glen Springs from their points of origin to their confluences with the St. Johns River; that section of the Intracoastal Waterway from its confluence with the St. Marys River on the Georgia-Florida border to the Florida State Highway A1A bridge south of Coastal City, Nassau and Duval Counties.

#### **Primary Constituent Elements/Physical or Biological Features**

With respect to the Florida manatee, the areas delineated below contain the largest concentrations in the United States. and are the only areas that presently can be defined as having major dependent populations. The Crystal River and Its King's Bay headwaters form one of the largest natural warm water resources for Manatees. Up to 60 Manatees possibly representing six to ten percent of the total population of the species in the United States utilize this refugium during cold weather periods. The Little Manatee, Manatee, Myakka, and Peace rivers, and Charlotte Harbor all support large Manatee concentrations. Manatees also utilize the Caloosahatchee River and associated coastal areas. The warm water discharge of the Florida Power and Light Company Ft. Meyers power plant into the Orange River on the south bank of the Caloosahatchee River at Tice is known to attract as many as 75 manatees during cold periods. The area off the coast of Collier and Monroe Counties, southwestern Florida is the center of a large, but uncounted Manatee population. This population is at least partially resident and is dependent on the extensive local growths of *Thalassia* and *Diplanthera* as a primary food resource. Concentrations of as many as 75 manatees are observed in Whitewater Bay. The waterway formed by Card, Barnes, Blackwater, and Buttonwood sounds may constitute the Manatee's essential thoroughfare between Miami-Biscayne Bay and the lower Keys and Florida Bay. Seaward movement along the upper Keys is very rare. Biscayne Bay, with its adjoining waterways are of central importance to the large Manatee populations of southeastern Florida, Abundant food resources exist in the area, and the warm water flow from the Florida Power and Light Company Miami River plant provides an important refugium. Lake Worth supports a large Manatee population year-round and also serves as a warm water refugium for additional wintering manatees. The outfall from the Florida Power and Light Company Miami River plant supports up to 75 manatees during cold weather. The Indian and Banana rivers may contain the

largest manatee population in Florida. These areas provide warm, quiet waters and abundant food resources. The St. Johns River also provides ample food resources to a significant manatees population, and several of its spring-fed tributaries provide warm water during cold spells. In Lake Monroe, two power plants provide warm water outfalls which are used by manatees during cold periods. The Intracoastal Waterway from the St. Marys River to Highway A1A is a major concentration area and thoroughfare for manatees.

PCEs not specifically described. From the text above, it can be presumed that the following are primary constituent elements:

- (1) Warm water during cold periods.
- (2) *Thalassia* and *Diplanthera* as a primary food resource.

### **Special Management Considerations or Protections**

Not available

### ***Life History***

#### **Feeding Narrative**

Adult: Manatees are herbivores that feed opportunistically on a wide variety of submerged, floating, and emergent vegetation. Because of their broad distribution and migratory patterns, Florida manatees utilize a wider diversity of food items and are possibly less specialized in their feeding strategies than manatees in tropical regions (Lefebvre et al. 2000). Feeding rates and food preferences depend, in part, on the season and available plant species. Bengtson (1981, 1983) reported that the time manatees spent feeding in the upper St. Johns River was greatest (6 to 7 hrs/day) before winter (August to November), least (3 to 4 hrs/day) in spring and summer (April to July), and intermediate (about 5 hrs/day) in winter (January to March). He estimated annual mean consumption rates at 33.2 kg/day/manatee or about 4 to 9% of their body weight per day depending on season (Bengtson 1983). At Crystal River, Etheridge et al. (1985) reported cumulative daily winter feeding times from 0 to 6 hrs. 10 min. based on observations of three radio-tagged animals over seven 24-hour periods. The estimated daily consumption rates by adults, juveniles, and calves eating hydrilla (*Hydrilla verticillata*) were 7.1, 9.6, and 15.7% of body weight per day, respectively. Seagrasses appear to be a staple of the manatee diet in coastal areas (Ledder 1986; Provancha and Hall 1991; Kadel and Patton 1992; Koelsch 1997; Lefebvre et al. 2000). Packard (1984) noted two feeding methods in coastal seagrass beds: (1) rooting, where virtually the entire plant is consumed; and (2) grazing, where exposed grass blades are eaten without disturbing the roots or sediment. Manatees may return to specific seagrass beds to graze on new growth (Koelsch 1997; Lefebvre et al. 2000). In the upper Banana River, Provancha and Hall (1991) found spring concentrations of manatees grazing in beds dominated by manatee grass (*Syringodium filiforme*). They also reported an apparent preference for manatee grass and shoalgrass (*Halodule wrightii*) over the macroalga *Caulerpa* spp. Along the Florida-Georgia border, manatees feed in salt marshes on smooth cordgrass (*Spartina alterniflora*) by timing feeding periods with high tide (Baugh et al. 1989; Zoodsma 1991) (USFWS, 2001).

#### **Reproduction Narrative**

Adult: Breeding takes place when one or more males (ranging from 5 to 22) are attracted to an estrous female to form an ephemeral mating herd (Rathbun et al. 1995). Mating herds can last up to 4 weeks, with different males joining and leaving the herd daily (Hartman 1979; Bengtson 1981; Rathbun et al. 1995. Cited in Rathbun 1999). Permanent bonds between males and females do not form. During peak activity, the males in mating herds compete intensely for access to the female (Fig. 9; Hartman 1979). Successive copulations involving different males have been reported. Some observations suggest that larger, presumably older, males dominate access to females early in the formation of mating herds and are responsible for most pregnancies (Rathbun et al. 1995), but males as young as three years old are spermatogenic (Hernandez et al. 1995). Although breeding has been reported in all seasons, Hernandez et al. (1995) reported that histological studies of reproductive organs from carcasses of males found evidence of sperm production in 94% of adult males recovered from March through November. Only 20% of adult males recovered from December through February showed similar production. Females appear to reach sexual maturity by about age five but have given birth as early as four (Marmontel 1995; Odell et al. 1995; O'Shea and Hartley 1995; Rathbun et al. 1995), and males may reach sexual maturity at 3 to 4 years of age (Hernandez et al. 1995). Manatees may live in excess of 50 years (Marmontel 1995), and evidence for reproductive senescence is unclear (Marmontel 1995; Rathbun et al. 1995). Catalogued Florida manatee CR 28, a wild manatee that overwinters in Crystal River, was last documented with a calf in 1998, at which time she was estimated to be at least 34 years of age (USGS-Sirenia, unpublished data). A captive animal, MSTm-5801, gave birth to a calf in 1990, at which time she was estimated to be 43 to 48 years of age (FWS, unpublished data). The length of the gestation period is uncertain but is thought to be between 11 and 14 months (Odell et al. 1995; Rathbun et al. 1995; Reid et al. 1995). The normal litter size is one, with twins reported rarely (Marmontel 1995; Odell et al. 1995; O'Shea and Hartley 1995; Rathbun et al. 1995). Calf dependency usually lasts one to two years after birth (Hartman 1979; O'Shea and Hartley 1995; Rathbun et al. 1995; Reid et al. 1995). Calving intervals vary greatly among individuals. They are probably often less than 2 to 2.5 years, but may be considerably longer depending on age and perhaps other factors (Marmontel 1995; Odell et al. 1995; Rathbun et al. 1995; Reid et al. 1995). Females that abort or lose a calf due to perinatal death may become pregnant again within a few months (Odell et al. 1995), or even weeks (Hartman 1979) (USFWS, 2001).

#### **Environmental Specificity**

Adult: Moderate (based on species habitat)

#### **Tolerance Ranges/Thresholds**

Adult: Low (inferred from USFWS, 2001)

#### **Habitat Narrative**

Adult: The Florida manatee lives in freshwater, brackish and marine habitats. Submerged, emergent, and floating vegetation are their preferred food. During the winter, cold temperatures keep the population concentrated in peninsular Florida and many manatees rely on the warm water from natural springs and power plant outfalls. During the summer they expand their range and on rare occasions are seen as far north as Rhode Island on the Atlantic coast and as far west as Texas on the Gulf coast. The most significant problem presently faced by manatees in Florida is death or injury from boat strikes. The long-term availability of warm-water refuges for manatees is uncertain if minimum flows and levels are not established for the natural springs on which many manatees depend, and as deregulation of the power industry in

Florida occurs. Their survival will depend on maintaining the integrity of ecosystems and habitat sufficient to support a viable manatee population (USFWS, 2001). Low tolerance ranges are inferred based on temperature requirements (USFWS, 2001).

***Dispersal/Migration*****Motility/Mobility**

Adult: High

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Yes (USFWS, 2001)

**Dispersal**

Adult: High (temperature dependant (USFWS, 2001)

**Immigration/Emigration**

Adult: Immigrates (USFWS, 2001)

**Dispersal/Migration Narrative**

Adult: This is a large aquatic mammal that is highly mobile. The species migrates to warm water outflows (power plant outflow and warm springs) during cold weather (USFWS, 2001). In addition, USFWS (2001) notes that immigration of animals from other populations to Crystal River and Blue Springs. The species has the ability to disperse if water temperatures are warm enough and have been found as far North as Rhode Island during certain years)

***Population Information and Trends*****Population Trends:**

Decreasing (NatureServe, 2015)

**Number of Populations:**

6-20 (NatureServe, 2015)

**Population Size:**

Total population size (2500 - 10,000 (NatureServe, 2015)

**Population Narrative:**

NatureServe, 2015) notes that the number of populations is between 6 and 20 and the overall population numbers are between 2,500 and 10,000. In addition long -term and short term population trends are declining. Moderate resiliency is inferred based on low number of offspring and gestation period and widespread populations (USFWS, 2001 and NatureServe, 2015). Low representation is inferred based on long gestation period, low number of offspring produced and relatively long parental care (USFWS, 2001). Low redundancy is inferred based on limited number of populations (USFWS, 2001).

***Threats and Stressors***

**Stressor:** Boat strikes (USFWS, 2001)

**Exposure:****Response:****Consequence:** Death/Injury**Narrative:** The most significant problem presently faced by manatees in Florida is death or serious injury from boat strikes (USFWS, 2001).**Stressor:** Lack of warm water refuge (USFWS, 2001)**Exposure:****Response:****Consequence:** Habitat Loss/Death**Narrative:** The availability of warm-water refuges for manatees is uncertain if minimum flows and levels are not established for the natural springs on which many manatees depend, and as deregulation of the power industry in Florida occurs (USFWS, 2001).**Stressor:** Human development (USFWS, 2001)**Exposure:****Response:****Consequence:** Habitat Loss**Narrative:** Consequences of an increasing human population and intensive coastal development are long-term threats to the Florida manatee. Their survival will depend on maintaining the integrity of ecosystems and habitat sufficient to support a viable manatee population (USFWS, 2001).**Stressor:** Red tide (USFWS, 2001)**Exposure:****Response:****Consequence:** Death**Narrative:** Although the exact mechanism of manatee exposure to the red tide brevetoxin is unknown in the 1982 and 1996 outbreaks, ingestion, inhalation, or both are suspected (Bossart et al. 1998). The critical circumstances contributing to high red tide-related deaths are concentration and distribution of the red tide, timing and scale of manatee aggregations, salinity, and timing and persistence of the bloom (Landsberg and Steidinger 1998). It is difficult to manage for these rare but catastrophic causes of mortality (USFWS, 2001).**Stressor:** Navigational Locks (USFWS, 2001)**Exposure:****Response:****Consequence:** Death**Narrative:** The next largest human-related cause of manatee deaths is entrapment or crushing in water control structures and navigational locks and accounts for 4% of the total mortality between 1976 and 2000 (Ackerman et al. 1995; FWC, unpublished data) (USFWS, 2001).***Recovery*****Reclassification Criteria:**

Identify minimum flow levels for important springs used by wintering manatees (82 FR 16668).

Protect a network of warm-water refuges as manatee sanctuaries, refuges, or safe havens (82 FR 16668).

Identify foraging sites associated with the network of warm-water sites for protection (82 FR 16668).

Identify for protection a network of migratory corridors, feeding areas, and calving and nursing areas (82 FR 16668).

Address harassment at wintering and other sites to achieve compliance with the Marine Mammal Protection Act (MMPA) and the Endangered Species Act and as a conservation benefit to the species (82 FR 16668).

Protect important manatee habitats (82 FR 16668).

Reduce or remove unauthorized take (82 FR 16668).

Create and enforce manatee safe havens and/or Federal manatee refuges (82 FR 16668).

Retrofit one half of all water control structures with devices to prevent manatee mortality (82 FR 16668).

Draft guidelines to reduce or remove threats of injury or mortality from fishery entanglements and entrapment in storm water pipes and structures (82 FR 16668).

**Delisting Criteria:**

Protect Habitat (USFWS, 2007)

Protect from 'take' (USFWS, 2007)

Resolve the inadequacy of existing regulatory mechanisms (USFWS, 2007)

Protect from human related mortality (USFWS, 2007)

**Recovery Actions:**

- In order to ensure the long-term recovery needs of the manatee and provide adequate assurance of population stability (i.e., achieving the demographic criteria), threats to the manatee's habitat or range must be reduced or removed. This can be accomplished through Federal, State or local regulations to establish and maintain minimum spring flows and protect the following areas of important manatee habitat: a. Minimum flow levels to support manatees at the Crystal River Spring Complex, Homosassa Springs, Blue Springs, Warm Mineral Spring, and other spring systems as appropriate, in terms of quality (including thermal) and quantity have been adopted by regulation and are being maintained. (Task 3.2.4.3) 9 b. A network of level 1 (Primary), 2(Secondary) and 3 (Tertiary) warm-water refuge sites have been protected as either manatee sanctuaries, refuges or safe havens. (Task 1.2.3, 1.3, 3.2.2, 3.2.3, 3.2.4, 3.3.1) c. Adequate feeding habitat sites (extent, quantity and quality) associated with the network warm-water refuge sites identified by the HWG and are protected. (Task 3.1(3), 3.3.8). d. The network of migratory corridors, feeding

- areas, calving and nursing areas identified by the HWG are protected as manatee sanctuaries, refuges or safe havens. (Task 1.3, 3.3.1) (USFWS, 2007).
- “Take” in the form of harassment, is currently occurring at some of the winter refuge sites and other locations. This “take” is presently not authorized under the MMPA or ESA. However, there are no data at this time to indicate that this issue is limiting the recovery of the Florida manatee. The actions in this plan that address harassment are recommended in order to achieve compliance with the MMPA and ESA and as a conservation benefit to the species. Statutory mechanisms outlined in Factor D to protect and enact protection regulations for important manatee habitats identified in Factor A and enact regulations to address unauthorized “take” identified in Factor E, will also assist to reduce or remove these threats. Recovery actions and their subtasks specifically addressing this issue are 1.1, 1.11, 4.4 and those tasks identified in Factors A, D and E (USFWS, 2007).
  - The current legal framework outlined below allows Federal and State government agencies to take both broad scale and highly protective action for the conservation of the manatee and its habitat. The FWS believes these regulatory mechanisms are adequate for recovery. However, additional specific actions under these laws such as those listed pursuant to Factor A and E must be accomplished (as well as meeting the demographic criteria) before the FWS will consider this species for removal from the List of Endangered and Threatened Wildlife. Factor A (a) Establish Minimum Flows (Task 3.2.4.3) STATE Florida Water Resources Act of 1972, Chapter 373, F.S. (specifically Minimum Flows and Levels, Sect. 370.42, F.S. and Establishment and Implementation of Minimum Flows and Levels, Sect. 370.421, F.S.) Factor A (b)(c) and (d) Protect Important Manatee Habitats (Task 1.2, 1.3.1, 1.3.2, 1.4, 3.2.2, 3.2.3, 3.2.4, 3.3.1, 3.3.8) FEDERAL Marine Mammal Protection Act; Clean Water Act, Sect. 401, 402 and 404; Rivers and Harbors Act, Sect. 10; National Environmental Policy Act; and Coastal Zone Management Act; STATE Florida Manatee Sanctuary Act, Sect. 370.12(2), F.S.; Florida Water Resources Act of 1972, Chapter 373, F.S.; Florida Air and Water Pollution Control Act, Chapter 403, F.S.; State Lands, Chapter 253, F.S.; and State Parks and Preserves, Chapter 258, F.S.; and LOCAL Florida Manatee Sanctuary Act, Sect. 370.12(o), F.S. which allows local governments to regulate by ordinance, motorboat speed and operations to protect manatees. Factor E (a)(b)(c) Reduce or Remove Unauthorized “take” (Task 1.1, 1.2, 1.3.1, 1.3.2, 1.4, 1.6, 1.7, 3.3.1) FEDERAL Marine Mammal Protection Act; and STATE Florida Manatee Sanctuary Act, 370.12(2), F.S. (USFWS, 2007).
  - The most predictable and controllable threat to manatee recovery remains human-related mortality. In order to ensure the long-term recovery needs of the manatee and provide adequate assurance of population stability (i.e., achieving the demographic criteria), natural and manmade threats to manatees need to be reduced or removed. This can be accomplished through establishing the following Federal, State or local regulations, tasks and guidelines to reduce or remove human caused “take” of manatees: a. State, Federal and local government manatee conservation measures (such as, but not limited to speed zones, refuges, sanctuaries, safe havens, enforcement, education programs, county MPPs etc.) have been adopted and implemented to reduce or remove unauthorized watercraft-related “take” in the following Florida counties: Duval (including portions of Clay and St. Johns in the St. Johns River), Volusia, Brevard, Indian River, Martin, Palm Beach, Broward, Dade and Monroe on the Florida Atlantic Coast; Citrus, Pinellas, Hillsborough, Manatee, Sarasota, Charlotte, Lee and Collier on the Florida Gulf Coast; and Glades County on the Okeechobee Waterway. These measures are not only necessary to achieve recovery, but may ultimately help to comply with the MMPA. (Task 1.3, 1.4, 1.5, 3.3.1). Stable or positive population benchmarks as outlined in the demographic criteria provide measurable

- population parameters that will assist in measuring the stabilization, reduction, or minimization of watercraft related “take.” Two other indices (weight of evidence) will assist in measuring success include: (1) watercraft-related deaths as a proportion of the total known mortality; and (2) watercraft-related deaths as a proportion of a corrected estimated population. These and other indices should be monitored. b. All water control structures and navigational locks listed as needing devices to prevent mortality have been retrofitted. (Task 1.6) c. Guidelines have been established and are being implemented to reduce or remove threats of injury or mortality from fishery entanglements and entrapment in storm water pipes and structures. (Task 1.7, 1.6.3) (USFWS, 2007).
- For Florida manatees: Establish minimum flow requirements to guarantee sufficient manatee winter habitat at key natural springs and restore access to springs in the St. Johns River watershed, Homosassa Springs, and other sites. Develop and implement a comprehensive management strategy to address manatee protection in the Ten Thousand Islands National Wildlife Refuge and Everglades National Park. Assess forage availability near wintering sites to determine the potential carrying capacity of these sites, assess the long-term effect of habitat modification on the population, and manage accordingly. Ensure that contingency plans and cooperative agreements with key industry and government partners are developed and utilized to mitigate the adverse effects of anticipated changes in artificial sources of warm water (USFWS, 2007).
  - For Florida manatees: Propose regulations (in consultation with the Corps of Engineers and other Federal agencies) pursuant to section 112(a) of the MMPA to address direct, indirect, and cumulative threats from future development and resolve conflicts with the current consultation process under section 7 of the ESA. Ensure that the State of Florida’s manatee management plan will be sufficient to control watercraft injury and mortality. Ensure losses of power plant warm water effluents are adequately mitigated through coordination/consultation with EPA in association with Clean Water Act section 316 (b) requirements for once-through cooling systems (USFWS, 2007).
  - For Florida manatees: Update the Florida Manatee Recovery Plan and, at a minimum, revise the demographic recovery criteria; as written, these criteria are considered inadequate (Section IIB2a). Consider restructuring the Florida Manatee Recovery Team. Continue to monitor the status of Florida manatees through surveys, photo identification and genetics research. New research on population genetics in Florida and in Puerto Rico is underway, and we will investigate whether manatees in each of these areas could be considered as distinct populations when that information becomes available. Use the results of new research to review and update the scientific information used in the manatee Core Biological Model, especially to gain a better understanding of manatee population dynamics in southwest Florida. Use updated demographic information to assess the effects of improved State and Federal management efforts since 2000 (USFWS, 2007).
  - For Florida manatees: Expedite the next Federal status review and conduct it in 2009-2010, when updated adult survival rates will be available. If the above issues are satisfactorily addressed, it may be most appropriate to remove the manatee from the list of threatened and endangered species at the Federal level and provide protection under the MMPA only (USFWS, 2007).
  - For Antillean Manatee in Puerto Rico and the U.S. Virgin Islands: In Puerto Rico, further discussions about State safe havens (manatee refuges and sanctuaries) and/or Federal manatee protection areas (including speed restricted and exclusion areas, as defined in 50 CFR 17 Subpart J) should be held between the Service and the Puerto Rico Department of Environment and Natural Resources regarding the following municipalities:

- Fajardo, Ceiba, Naguabo, Vieques, Arroyo, Patillas, Guayama, Lajas, and Cabo Rojo. More specifically, refuges should be established in Jobos Bay in Guayama; in Pelican Cove, Ensenada Honda and the Cape Hart Sewage Plant in the RRNS area; in that area west of Mosquito Pier to Punta Arenas in Vieques; in La Parguera and Bahía Montalva in Lajas; and in Laguna Rincón, Bahía Boquerón and Puerto Real in Cabo Rojo. Other areas may be included as information on distribution and use is further refined. The loss of habitat, including the loss of freshwater sources and seagrasses due to a variety of causes, should be monitored and prevented (USFWS, 2007).
- For Antillean Manatee in Puerto Rico and the U.S. Virgin Islands: If established, manatee protection areas should be adequately enforced to minimize unauthorized watercraft-related “takings.” An outreach program should be developed to reach younger generations who take “boat” training courses. Manatee conservation efforts should be properly marketed to target boating communities, developers, and non-users with the message of “losing one manatee is one too many.” Marina and other boating access development projects should be reviewed to address potential increases in the likelihood of manatee-boat collisions resulting from these projects. Construction of marinas and other boat access should be assessed to identify, quantify, avoid, and minimize threats to manatees. 6. Guidelines should be drafted to further reduce or remove threats of injury or mortality from fishery entanglements (USFWS, 2007).
  - For Antillean Manatee in Puerto Rico and the U.S. Virgin Islands: Update the Recovery Plan and develop delisting criteria. Continue to monitor the status of manatees in Puerto Rico through improved (statistically-sound) survey methodology and genetics research. Continue to monitor and report on sources of manatee mortality through the carcass salvage effort, and statistically evaluate the fractions of mortality due to the various causes. Initiate demographic studies to better understand adult survival, juvenile recruitment and population growth. Initiate new research to investigate the importance of freshwater resources in Puerto Rico to the manatee population. Assess whether manatees in Puerto Rico can be considered as a DPS following advancements in genetics research (USFWS, 2007).

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## SPECIES ACCOUNT: *Enhydra lutris nereis* (Southern sea otter)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Threatened; 1/14/1977; Pacific Southwest Region (R8) (42 FR 2965).

### **Physical Description**

The southern sea otter is the largest member of the family Mustelidae and the smallest species of marine mammal in North America. Southern sea otters have thick, reddish, dark brown or black pelage (fur), with the head whitish in older individuals, especially males. The pelage consists of sparse guard hairs and dense underfur; underfur density may reach 100,000 or more follicles per square centimeter (650,000 per square inch). There is little subcutaneous fat and no layer of blubber for energy storage and thermo-insulation as in pinnipeds (seals) and cetaceans (whales). Insulation from cold sea water is provided entirely by air trapped in the fur. Adult southern sea otters average about 30 kilograms (kg) (65 pounds [lbs.]) for males and 20 kg (45 lbs.) for females; average lengths are about 135 centimeters (cm) (4.5 feet [ft.]) and 125 cm (4 ft.) for males and females, respectively. Forepaws are padded, relatively small and round, have claws, and are used in feeding and grooming. Hind limbs are posteriorly oriented and flipperlike for swimming, with the hind feet flattened and webbed. Their tail is uniform in thickness from base to tip (to about 35 cm [1.15 ft.]) and about a quarter of total of body length. Their lung capacity is 2.5 times that of land mammals of the same size. They have good eyesight and use their whiskers to sense vibrations in the water (NatureServe 2015; USFWS 2003; USFWS 2015).

### **Taxonomy**

There are three recognized subspecies of sea otters: the California or southern sea otter (*Enhydra lutris nereis*), the Alaskan or northern sea otter (*Enhydra lutris kenyoni*), and the Russian or Asian sea otter (*Enhydra lutris lutris*) (USFWS 2003). *Enhydra lutris nereis* differs from subspecies *E. l. lutris* by having a shorter overall skull length, longer nasal length, narrower skull, lack of frontal notch, shorter mandibles, and smaller teeth; it differs from subspecies *E. l. kenyoni* in shorter overall skull length, longer nasal length, narrower skull, greater palatal width, smaller teeth, and shorter mandibles (NatureServe 2015).

### **Historical Range**

The historic range of the southern sea otter appears to have been as far north as Newport, Oregon and as far south as Punta Abreojos, Baja California, Mexico (USFWS 2003).

### **Current Range**

The current range of the southern sea otter population in the United States is approximately from Pigeon Point, San Mateo County, California in the north, to Gaviota State Beach, Santa Barbara County, California in the south, spanning roughly 500 kilometers (km) (310 miles [mi.]) of coastline. An additional translocated population occurs at San Nicolas Island, in the Channel Islands off of Ventura County, California (USFWS 2003; USFWS 2015). There is some evidence that a third population may be slowly establishing itself in Baja California, Mexico, from otters emigrating from southern California (NatureServe 2015; Schramm et. al. 2014).

### **Distinct Population Segments Defined**

No

**Critical Habitat Designated**

No;

***Life History*****Feeding Narrative**

Adult: Southern sea otters forage in both rocky and soft-sediment communities in water depths generally 25 m (82 ft.) or less. Southern sea otters occasionally make dives of up to 100 m (328 ft.), but the vast majority of feeding dives (about 95 percent) occur in shallower waters. They feed during the day, and possibly also at night, on invertebrates often associated with kelp forests, such as abalone, rock crabs, sea urchins, kelp crabs, clams, turban snails, mussels, octopus, barnacles, scallops, sea stars, and chitons. They have strong canines and molars to tear and crush their food. They are known to use rocks or other hard objects as tools to break the exoskeletons of their invertebrate prey. Southern sea otters also feed on fish. Southern sea otters need to maintain a high level of internal heat production to compensate for their lack of blubber. Consequently, their energetic requirements are high, and they consume an amount of food equivalent to 20 to 25 percent of their body mass per day. Competition for food resources includes other invertivores and human commercial fisheries, especially for sea urchins. Southern sea otters are considered to be a keystone predator; their predation on herbivores, such as sea urchins, determines the structure of offshore kelp communities (NatureServe 2015; USFWS 2003; USFWS 2015).

**Reproduction Narrative**

Adult: Southern sea otters are strongly polygynous, with the males mating with more than one female. Males defend contiguous territories from which they exclude other males. Males may move up to 50 to 100 km (30 to 60 mi.) along the coast; females generally stay within an area 8 to 16 km (5 to 10 mi.) long. Females are sexually mature usually in 3 to 5 years. The age of sexual maturity in males is less well known, but appears to be about 5 years. Typical life spans are 12 to 18 years for females and 10 to 15 years for males. Mating and pupping occur throughout the year and are weakly seasonal. The gestation period lasts approximately 6 months, consisting of a phase of 2 to 3 months during which the embryo remains unattached to the uterine wall (delayed implantation) and an implanted phase of 4 months. A peak period of pupping occurs from October to January, with a secondary peak in March and April. In California, most births occur from late February to early April. Females typically give birth to a single pup, with care provided solely by the female for the approximately 6 to 7 months until weaning. Twin births occur rarely; seldom, if ever, do both young survive to weaning (NatureServe 2015; USFWS 2003; USFWS 2015).

**Geographic or Habitat Restraints or Barriers**

Adult: Restraints are the spatial configuration of available habitat. Southern sea otter habitat is typically defined by the 40-m (131-ft.) depth contour, which is the long, narrow strip of coastal shelf characteristic of California. Depending on local bathymetry, most southern sea otters reside within 2 km (1.2 mi.) of shore. Combined with the high degree of spatial structure in southern sea otter populations (due to limited mobility of reproductive females), the total available nearshore area in California (to the 40-m [131-ft.] depth contour) is 7,569 square kilometers (km<sup>2</sup>) (2,922 square miles [sq. mi.]), of which 27 percent is rocky habitat, 51 percent is sandy habitat, and 22 percent is mixed habitat (NatureServe 2015; USFWS 2003).

**Spatial Arrangements of the Population**

Adult: Clumped within the 2-km (1.2-mi.) nearshore band, with kelp beds and abundant shellfish (NatureServe 2015; USFWS 2003).

**Environmental Specificity**

Adult: Narrow/specialist nearshore habitat.

**Tolerance Ranges/Thresholds**

Adult: Low

**Site Fidelity**

Adult: High

**Dependency on Other Individuals or Species for Habitat**

Adult: Nearshore/ kelp forest for food and shelter.

**Habitat Narrative**

Adult: Southern sea otters inhabit a narrow marine nearshore band in waters along the coast, especially shallows with kelp beds and abundant shellfish. In rough weather, they take refuge among kelp beds, or in coves and inlets. The species is restrained by the spatial configuration of available habitat. Southern sea otter habitat is typically defined by the 40-m (131-ft.) depth contour, which is the long, narrow strip of coastal shelf characteristic of California. Depending on local bathymetry, most southern sea otters in California reside within 2 km (1.2 mi.) of shore. Combined with the high degree of spatial structure in southern sea otter populations (due to limited mobility of reproductive females), the total available nearshore habitat in California (to the 40-m [131-ft.] depth contour) is 7,569 km<sup>2</sup> (2,922 sq. mi.). Of this area, 27 percent is rocky habitat, 51 percent is sandy habitat, and 22 percent is mixed habitat (NatureServe 2015; USFWS 2003; USFWS 2015).

***Dispersal/Migration*****Motility/Mobility**

Adult: Moderate; may move into historic range.

**Migratory vs Non-migratory vs Seasonal Movements**

Adult: Nonmigratory

**Dispersal**

Adult: Moderate

**Immigration/Emigration**

Adult: Unlikely due to habitat constraints.

**Dependency on Other Individuals or Species for Dispersal**

Adult: No

**Dispersal/Migration Narrative**

Adult: Southern sea otters occupy only a portion of their historic range. Dispersal into the historic range could occur if contiguous habitat and adequate prey resources occur within the home range/territories of the individuals. Males may move up to 50 to 100 km (30 to 60 mi.) along the coast; females generally stay within an area 8 to 16 km (5 to 10 mi.) long. Most female southern sea otters are more sedentary than the males, with adult females rarely dispersing more than 20 km (12 mi.) within a 1-year period (NatureServe 2015; USFWS 2003; USFWS 2015).

**Additional Life History Information**

Adult: Males may move up to 50 to 100 km (30 to 60 mi.) along the coast; females generally stay within an area 8 to 16 km (5 to 10 mi.) long. Most female southern sea otters are more sedentary than the males, with adult females rarely dispersing more than 20 km (12 mi.) within a 1-year period (USFWS 2003; USFWS 2015).

***Population Information and Trends*****Population Trends:**

Stable, increasing.

**Species Trends:**

Stable, increasing.

**Population Growth Rate:**

Slow

**Number of Populations:**

Two populations: the mainland coastal population and the San Nicolas Island Population. On August 11, 1987, the U.S. Fish and Wildlife Service (USFWS) issued a final rule to establish and manage an experimental population of southern sea otters at San Nicolas Island, California (52 FR 29754). In 2012, USFWS removed the experimental population designation and terminated the translocation program due to failure maintain a management or “no otter” zone in the Southern California Bight and its respective translocation and management zones (77 FR 75266). There is some evidence that a third population may be slowly establishing in Baja California, Mexico, from otters emigrating from southern California (NatureServe 2015; Schramm et al. 2014).

**Population Size:**

2,944 in 2014 (USFWS 2015).

**Minimum Viable Population Size:**

500 individuals (USFWS 2015).

**Adaptability:**

Low

**Additional Population-level Information:**

The historic abundance of sea otters in California is between 16,000 and 20,000 animals. The southern sea otter currently occupies approximately 13 percent of its historic range. Because sea otters in the central portion of the occupied range have reached equilibrium densities

(increasing their susceptibility to natural and anthropogenic stressors and limiting the potential for additional population growth in that area), significant growth of the population as a whole will require range expansion into currently unoccupied habitat (USFWS 2015).

**Population Narrative:**

The estimated historic abundance of sea otters in California was between 16,000 and 20,000 animals. The population has grown from its remnant population of approximately 50 individuals in 1914, to approximately 1,760 animals in 1975 (42 FR 2965). Data on population size have been gathered for more than 50 years. The 3-year running average for 2014 is 2,944 for the mainland population and San Nicolas Island, or 2,881 and 68, respectively. Whereas the trend in abundance for the mainland population over the past 5 years remains essentially flat, the San Nicolas Island population has begun to grow rapidly, averaging approximately 16 percent annually over the past 5 years. The southern sea otter currently occupies approximately 13 percent of its historic range. Because sea otters in the central portion of the occupied range have reached equilibrium densities (increasing their susceptibility to natural and anthropogenic stressors and limiting the potential for additional population growth in that area), significant growth of the population as a whole will require range expansion into currently unoccupied habitat. In 1987, USFWS issued a final rule to establish and manage an experimental population of southern sea otters at San Nicolas Island, California (52 FR 29754). In 2012, USFWS removed the experimental population designation and terminated the translocation program due to failure maintain (keep the otters from moving into) the management or “no otter” zone in the Southern California Bight and its respective translocation and management zones (77 FR 75266). There is some evidence that a third population may be slowly establishing in Baja California, Mexico, from otters emigrating from southern California (NatureServe 2015; Schramm et al. 2014).

**Threats and Stressors**

**Stressor:** Oil spill pollution

**Exposure:** Oil pollution in waters of occupied habitat.

**Response:** Reduced growth; increased illness and injury; increased vulnerability to predation and death.

**Consequence:** Reduction in population numbers, decreased reproductive success.

**Narrative:** Southern sea otters are particularly vulnerable to oil contamination. When sea otters come into contact with oil, it causes their fur to mat, which prevents them from insulating their bodies. Without this natural protection from the frigid water, sea otters can quickly die from hypothermia. The toxicity of oil can also be harmful to sea otters, causing liver and kidney failure and damage to the lungs and eyes (USFWS 2003).

**Stressor:** Non-oil-spill pollution

**Exposure:** Non-oil-spill pollution (chemical or biological) in waters of occupied habitat.

**Response:** Reduced growth; increased illness and injury; increased vulnerability to predation and death.

**Consequence:** Reduction in population numbers, decreased reproductive success.

**Narrative:** Non-oil-spill pollution refers to chemical and biological contaminants. Due to its proximity to coastlines densely occupied by humans, sea otter habitat is potentially subject to degradation resulting from anthropogenic activities and inputs, including contaminants and terrestrial disease-causing pathogens (USFWS 2003).

**Stressor:** Loss of genetic diversity

**Exposure:** Current population of southern sea otters is derived from a single remnant population.

**Response:** Reduced growth; increased illness and injury; increased vulnerability to predation and death.

**Consequence:** Reduction in population numbers, decreased reproductive success.

**Narrative:** Because the current population of southern sea otters is derived from a single remnant population, a number of studies since the time of listing have identified particularly low levels of genetic diversity in the species. Sea otter populations have a very low level of variation at major histocompatibility complex genes (a set of genes that control a major part of the immune system). Low genetic variability has the potential to compromise the ability of southern sea otters to withstand environmental change and increase their susceptibility to land-borne (novel) pathogens. However, inbreeding depression in southern sea otters has not been demonstrated (USFWS 2003).

**Stressor:** Mortality in fishing gear

**Exposure:** Commercial or recreational fishing in occupied habitat.

**Response:** Injury, death.

**Consequence:** Reduction in population numbers, decreased reproductive success.

**Narrative:** Sea otters may become entangled/entrapped and drown in commercial fishing gear that is deployed or abandoned in the nearshore marine environment. A period of decline in the southern sea otter population from 1976 to 1984 was likely due to incidental mortality in set-net fisheries, although gill and trammel nets have since been restricted throughout most of the range of the southern sea otter. The potential exists for sea otters to drown in traps set for crabs, lobsters, and finfish, but only limited documentation of mortalities is available (USFWS 2003).

**Stressor:** Disease

**Exposure:** Exposure to pathogens or parasites in occupied habitat.

**Response:** Reduced growth; increased illness and injury; increased vulnerability to predation and death.

**Consequence:** Reduction in population numbers, decreased reproductive success.

**Narrative:** Infectious disease is an important cause of death for southern sea otters. Increased susceptibility to pathogens and parasites appears to be the consequence of poor body condition resulting from low per-capita prey availability in portions of the range that are at or near local carrying capacity (USFWS 2003).

**Stressor:** Predation

**Exposure:** Exposure to predators with little or no refuge for escape.

**Response:** Injury, death.

**Consequence:** Reduction in population numbers, decreased reproductive success.

**Narrative:** White shark attacks are currently the single most important cause of mortality for southern sea otters. Shark bite mortality has long been an important factor at the northern end of the mainland range, but it has increased dramatically over the past 10 years in the southern portion of the mainland range between Cayucos and Point Conception (USFWS 2003).

**Stressor:** Climate change

**Exposure:** Climate change in habitat range.

**Response:** Reduced growth; increased illness and injury; increased vulnerability to predation and death.

**Consequence:** Reduction in population numbers, decreased reproductive success.

**Narrative:** Coastal zones are particularly vulnerable to climate variability and change through sea-level rise and changes in rainfall-runoff patterns. Climate change may affect southern sea otters by modifying hydrological processes that influence the transport of pathogens and contaminants from land to the nearshore marine environment. It also has the potential to alter (in unknown ways) the frequency of algal blooms in both freshwater and the marine environment. Increasing ocean temperatures may increase the incidence and spread of disease among marine organisms, with potentially negative or positive effects on sea otters, depending on the particular ecological relationships affected. In addition to increasing ocean temperatures, changes in the carbonate chemistry of the oceans due to increasing atmospheric carbon dioxide levels (ocean acidification) may pose a serious threat to marine organisms, particularly calcifying organisms, many of which are important prey for sea otters. Because of the apparent synergistic relationship between food limitation and disease, potential climate-driven declines in food availability may in turn result in increased susceptibility to disease (USFWS 2003).

### **Recovery**

#### **Reclassification Criteria:**

The southern sea otter population should be considered for reclassification as endangered under the Endangered Species Act if the population declines to a level less than or equal to an effective population size of 500 animals. Until better information is available, a multiplier of 3.7 should be used to convert effective population size to actual population size, or 1,850 animals. Therefore, the southern sea otter population should be considered endangered if, based on standard survey counts (i.e., spring surveys), the average population level over a 3-year period is less than or equal to 1,850 animals (USFWS 2003).

The southern sea otter population should be considered threatened under the Endangered Species Act if the average population level over a 3-year period is greater than 1,850 animals, but less than 3,090 animals (USFWS 2003).

#### **Delisting Criteria:**

The southern sea otter population should be considered for delisting under the Endangered Species Act when the average population level over a 3-year period exceeds 3,090 animals (USFWS 2003).

#### **Recovery Actions:**

- Monitor existing and translocated populations (USFWS 2003).
- Implement plans to reduce the probability of an oil spill in the sea otter range and minimize effects of a spill on the otter population, in the event that one occurs (USFWS 2003).
- Monitor the incidental take of sea otters in commercial fisheries, and minimize intentional take of southern sea otters (USFWS 2003).
- Develop and implement a public education and outreach to minimize recreation-related and other harassment, and increase public understanding of the broad range of sea otter community effects and ecosystem services (USFWS 2003).
- Monitor existing and translocated populations (USFWS, 2015).

- Implement plans to reduce the probability of an oil spill in the sea otter range and minimize effects of a spill on the otter population, in the event that one occurs (USFWS, 2015).
- Monitor the incidental take of sea otters in commercial fisheries and minimize intentional take of southern sea otter (USFWS, 2015).
- Develop and implement a public education and outreach program to minimize recreation-related and other harassment (i.e., the Be Sea Otter Savvy program) and 2) to increase public understanding of the broad range of sea otter community effects and ecosystem services) (USFWS, 2015).
- Complete spatially explicit Integrated Population Model to prioritize recovery actions with rigorous demographic sensitivity analyses (USFWS, 2015).
- Determine the cause(s) of the geographic shift/increase in shark-bite mortality (USFWS, 2015).
- Develop and implement a plan to enhance natural range expansion through releases of small numbers of rehabilitated live-stranded sea otter (USFWS, 2015).

***Additional Threshold Information:***

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## SPECIES ACCOUNT: *Enhydra lutris kenyoni* (Northern Sea Otter)

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### *Species Taxonomic and Listing Information*

**Listing Status:** Threatened; 08/09/2005; Alaska Region (R7) (USFWS, 2016)

### **Physical Description**

It is the smallest marine mammal in the world, except for the South American marine otter (*Lontra* (= *Lutra*) *felina*) (Reidman and Estes 1990). Adult males average 130 centimeters (cm) (4.3 feet (ft)) in length and 30 kilograms (kg) (66 pounds (lb)) in weight; adult females average 120 cm (3.9 ft) in length and 20 kg (44 lb) in weight (Kenyon 1969). The northern sea otter in Russian waters (*E. l. lutris*) is the largest of the three subspecies, characterized as having a wide skull with short nasal bones (Wilson et al. 1991). The southern sea otter (*E. l. nereis*) is smaller and has a narrower skull with a long rostrum and small teeth. The northern sea otter in Alaska (*E. l. kenyoni*) is intermediate in size and has a longer mandible than either of the other two subspecies. Sea otters lack the blubber layer found in most marine mammals and depend entirely upon their fur for insulation (Riedman and Estes 1990). Their pelage consists of a sparse outer layer of guard hairs and an underfur that is the densest mammalian fur in the world, averaging more than 100,000 hairs per square centimeter (645,000 hairs per square inch) (Kenyon 1969). As compared to pinnipeds (seals and sea lions) that have a distinct molting season, sea otters molt gradually throughout the year (Kenyon 1969) (USFWS, 2005). Adult males average 130 cm (4.3 ft) in length and 30 kg (66 lb) in weight; adult females average 120 cm (3.9 ft) in length and 20 kg (44 lb) in weight (Kenyon 1969). Sea otters lack blubber and depend entirely upon their fur for insulation (Riedman and Estes 1990). They molt gradually throughout the year (Kenyon 1969).

### **Taxonomy**

The sea otter is a mammal in the family Mustelidae and it is the only species in the genus *Enhydra*. Three subspecies are recognized: 1) the Asian northern sea otter (*E. l. lutris*), which occurs west of the Aleutian Islands; 2) the southern sea otter (*E. l. nereis*), which occurs off the coast of California and Oregon; and 3) the Alaskan northern sea otter (*E. l. kenyoni*), which occurs from the west end of the Aleutian Islands to the coast of the State of Washington (Wilson et al. 1991).

### **Current Range**

The northern sea otter has a range that extends from the Aleutian Islands in southwestern Alaska to the coast of the State of Washington. Three stocks of sea otters are recognized in Alaska: southwest, southcentral and southeast (Figure A-20). The southwest Alaska stock ranges from Attu Island at the western end of the Near Islands in the Aleutians, east to Kamishak Bay on the western side of lower Cook Inlet, and includes waters adjacent to the Aleutian Islands, the Alaska Peninsula, the Kodiak archipelago, and the Barren Islands (USFWS 2005) and is currently estimated to contain about 55,000 animals (USFWS 2014b). Within the range of northern sea otters (*E. l. kenyoni*), there may be physical barriers to movement across the upper and the lower portions of Cook Inlet, and there are morphological and some genetic differences between sea otters that correspond to the southwest and south-central Alaska stocks (USFWS 2005). Genetic analyses show some similarities between sea otters in the Commander Islands, Russia, and Alaska (Cronin et al. 1996), which indicates that movements between these areas has occurred, at least over evolutionary time scales. All existing sea otter populations have

experienced at least one genetic bottleneck caused by the commercial fur harvests from 1741 - 1911. As part of efforts to re-establish sea otters in portions of their historical range, otters from Amchitka Island (part of the Aleutian Islands) and Prince William Sound were translocated to other areas outside the range of what we now recognize as the southwest Alaska distinct population segment, but within the range of *E. l. kenyoni* (Jameson 2002).

**Distinct Population Segments Defined**

Yes

**Critical Habitat Designated**

Yes; 10/8/2009.

**Legal Description**

On October 8, 2009, the U.S. Fish and Wildlife Service (Service), designated critical habitat for the southwest Alaska Distinct Population Segment (DPS) of the northern sea otter (*Enhydra lutris kenyoni*) under the Endangered Species Act of 1973, as amended (Act). In total, approximately 15,164 square kilometers (km<sup>2</sup>) (5,855 square miles (mi<sup>2</sup>)) fall within the boundaries of the critical habitat designation.

**Critical Habitat Designation**

Five units are designated as critical habitat for the southwest Alaska DPS of the northern sea otter. The 5 units are: (1) Western Aleutian Unit; (2) Eastern Aleutian Unit; (3) South Alaska Peninsula Unit; (4) Bristol Bay Unit; and (5) Kodiak, Kamishak, Alaska Peninsula Unit.

Unit 1: Western Aleutian Unit: Unit 1 consists of at least 1,551 km<sup>2</sup> (599 mi<sup>2</sup>), collectively, of the nearshore marine waters ranging from the mean high tide line to the 20-m (65.6-ft) depth contour as well as waters occurring within 100 m (328.1 ft) of the mean high tide line. Hydrographic survey data in the vicinity of Atka and Admi islands is insufficient to delineate the 20-m (65.6-ft) depth contour, so our area calculation may slightly underestimate the total area of this unit. This unit ranges from Attu Island in the west to Kagamil Island in the east, was occupied at the time of listing, and is currently occupied. The majority (80.2 percent) of the lands bordering this unit are federally owned within the Alaska Maritime National Wildlife Refuge. In addition, all critical habitat within this unit is located within State of Alaska waters (defined as those within 3 mi (4.82 km) of mean high tide). The Western Aleutian Unit contains all of the PCEs essential for the conservation of the southwest Alaska DPS of the northern sea otter. Special management considerations and protections may be needed to minimize the risk of oil and other hazardous material spills from commercial shipping within the region and along the northern great circle route.

Unit 2: Eastern Aleutian Unit: Unit 2 consists of an estimated 832 km<sup>2</sup> (321 mi<sup>2</sup>), collectively, of the nearshore marine waters ranging from the mean high tide line to the 20-m (65.6-ft) depth contour as well as waters occurring within 100 m (328.1 ft) of the mean high tide line. This unit ranges from Samalga Island in the west to Ugamak Island in the east, was occupied at the time of listing, and is currently occupied. The majority (89.8 percent) of the lands bordering this unit are owned or selected by (but not yet conveyed to) Alaska Natives. In addition, all the critical habitat within this unit is located within State of Alaska waters. The Eastern Aleutian Unit contains all of the PCEs essential for the conservation of the southwest Alaska DPS of the northern sea otter. Special management considerations and protections may be needed to minimize the risk of oil

and other hazardous material spills from commercial shipping within the region and along the northern great circle route.

**Unit 3: South Alaska Peninsula Unit:** Unit 3 consists of an estimated 4,946 km<sup>2</sup> (1,909 mi<sup>2</sup>), collectively, of the nearshore marine waters ranging from the mean high tide line to the 20-m (65.6-ft) depth contour as well as waters occurring within 100 m (328.1 ft) of the mean high tide line. Available hydrographic survey data for this unit have considerably lower spatial resolution than the other units. This unit ranges from Unimak Island in the west to Castle Cape in the east, was occupied at the time of listing, and is currently occupied. The majority (78.5 percent) of the lands bordering this unit are owned or selected by (but not yet conveyed to) Alaska Natives. The vast majority (85 percent) of the critical habitat within this unit is located within State of Alaska waters. The South Alaska Peninsula Unit contains all of the PCEs essential for the conservation of the southwest Alaska DPS of the northern sea otter. Special management considerations and protections may be needed to minimize the risk of oil and other hazardous material spills from commercial shipping within this region and along the northern great circle route.

**Unit 4: Bristol Bay Unit:** Unit 4 consists of an estimated 1,080 km<sup>2</sup> (417 mi<sup>2</sup>) of the nearshore marine environment. This unit is further subdivided into 3 subunits: (4a) Amak Island; (4b) Izembek Lagoon; and (4c) Port Moller/Herendeen Bay. With the exception of Amak Island, the coastline contained within this unit is relatively simple and lacks kelp forests. For most of this unit, the 20-m (65.6-ft) depth contour used as a criterion for critical habitat in other units does not identify features that provide protection from marine predators, and is applicable only to the Amak Island subunit. Other criteria are used to identify the Izembek Lagoon and Port Moller/Herendeen Bay subunits, as described below. All three subunits within the Bristol Bay unit were occupied at the time of listing, and are currently occupied. Additional information about each subunit is included below.

**Subunit 4a: Amak Island Subunit** Subunit 4a consists of an estimated 31 km<sup>2</sup> (12 mi<sup>2</sup>), collectively, of the nearshore marine waters ranging from the mean high tide line to the 20-m (65.6-ft) depth contour as well as waters occurring within 100 m (328.1 ft) of the mean high tide line. This subunit surrounds Amak Island in Bristol Bay, was occupied at the time of listing, and is currently occupied. Large groups of sea otters have been observed within the kelp forests within this subunit (USFWS unpublished information). All of the lands bordering this subunit are federally owned within the Alaska Maritime National Wildlife Refuge. Most (77 percent) of the critical habitat within this subunit is located within State of Alaska waters, a small portion of which (1.2 km<sup>2</sup>, 0.46 mi<sup>2</sup>) is also located within the boundaries of the Izembek State Game Refuge. The Amak Island Subunit contains all of the PCEs essential for the conservation of the southwest Alaska DPS of the northern sea otter. Special management considerations and protections may be needed to minimize the risk of oil and other hazardous material spills from commercial shipping within Bristol Bay. In addition, offshore oil and gas development are under consideration in the Lease Sale Area 92 in the North Aleutian Basin region immediately offshore from this subunit. An environmental impact statement is in preparation, and will be completed prior to the lease sale. Additional management considerations and protections may be needed to minimize the risk of crude-oil spills associated with oil and gas development and production that may impact this subunit.

**Subunit 4b: Izembek Lagoon Subunit** Subunit 4b consists of an estimated 337 km<sup>2</sup> (130 mi<sup>2</sup>) of the nearshore marine environment within the Izembek Lagoon and Moffett Lagoon systems. Sea otters are known to frequent the lagoon system and regularly haul out on the islands and sandbars that form the northern boundary of these systems, such as Glen, Operl, and Neumann Islands (USFWS unpublished information). Large numbers of otters have also been observed hauling out along

the edges of the sea ice within the lagoon in winter (USFWS unpublished information). This subunit was occupied at the time of listing, and is currently occupied. The majority (89.4 percent) of the lands bordering this subunit are federally owned within the Izembek National Wildlife Refuge. The critical habitat within this subunit is located within State of Alaska waters, most of which (99 percent) is also within the boundaries of the Izembek State Game Refuge. The Izembek Lagoon Subunit contains some of the PCEs (1, 2 and 4) essential for the conservation of the southwest Alaska DPS of the northern sea otter. Special management considerations and protections may be needed to minimize the risk of oil and other hazardous material spills from commercial shipping within Bristol Bay. In addition, offshore oil and gas development are under consideration in the Lease Sale Area 92 in the North Aleutian Basin region immediately offshore from this subunit. Additional management considerations and protections may be needed to minimize the risk of crude-oil spills associated with oil and gas development and production that may impact this subunit.

Subunit 4c: Port Moller/Herendeen Bay Subunit Subunit 4c consists of an estimated 712 km<sup>2</sup> (275 mi<sup>2</sup>) of the nearshore marine environment within the Port Moller and Herendeen Bay systems. This subunit was occupied at the time of listing, and is currently occupied. Aerial surveys conducted in 2000 and 2004, as well as additional reported observations, indicate that these areas may contain several thousand sea otters at any given time (Burn and Doroff 2005, p. 277; USFWS unpublished information). The seaward boundary of this subunit extends from Point Edward on the Alaska Peninsula to the western tip of Walrus Island, and from Wolf Point on the eastern tip of Walrus Island to Entrance Point on the Alaska Peninsula. The majority (66.1 percent) of the lands bordering to this subunit are owned or selected by (but not yet conveyed to) the State of Alaska. Most (94 percent) of the critical habitat within this subunit is located within State of Alaska waters, with a portion (140.8 km<sup>2</sup> (54.4 mi<sup>2</sup>)) located within the boundaries of the Port Moller State Critical Habitat Area. The Port Moller/Herendeen Subunit contains some of the PCEs (1, 2, and 4) essential for the conservation of the southwest Alaska DPS of the northern sea otter. Special management considerations and protections may be needed to minimize the risk of oil and other hazardous-material spills from commercial shipping within Bristol Bay. In addition, offshore oil and gas development are under consideration in the Lease Sale Area 92 in the North Aleutian Basin region immediately offshore from this subunit. Additional management considerations and protections may be needed to minimize the risk of crude-oil spills associated with oil and gas development and production that may impact this subunit.

Unit 5: Kodiak, Kamishak, Alaska Peninsula Unit: Unit 5 consists of an estimated 6,755 km<sup>2</sup> (2,607 mi<sup>2</sup>), collectively, of the nearshore marine environment ranging from the mean high tide line to the 20- m (65.6-ft) depth contour as well as waters occurring within 100 m (328.1 ft) of the mean high tide line. Available hydrographic survey data for parts of this unit have considerably lower spatial resolution than the other units. This unit ranges from Castle Cape in the west to Tuxedni Bay in the east, and includes the Kodiak archipelago. This unit was occupied at the time of listing, and is currently occupied. Slightly more than half (52.4 percent) of the lands bordering this unit are either owned or selected by (but not yet conveyed to) Alaska Natives. The majority (89 percent) of the critical habitat within this unit is located within State of Alaska waters, and a small portion (41.0 km<sup>2</sup>, 15.8 mi<sup>2</sup>) is also located within the boundaries of the Tugidak Island State Critical Habitat Area. The Kodiak, Kamishak, Alaska Peninsula Unit contains all the PCEs essential for the conservation of the southwest Alaska DPS of the northern sea otter. Special management considerations and protections may be needed to minimize the risk of oil and other hazardous material spills from commercial shipping within this region.

**Primary Constituent Elements/Physical or Biological Features**

Critical habitat units are in Alaska. The primary constituent elements of critical habitat for the southwest Alaska distinct population segment (DPS) of the northern sea otter are:

- (i) Shallow, rocky areas where marine predators are less likely to forage, which are in waters less than 2 m (6.6 ft) in depth;
- (ii) Nearshore waters within 100 m (328.1 ft) from the mean high tide line;
- (iii) Kelp forests, which occur in waters less than 20 m (65.6 ft) in depth; and
- (iv) Prey resources within the areas identified in paragraphs (2)(i), (2)(ii), and (2)(iii) of this entry that are present in sufficient quantity and quality to support the energetic requirements of the species.

**Special Management Considerations or Protections**

Critical habitat does not include manmade structures (including, but not limited to, docks, seawalls, pipelines, or other structures) and the land on which they are located existing within the boundaries on the effective date of this rule.

Potential activities that could harm the identified physical and biological features include, but are not limited to, dredging or filling associated with construction of airports, seaports, and harbors; commercial shipping; and oil and gas development and production. Because of the vulnerabilities to pollution sources, these features may require special management or protection through such measures as placing conditions on Federal permits or authorizations to stimulate special operational restraints, mitigative measures, or technological changes.

***Life History*****Feeding Narrative**

Adult: Feeds on benthic invertebrates

**Reproduction Narrative**

Adult: There is variation in age of first reproduction, but generally, male sea otters appear to reach sexual maturity at 5–6 years of age and females reach sexual maturity at 3–4 years (Garshelis et al. 1984; von Biela et al. 2007). The interval between pups is typically one year. The presence of pups and fetuses at different stages of development throughout the year suggests that reproduction occurs at all times of the year. Most areas that have been studied show evidence of one or more seasonal peaks in pupping (Rotterman and Simon-Jackson 1988). Similar to other mustelids, sea otters can have delayed implantation of the blastocyst (developing embryo) (Sinha et al. 1966). As a result, pregnancy can have two phases: from fertilization to implantation, and from implantation to birth (Rotterman and Simon-Jackson 1988). The average time between copulation and birth is 6–7 months. Female sea otters typically will not mate while accompanied by a pup (Lensink 1962; Kenyon 1969; Garshelis et al. 1984). Estimating the rate of recruitment of sea otters into a population is difficult primarily because of asynchronous pupping and an inability to reliably distinguish males from females and juveniles from adults externally. For long-lived species, we expect that survivorship of offspring is related to maternal age and experience, and that recruitment rate is more sensitive than

survival rate to environmental fluctuations (Eberhardt 1977). The maximum life span of a wild sea otter is believed to be 23 years (Nowak 1999).

### **Habitat Narrative**

Adult: Sea otters generally occur in shallow water areas near the shoreline. They are most commonly observed within the 40 m (131 ft) depth contour (USFWS 2008a), although they can be found in waters up to 100 m (328 ft) deep. Most foraging dives take place in waters less than 30 m (98 ft) deep (Bodkin et al. 2004). As water depth is generally correlated with distance to shore, sea otters typically inhabit waters within 1-2 km (0.62–1.24 miles) of shore (Riedman and Estes 1990). Much of the marine habitat of the sea otter in southwest Alaska is characterized by a rocky substrate. In these areas, sea otters typically are concentrated between the shoreline and the outer limit of the kelp canopy (Riedman and Estes 1990), but they also occur further seaward. Sea otters also inhabit marine environments that have soft sediment substrates, such as areas in Bristol Bay and the Kodiak archipelago. As communities of benthic invertebrates differ between rocky and soft sediment substrate areas, so do sea otter diets.

### ***Dispersal/Migration***

#### **Dispersal/Migration Narrative**

Adult: Sea otters in Alaska are non-migratory and generally do not disperse over long distances (USFWS 2008a). They usually remain within a few kilometers of their established feeding grounds (Kenyon, 1981); however they are capable of long distance travel. Translocated populations are known to shift and expand their distribution in favorable habitats, sometimes traversing distances up to 350 km (217 mi) over a relatively short period (Ralls et al. 1992; Jameson 2002). Juvenile males (1–2 years of age) are known to disperse up to 120 km (75 mi) from their natal (birth) area; young females traveled up to 38 km (23.6 mi) (Garshelis and Garshelis 1984; Monnett and Rotterman 1988; Riedman and Estes 1990). Routine movements between feeding and resting areas as large as 35 - 60 miles (57 - 97 km) have also been observed by VanBlaricom et al. (2001). Once a population has become established and has reached equilibrium density within the habitat, the home ranges of sea otters are relatively small. Home range and movement patterns vary depending on the gender and breeding status. In the Aleutian Islands, breeding males remain for all or part of the year within the bounds of their breeding territory, which constitutes a length of coastline anywhere from 100 m (328 ft) to approximately 1 km (0.62 miles). Sexually mature females have home ranges of approximately 8–16 km (5–10 miles), which may include one or more male territories. Male sea otters that do not hold territories may move greater distances between resting and foraging areas than territorial males (Lensink 1962; Kenyon 1969; Riedman and Estes 1990; Estes and Tinker 1996). Typical daily movement distances may exceed 3 km at rates of speed up to 5.5 km per hour (Garshelis and Garshelis 1984). Sea otter movements are also influenced by local climatic conditions such as storm events, prevailing winds, and in some areas, tidal states. Sea otters tend to move to protected or sheltered waters (bays, inlets, or lees) during storm events or high winds. In calm weather conditions, sea otters may be encountered further from shore (Lensink 1962; Kenyon 1969). In the Commander Islands, Russia, weather, season, time of day, and human disturbance have been cited as factors that induce sea otter movement (Barabash-Nikiforov 1947; Barabash-Nikiforov et al. 1968).

### ***Population Information and Trends***

**Population Trends:**

Decreasing

**Population Narrative:**

Historically, sea otters occurred throughout the coastal waters of the North Pacific Ocean from the northern Japanese archipelago around the North Pacific Rim to central Baja California. Between 1741 and 1911, sea otters were hunted to the brink of extinction by Russian and American fur hunters. Prior to commercial exploitation, the worldwide population of sea otters was estimated at 150,000 - 300,000 animals (Kenyon 1969; Johnson 1982). Sea otters were protected from further commercial harvests under the International Fur Seal Treaty of 1911. At that time, only 13 small remnant populations were believed to have persisted. The total worldwide population may have been only 1,000 - 2,000 animals. Two of these remnant populations (Queen Charlotte Island and San Benito islands) declined to extinction (Kenyon 1969; Estes 1980). The remaining 11 populations began to grow in number, and expanded to recolonize much of the former range. Six of these remnant populations (Rat Islands, Delarof Islands, False Pass, Sandman Reefs, Shumagin Islands, and Kodiak Island) were located within the bounds of the southwest Alaska DPS. Because of the remote, pristine nature of southwest Alaska, these remnant populations grew rapidly during the first 50 years following protection from further commercial hunting. The population in southwest Alaska had grown in numbers and re-colonized much of the former range by the mid- to late-1980s. At that time, numbers were believed to be around 92,800 - 126,900 in southwest Alaska. Aleutian Islands From the mid-1960s to the mid-1980s, otters expanded their range, and presumably their numbers as well, until they had recolonized all the major island groups in the Aleutians. Although the maximum size reached by the sea otter population is unknown, a habitat-based computer model estimates that the population in the late 1980s may have numbered approximately 74,000 individuals in the Aleutians (Burn et al. 2003). But in a 1992 aerial survey of the entire Aleutian archipelago, only 8,048 otters were counted (Evans et al. 1997); approximately 19% fewer than the total reported for a 1965 survey Kenyon (1969). In April 2000, 2,442 sea otters were counted; a 70% decline from the count eight years previous (Doroff et al. 2003). Along the more than 5,000 km (3,107 miles) of shoreline surveyed, sea otter density was at a uniformly low level, which clearly indicated that sea otter abundance had declined throughout the archipelago. Doroff et al. (2003) calculated that the decline occurred at an average rate of 17.5% per year throughout the Aleutians. Alaska Peninsula Remnant colonies along the Alaska Peninsula expanded through the 1950s and early 1960s, (Kenyon 1969). Schneider (1976) estimated 17,000 sea otters on the north side of the Alaska Peninsula in 1976 (Burn and Doroff 2005), which he believed to have been within the carrying capacity for that area. In 1986, an estimated 6,474–9,215 sea otters occupied this area (Burn and Doroff 2005). By May 2000, estimates had dropped 27–49% from 1986 (Burn and Doroff 2005). Declines were also occurring along the south side of the Alaska Peninsula between the mid-1960s and early 2000s (Kenyon 1969; Brueggeman et al. 1988; DeGange et al. 1995). Rates of decline as high as 93% were documented in some areas (Burn and Doroff 2005). Overall, the combined counts for the entire Alaska Peninsula have declined by 65–72% since the mid-1980s. The estimated number of sea otters along the Alaska Peninsula was 9,658 as of 2014. Kamishak Bay, the Kodiak Archipelago and Cook Inlet The eastern extent of the population decline of the 1960s-1990s appears to occur at about Castle Cape. Populations around Kodiak, Katmai, Kamishak, and lower Cook Inlet are stable or increasing (Coletti et al. 2009, Estes et al. 2010). In 2002, Bodkin (2003) found sea otters to be relatively abundant within Kamishak Bay (6,918 otters). In 1994, there were an estimated 9,817 otters in the Kodiak archipelago (USFWS, unpublished data). An aerial

survey of the Kodiak Archipelago conducted in 2004 resulted in an estimate of 11,005 sea otters (CV = 0.19; USFWS unpublished data). Population trends in southwest Alaska changed during the period 2003 - 2011. Declines leveled off and average growth rates approached zero. Some variation was evident but the overall trends were consistent among islands. These results suggest that population declines may have recently stabilized in the western Aleutian Islands, although there is still no evidence of recovery (USGS unpublished data, USFWS unpublished data).

### ***Threats and Stressors***

**Stressor:** Predation

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Available information suggests that predation by killer whales (*Orcinus orca*) may be the most likely cause of the sea otter decline in the Aleutian Islands (Estes et al. 1998). Data that support this hypothesis include: 1) a significant increase in the number of killer whale attacks on sea otters during the 1990s (Hatfield et al. 1998); 2) the number of observed attacks fits expectations from computer models of killer whale energetics; 3) the scarcity of beach cast otter carcasses that would be expected if disease or starvation were occurring; 4) markedly lower mortality rates for sea otters in a sheltered lagoon (where killer whales cannot go) than for those in an adjacent exposed bay; and 5) documentation of elevated mortality rate as the cause of decline, rather than reduced fertility or redistribution (Laidre et al. 2006). The hypothesis that killer whales may be the principal cause of the sea otter decline suggests that there may have been significant changes in predator-prey relationships in the Bering Sea ecosystem (Estes et al. 1998; Springer et al. 2003). For the past several decades, harbor seals (*Phoca vitulina*) and Steller sea lions (*Eumetopias jubatus*), the preferred prey species of transient, marine mammal eating killer whales, have been in decline throughout the western North Pacific. In 1990, Steller sea lions were listed as threatened under the ESA (55 FR 49204). Estes et al. (1998) hypothesized that killer whales may have responded to declines in their preferred prey species, harbor seals and Steller sea lions, by broadening their prey base to include sea otters. Springer et al. (2003) suggest that modern industrial whaling led to declines in great whale populations in the North Pacific, which in turn resulted in killer whales “fishing down” the marine food web; first harbor seals, then fur seals (*Callorhinus ursinus*), sea lions, and finally sea otters in succession as preferred prey were depleted.

**Stressor:** Subsistence Harvest

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Subsistence harvest has reportedly removed fewer than 1,400 sea otters from the southwest Alaska DPS between 1989 and 2011 (average from 2006 - 2010 = 76 per year; range = 30 - 122 per year; USFWS unpublished data; USFWS 2014b). The majority of the subsistence harvest in southwest Alaska occurs in the Kodiak archipelago. Given the estimated population growth rate of 10% per year estimated for the Kodiak archipelago by Bodkin et al. (1999), we would expect that these harvest levels by themselves would not cause a population decline. Some of the largest observed sea otter declines have occurred in areas where subsistence harvest is either nonexistent or extremely low. The best available scientific information does not

indicate that subsistence harvest by Alaska Natives has had a major impact on the southwest Alaska DPS of the northern sea otter.

**Stressor:** Interaction with Commercial Fisheries

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Sea otters are sometimes taken incidentally in commercial set net, trawl, and finfish pot fisheries fishing operations (76 FR 73912). Entanglements of single animals have been reported from the Bering Sea, Bristol Bay, and PWS. In 1992, a total of eight sea otters were observed caught in the Pacific cod pot fishery in the Aleutian Islands (Perez 2006, 2007). In 2002, four incidents of entanglement with no mortality or serious injury were recorded in the Kodiak salmon set net fishery (Manly et al. 2003). Based on Kodiak fisheries data, coupled with self-reporting records from the Bering Sea and Aleutian Island ground fish trawl fishery, it is estimated that fewer than 10 sea otters per year might be killed or seriously injured as a result of entanglement with fishing gear (USFWS 2008b).

**Stressor:** Development

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Habitat destruction or modification is not known to be a major factor in the decline of the southwest Alaska DPS of the northern sea otter. Development of harbors and channels by dredging may affect sea otter habitat on a local scale by disturbing the sea floor and affecting benthic invertebrates that sea otters eat. As harbor and dredging projects typically impact an area of 50 hectares or less, the overall impact of these projects on sea otter habitat is considered to be negligible (USFWS 2008c). However, the cumulative effect of incremental, small losses of critical habitat may affect the population by removing or reducing the availability of PCEs. Between 2002 and 2014, section 7 consultation documented an estimated 20.24 hectares of habitat impacted (including both temporary and permanent impacts) and take by disturbance of 36 otters.

**Stressor:** Research

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Scientific research on sea otters occurs primarily as annual aerial and skiff surveys. When they occur, they last for very short durations of time. Other research includes capture and handling of individuals. During the 1990s, 198 otters were captured and released as part of health monitoring and radio telemetry studies at Adak and Amchitka. In the 2000s, 98 sea otters from the southwest Alaska DPS were live-captured and released as part of a multi-agency health monitoring study (USFWS 2005, 2008b). Accidental capture-related deaths have been rare, with research activities carefully monitored by the Division of Management Authority (DMA).

**Stressor:** Disease

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** Parasitic infection was identified as a cause of increased mortality of sea otters at Amchitka Island in 1951 (Rausch 1953). These highly pathogenic infestations were apparently the result of sea otters foraging on fish, combined with a weakened body condition brought about by nutritional stress. More recently, sea otters have been impacted by parasitic infections resulting from the consumption of fish waste. Necropsies of carcasses recovered in Orca Inlet, Prince William Sound, revealed that some otters in these areas had developed parasitic infections and fish bone impactions that contributed to their deaths (Ballachey et al. 2002). Valvular endocarditis and septicemia have been isolated as a major, proximate cause of sea otter deaths in Alaska (Goldstein et al. 2009). The majority of these deaths are ultimately related to exposure to and infection from *Streptococcus* bacteria.

**Stressor:** Oil Spills

**Exposure:**

**Response:**

**Consequence:**

**Narrative:** The effects of oil on sea otters include short-term acute oiling of fur, resulting in death from hypothermia, smothering, drowning, or ingestion of toxics during preening. While these acute effects are not disputed, a growing body of evidence suggests that oil also affects sea otters over the long term, with interactions between natural environmental stressors and the compromised health of animals exposed to oil lingering well beyond the acute mortality phase (Peterson et al. 2003). The myriad studies that have been undertaken since EVOS provide the most comprehensive data by which to evaluate the effects to wild populations of sea otters to long-term, low-level exposure to hydrocarbons (Bodkin et al. 2002; Stephensen et al. 2001), but documenting chronic effects of EVOS on sea otters has been difficult due to lack of appropriate controls combined with the natural variability among affected resources. Without experimental controls, correlation analysis is the best available inferential tool in assessing the impacts of unpredictable environmental perturbations. Sublethal exposure compromises health, reproduction, and survival across generations (Bodkin et al. 2002). Sea otters consuming prey in habitats contaminated by residual oil have a high likelihood of encountering subsurface oil while excavating prey from sediments (Bodkin et al. 2002). Unlike vertebrates, invertebrates do not metabolize hydrocarbons; thus they accumulate hydrocarbon burdens in their tissues (Short and Harris 1996). Sea otters are, therefore, potentially exposed to residual oil through two pathways: physical contact with oil while digging for prey, and ingestion of contaminated prey. Research has confirmed the persistent exposure of sea otters to residual oil in western PWS. Several authors reported higher levels of a biomarker (P450 1A), which indicates exposure to aromatic hydrocarbons in sea otters sampled from oiled areas of PWS compared to animals sampled from un-oiled areas (Ballachey et al. 2000a; Ballachey et al. 2000b; Bodkin et al. 2002). Chronic, persistent exposure to oil appears to cause reduced productivity and reduced survival of young (Mazet et al. 2001; Ballachey et al. 2003). A comparison of body lengths of sea otters that attained adulthood prior to the spill, relative to post-spill measurements, suggests that food resources were approximately equivalent before and after. These results imply that factors other than body condition are affecting pup survival in western Prince William Sound (Ballachey et al. 2003). Trans-generational effects may arise from direct exposure to a mutagen such as petroleum hydrocarbons, and therefore may be realized long after the contaminant exposure has ceased (Bickham and Smolen 1994). Sea otters are long-lived, with relatively low annual reproductive rates and high annual adult survival; factors that result in reduced reproduction, increased mortality, or increased emigration will eventually lead to depressed population growth rates (Riedman and Estes 1990). Finally, exposure to pollutants such as crude oil may affect sea

otters at a variety of levels of organization, beginning with somatic or germinal cell mutations and leading to a cascade of alterations that go beyond the individual or community to threaten the long-term survival of the population (Bickham et al. 2000; Clements 2000).

### **Recovery**

#### **Recovery Actions:**

- Continue to estimate sea otter population size and trends in southwest Alaska.
- Identify important habitats or areas of special biological significance.
- Ensure that Alaska Native subsistence harvest does not affect recovery.
- Evaluate the potential role of disease as a threat to recovery.
- Continue to evaluate the role of predation as a threat to recovery
- Develop predation management plans, where practical
- Ensure that oil spills do not impede recovery of sea otters and/or negatively affect the nearshore marine environment in southwest Alaska
- Establish an outreach program to mariners on how to avoid striking sea otters
- Monitor occurrence of biotoxins in sea otters and their prey
- Evaluate the feasibility of translocating sea otters to enhance recovery
- Evaluate potential impacts of recreational activities, tourism, and other forms of direct human disturbance
- Maintain the Southwest Alaska Sea Otter Coordinator position within FWS
- Continue and enhance coordination of management efforts among FWS, other Federal agencies, Alaska Natives, and the State of Alaska
- Continue and enhance coordination of research efforts among FWS, USGS, other Federal agencies, the State of Alaska, Alaska Natives, academic institutions, and others
- Develop and continue a program of outreach to stakeholders
- Secure adequate funding for southwest Alaska sea otter management and research needs

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