United States Department of the Interior

FISH AND WILDLIFE SERVICE
Washington, D.C. 20240

In Reply Refer To:
FWS/AES/DCHRS/023287

NOV 30 2005

Ms. Kathleen Clarke
Director
Bureau of Land Management
1849 C Street, N.W.
Washington, D.C. 20240

Dear Ms. Clarke:

The Bureau of Land Management has requested formal programmatic consultation on the amendment of land use plans to adopt a proposed wind energy development program. Effects would extend to nine States. The present document is a biological opinion on the likely effects of that program. We have relied on the Service's Regional Offices in Portland, Oregon, Denver, Colorado, and Albuquerque, New Mexico, for information and analysis to support this opinion. We greatly appreciate the cooperative efforts of BLM staff, particularly Messrs. Lee Otteni and Jim Ramakka, over the past year in carrying out this wide-ranging and complex consultation.

Consultation History

Following several meetings that commenced in late 2004, a request for formal consultation and a Biological Assessment were forwarded to the Service on July 28, 2005, and an electronic file of a map depicting pending and authorized rights of way was received on August 15. Additional meetings and telephone contacts have taken place since then to discuss and clarify issues related to the consultation.

Description of the Proposed Action

The proposed action evaluated in this consultation is the amendment of 52 BLM land use plans in nine western States to adopt the proposed Wind Energy Development Program evaluated in the *Final Programmatic Environmental Impact Statement on Wind Energy Development on BLM-Administered Lands in the Western United States*. As an analytical tool in its biological assessment, BLM based its evaluation of likely effects of its program on the 73 rights of way pending or authorized at the time the assessment was prepared. Consequently, this consultation addresses only effects associated with those rights of way. The plans that are proposed for amendment are listed in Table 1 of the biological assessment provided by BLM. The assessment anticipates adverse effects on 11 listed species and associated critical habitat.

Species Addressed

The following nine species are subjects of formal consultation:
whooping crane (*Grus americana*)
northern aplomado falcon (*Falco femoralis septentrionalis*)
southwestern willow flycatcher (*Empidonax traillii extimus*)
Mexican spotted owl (*Strix occidentalis lucida*)
bald eagle (*Haliaeetus leucocephalus*)
piping plover (*Charadrius melodus*)
least tern (*Sterna antillarum*)
grizzly bear (*Ursus arctos horribilis*)
desert tortoise (*Gopherus agassizii*)

Summary Conclusions

For each of the nine species listed above as likely to be adversely affected, the Service concludes that the program is not likely to jeopardize the species or to destroy or adversely modify any critical habitat designated for it. The bases for these conclusions are presented in the enclosed discussions of individual species. We also concur with conclusions reached in the Biological Assessment that the program is not likely to adversely affect additional listed species or critical habitat.

Effects to the California condor (*Gymnogyps californianus*) would be confined to condors in a nonessential experimental population. Such populations, when they are not located on the National Wildlife Refuge System or National Park System are treated as species proposed to be listed and are thus subject to the conferral requirement of section 7(a)(4) of the Act. The discussion of the condor in the enclosure reflects this requirement and concludes that conferral is not necessary in this instance.

BLM anticipated some direct mortality of gray wolf (*Canis lupus*) during the construction phase of facilities. Nevertheless, based on the information contained in the biological assessement, including Best Management Practices (BMPs) and mitigation measures, the Service believes that the possibility of mortality is so unlikely as to be discountable. Consequently, consultation for this species is concluded informally with our determination that the program is not likely to adversely affect the species.

Assumptions

Based upon the materials provided by BLM and several subsequent clarifying conversations between Service and BLM representatives, we have relied upon the following assumptions regarding the likely effects of the wind energy program:

All disturbance of populations of listed plant species will be avoided.

There will be no effects to wetlands or waterways supporting listed aquatic species.

BMPs described in Appendix 2 of the Biological Assessment will be followed in all projects.

Appropriate mitigation measures described in Appendix 3 of the Biological Assessment will be incorporated into the design and operation of projects.

Individual projects undertaken as parts of BLM's wind energy program that may affect listed species or critical habitat will undergo further consultation to ensure their compliance with section 7 of the Endangered Species Act.

Common elements

Certain elements of this opinion apply equally to all the species addressed. These are summarized below.

Incidental Take

It is not possible at this time to estimate incidental take of listed species likely to occur in the course of the wind energy program. Consequently, all consideration of incidental take and any reasonable and prudent measures required to minimize its effect on the species addressed in this consultation is deferred to further consultation on individual projects.

Conservation Recommendation

We recommend that BLM coordinate with the appropriate Service Field Office prior to planning the construction of any wind energy projects for the most current information on distribution of listed species in the action area.

Reinitiation

This concludes formal consultation on the proposed action outlined in the request. Reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if new information reveals (1) effects of the proposed action on listed species or critical habitat in a manner or to an extent not considered in this opinion, (3) the agency action is subsequently modified in a manner that causes an effect to listed species or critical habitat that was not considered in this opinion or, (4) a new species is listed or critical habitat is designated that may be affected by the proposed action.

Sincerely,

Richard E. Sayers Jr.

Dr. Richard E. Sayers
Chief, Division of Consultation, Habitat
Conservation Planning, Recovery, and
State Grants

Enclosure

Enclosure—Discussion of Individual Species

WHOOPING CRANE

STATUS OF THE SPECIES

Description

The Whooping Crane is in the Family Gruidae, Order Gruiformes (Krajewski 1989, Meine and Archibald 1996). The common name "whooping crane" probably originated from the loud, single- note vocalization given repeatedly by the birds when they are alarmed. As the tallest North American bird, males approach 1.5 m (5 ft) when standing erect, and exceed the greater sandhill crane in height by 12 to 20 cm (5 to 8 in). Males are generally larger than females. Adult plumage is snowy white except for black primaries, black or grayish alulae, sparse black bristly feathers on the carmine crown and malar region, and a dark gray-black wedge-shaped patch on the nape. The juvenile plumage is a reddish cinnamon color. Juveniles achieve typically adult plumage late in their second summer.

Distribution and Abundance

Whooping cranes occur only in North America within Canada and the United States. Approximately 96% of the wild nesting sites occur in Canada and the balance in Florida. Fifty-eight percent of the December, 2003 wild population (185 of 317 individuals) had summered in Canada, with 87 in Florida and 36 in the Wisconsin – Florida population. Seventeen percent of the captive individuals (19) remain in Canada and the balance (95 cranes) is in the United States.

The Aransas-Wood Buffalo National Park Population contained 194 individuals in December, 2003 and is the only self-sustaining wild population. This population nests in the Northwest Territories and adjacent areas of Alberta, Canada, primarily within the boundary of Wood Buffalo National Park (Johns 1998b). In 2003, 61 of the 64 known adult pairs nested. These cranes migrate southeasterly through Alberta, Saskatchewan and eastern Manitoba, stopping in southern Saskatchewan for several weeks in fall migration before continuing migration into the United States. They migrate through the Great Plains States of eastern Montana, North Dakota, South Dakota, Nebraska, Kansas, Oklahoma, and Texas. Their spring migration is more rapid and they simply reverse the route followed in fall. They winter along the Gulf of Mexico coast at Aransas National Wildlife Refuge and adjacent areas. The winter habitat extends 48-56 km along the coast from San Jose Island and Lamar Peninsula on the south to Welder Point and Matagorda Island on the north, and consists of estuarine marshes, shallow bays, and tidal flats (Allen 1952, Blankinship 1976). Some individuals occur occasionally on nearby privately owned pasture or croplands.

The second population of wild Whooping Cranes is nonmigratory (Nesbitt et al. 1997) and occurs in central Florida. This population, known as the Florida Population, has been designated experimental nonessential in the United States by the Service. A third population of wild Whooping Cranes is migratory and was reintroduced starting in 2001.

No Whooping Cranes remain in the Rocky Mountains. The last bird from the cross-fostering experiment disappeared during migration from the winter grounds in 2002 at the age of 19. These birds had summered in Idaho and Montana and wintered in New Mexico, staging in spring and fall near the Monte Vista National Wildlife Refuge, Colorado. In 1989, because of the lack of breeding attempts and high mortality (Garton et al.1989), the Recovery Team decided to discontinue the reintroduction attempt.

Habitat
The Whooping Crane breeds, migrates, winters and forages in a variety of habitats, including coastal marshes and estuaries, inland marshes, lakes, ponds, wet meadows and rivers, and agricultural fields.

Reasons for Listing
In the United States, the Whooping Crane was listed as Threatened with Extinction in 1967 and Endangered in 1970 – both listings were "grandfathered" into the Endangered Species Act of 1973. Critical Habitat was designated in 1978 in Oklahoma, Kansas, Nebraska, and Texas. The Service designated the Whooping Crane as experimental, non-essential on June 26, 2001, within several states including New Mexico and Montana (U.S. Fish and Wildlife Service 2001).

Historic population declines resulted from habitat destruction, shooting, and displacement by human activities. Current threats include limited genetics of the population, loss and degradation of migration stopover habitat, construction of additional power lines, degradation of coastal habitat and threat of chemical spills in Texas. The recovery goal is to establish multiple self-sustaining populations of Whooping Cranes in the wild in North America, allowing initially for reclassification to threatened status and, ultimately, removal from the List of Threatened and Endangered Species. Populations may be migratory or non- migratory.

ENVIRONMENTAL BASELINE

According to 50 CFR 17.11, the Whooping Crane is considered experimental, non-essential in New Mexico and Montana. The only natural wild population nests in Canada and winters on the Gulf coast of Texas (Lewis 1995). Attempts to establish a population in the Rocky Mountains were abandoned in the early 1990s. The Canadian population migrates through Montana, North Dakota, South Dakota, Nebraska, Kansas, Oklahoma, and Texas. Whooping cranes wintered in the Rio Grande Valley in New Mexico but have been absent from this State in the past several years. According to the BLM, the only State indicating that Whooping Cranes may occur near potential wind energy sites was Montana. Critical habitat does not occur within the action area.

EFFECTS OF THE ACTION

None of the ROW applications for wind energy development are in or near known nesting, staging, roosting, or wintering habitat for Whooping Cranes. However, power-line collisions have been documented as a mortality factor for Whooping Cranes (Lewis 1995). During the construction phase of wind power projects, the possibility exists that migrating birds might collide with construction cranes or other structures. Whooping Cranes are high altitude, diurnal migrants, thus, the likelihood of such collisions is low. In addition, there is a small likelihood that cranes could collide with wind turbine towers or elevated powerlines. The potential for cranes being affected is greater for wind energy project operation than that for construction.

The BLM has included conservation measures for the Whooping Crane that would reduce the potential impacts from wind energy activities. These include avoiding the use of guy wires, eliminating all unnecessary lighting at night to limit attracting migratory birds, and reporting wildlife mortalities immediately. However, although the chances are slight, take of Whooping Cranes could occur in the form of death if one were to collide with a turbine. All site-specific projects designed under the proposed Land Wind Energy Development Program would be subject to consultation requirements under Section 7 of the Endangered Species Act.

CONCLUSION

After reviewing the status of the Whooping Crane, the environmental baseline for the action area, effects of the proposed action, and the cumulative effects, it is our biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of the Whooping Crane. No critical habitat has been designated for this species within the action area. We base our conclusion on the following reasons: (1) the risk of Whooping Cranes colliding with energy equipment within the proposed action area is extremely low, (2) the proposed action is outside of Whooping Crane habitat, (3) surveys will be completed prior to construction to ensure that crane habitat is not altered and any potential cranes nearby are not disturbed, and (4) the use of guy wires will be avoided.

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NORTHERN APLOMADO FALCON

STATUS OF THE SPECIES

Description

The northern aplomado falcon (*Falco femoralis septentrionalis*) is one of three subspecies of the aplomado falcon and the only subspecies recorded in the United States (U.S. Fish and Wildlife Service 2005). The northern aplomado falcon (falcon) is intermediate in size between the American kestrel (Falco *sparverius*) and peregrine falcon (Falco *peregrinus*) (U.S. Fish and Wildlife Service 1986). Wings of the falcon are dark above with blackish wing linings and white-edged feathers that form a narrow white line on the trailing edges of the wings. The northern aplomado falcon has a long blackish tail marked with narrow white bands. The falcon has a bold black and white facial pattern, and the cere (nose area), eye-ring, legs, and feet are bright yellow. Within the U.S., the northern aplomado falcon occurs in southern Texas with sporadic sightings in southern New Mexico from migrants out of northern Mexico.

<u>Legal Status:</u> The northern aplomado falcon was listed as an endangered species on February 25, 1986 (U.S. Fish and Wildlife Service 1986). At the time of listing, the Service found that designation of critical habitat was not prudent for the falcon. A recovery plan for the falcon was published on June 8, 1990.

Distribution and Abundance

Historically, falcons occurred throughout coastal prairie habitat along the southern Gulf coast of Texas, and in savanna and grassland habitat along both sides of the Texas-Mexico border, southern New Mexico, and southeastern Arizona (U.S. Fish and Wildlife Service 2005). Falcons were also present in the Mexican states of Tamaulipas, Veracruz, Chiapas, Campeche, Tabasco, Chihuahua, Coahuila, Sinaloa, Jalisco, Guerrero, Yucatan, and San Luis Potosi, and on the Pacific coast of Guatemala and El Salvador (Keddy-Hector 2000). Falcons were common in these areas until the 1940s, but subsequently declined rapidly with no documented nesting attempts by wild birds in New Mexico between 1952 and 2001.

Until 1992, it was believed that the distribution of the falcon in Mexico was restricted to eastern Mexico, from southern Tamualipas south, even though no systematic survey efforts were conducted in northern or central Mexico. In 1992, a population of falcons was confirmed in northern Chihuahua, Mexico, approximately 80 miles south and 50 miles west of the U.S. border on private ranch land (Montoya and Zwank 1995). Since the confirmation of this population, nesting falcons have been located approximately 50 miles west and reliable observations have been reported from the Galeana and Gomez Friaz areas of Chihuahua, approximately 150 miles west of Montoya's study population.

Sporadic sightings of falcons have occurred in New Mexico in every decade since the 1970s (Williams 1997). Whether a remnant population of falcons is present in southern New Mexico, or falcons are immigrating from northern Mexico, is open to speculation. In 1952, a successful nesting of falcons was recorded near Deming, New Mexico (U.S. Fish and Wildlife Service 2005). After a 50-year absence, a nesting attempt was documented in Luna County, New Mexico in the spring of 2001. In 2002, this pair successfully fledged three chicks; in 2003 only a single female was seen in the area of the 2002 nest. The Service does not consider the 2002 nesting pair and any offspring produced as a population. Biologically, the term "population" is not normally applied to a single pair, but rather a minimum of two successfully-reproducing falcon pairs over multiple years. Thus, the few birds in New Mexico could be considered emigrants disconnected from the Chihuhuan population (U.S. Fish and Wildlife Service 2005).

Since 1985, falcons have been propagated and reintroduced to southern Texas on and around the Laguna Atascosa National Wildlife Refuge (NWR) and Matagorda Island NWR under a Safe Harbor Agreement with The Peregrine Fund. In 1995, a released pair nested and fledged one young on Port of Brownsville land; and in 1996, four territorial pairs produced three fledglings in the same vicinity (U.S. Fish and Wildlife Service 2005). These reintroduced falcons were the first known successful nestings in the United States since the previous recorded nesting near Deming, New Mexico in 1952 (Ligon 1961). Beginning in 2002, falcons were released in west Texas, also under a Safe Harbor Agreement with The Peregrine Fund. One hundred and twenty-five young were released at four sites on private ranches near Valentine, Texas. The releases in Texas resulted in at least 37 pairs, which had produced at least 92 young by 2002 (U.S. Fish and Wildlife Service 2005).

There have been no verified sightings of falcons in Arizona since 1940 (Corman 1992).

Habitat

Falcons require open habitats with scattered trees for hunting, roosting, and nesting and an understory of grass and shrubs. Habitat types include yucca-covered ridges in coastal prairie, riparian woodland in open grassland, palm and oak savannas, deciduous woodland, yucca-mesquite grassland, and a variety of other open desert grassland and shrub habitats. Within the Chihuahuan desert, aplomado falcons typically occur in open grassland areas with scattered mesquite and/or soap tree yucca or Torrey yucca (Ligon 1961, Montoya et.al. 1997).

Life History

Falcons are long-lived monogamous birds that court through a series of aerial displays by the male and mutual soaring and diving by the pair. They do not build their own nests; instead they use abandoned stick nests of other bird species, including other raptor species, crows, and ravens. Nests are usually situated in forks of yuccas, or in the tops of mesquite trees. In south Texas, an abandoned raven nest atop a 20-meter electrical tower was used by a pair of falcons in 1995 (Peregrine Fund unpubl. rpt.). Clutches of two to four eggs are laid between January and July with most clutches initiated in April and May (U.S. Fish and Wildlife Service 2005). Both sexes participate in an approximate 32-day incubation (Hector 1981), with young fledging approximately 35 days after hatching. Fledglings may remain in the vicinity of the nest for at least a month after fledging (Hector 1981). The exact lifespan of the falcon is unknown, but one northern aplomado falcon lived in captivity for 12 years.

Falcons feed upon medium-sized birds, insects, rodents, bats, and reptiles. Falcon pairs often hunt cooperatively (Keddy-Hector 2000).

Reasons for Listing

Falcon populations in the United States declined dramatically during the 1930s and 1940s. A number of factors contributed to the decline of the falcon throughout its range, including pesticide contamination, habitat destruction, habitat modification, and stream channelization that reduced riparian foraging habitat (Hector 1987, U.S. Fish and Wildlife Service 1990). Habitat changes brought about through cattle grazing and agricultural practices may have caused some decline in population numbers and distribution. Pesticide exposure was probably the most significant cause of the species' extirpation from the U.S. with the initiation of widespread DDT (dichloro-diphenyl-trichloroethane) use after World War II coinciding with the species' disappearance (U.S. Fish and Wildlife Service 1986). High pesticide residue levels are likely to cause reproductive failure through egg breakage.

Threats: Current factors that may be limiting the recovery of the falcon include: 1) continued pesticide influence; 2) shrub encroachment into Chihuahua grasslands; 3) further loss of habitat due to agricultural development; 4) fragmentation of habitat due to urban expansion and other land uses (e.g., oil and gas development); 5) low densities of avian prey in some areas; 6) the increased presence of the great horned owl (Bubo virginianus), which preys upon the falcon; 7) secondary lead poisoning; and 8) incidental shooting by hunters and poachers (U.S. Fish and Wildlife Service 2005, 1990).

Conservation Measures

Within the U.S., the current recovery emphasis for the aplomado falcon has centered around captive propagation and reintroduction. The Peregrine Fund has reintroduced 1,004 captive-bred falcons in Texas under Safe Harbor Agreements with the Service. On February 9, 2005, the Service published a proposed ruling to establish a Nonessential Experimental Population of northern aplomado falcons in New Mexico and Arizona. The Service has proposed to reintroduce falcons into southern New Mexico to establish a viable resident population.

ENVIRONMENTAL BASELINE

Status of the Species within the Action Area
Within the action area, single falcon sighting have been reported at various locations throughout southern New Mexico. Due to the proximity of a recently documented falcon population in nearby Chihuahua, Mexico, falcon sightings in New Mexico have been attributed to birds dispersing from northern Mexico. The sporadic sightings of falcons in New Mexico are not considered to be individuals of a remnant population, but rather individuals of the northern Mexico population.

To date, no individuals from the introduced populations in coastal southern Texas and west Texas have been documented straying into New Mexico or Chihuahua, Mexico (Meyer and Williams 2005).

Formal consultation was completed in early 1997 on the effects of the implementation of BLM's Resource Management Plan for the Mimbres Resource Area (Consultation #2-22-96-F-330) on the falcon. A non-jeopardy opinion was issued with an indeterminable amount of incidental take of falcons. Reasonable and prudent measures included determining the extent of potential falcon habitat, comparing the suitability of BLM habitat with occupied habitat in Mexico, comparing livestock management practices, evaluating the research described above, and reporting the findings.

In 1998, formal consultation (#2-22-96-F-334) was completed on the proposed and ongoing Air Force activities at Holloman Air Force Base, New Mexico, as well as the continued use of the Air National Guard's military training route (Visual Route 176). The biological opinion concluded that take of the species was anticipated.

In February of 2005, the Service proposed to reintroduce falcons throughout their historical range in Arizona and New Mexico; with release sites on private and public lands within New Mexico. Released falcons are expected to move around within the areas of their release, but may disperse to more distant areas. Thus, should the reintroduction proposal be finalized, falcons are likely to occur within the action area.

Factors Affecting the Species within the Action Area
Large-scale habitat alterations have been implicated as principal factors affecting the status of the falcon in southern New Mexico. Factors asociated with human activities (e.g., excessive livestock grazing) as well as climate change has caused degradation or loss of desert grasslands, including shrub encroachment into these grasslands (Meyer and Williams 2005). The encroachment of mesquite (Prosopis glandulosa) and proliferation of "weed" species such as snakeweed (Gutierrezia spp.) has limited habitat suitability for the falcon.

EFFECTS OF THE ACTION

Direct adverse effects to falcons have the potential to occur from wind energy project construction and operation. The potential exists for falcons to collide with construction cranes, elevated power lines, and vehicles during project construction. Such collisions are likely to cause direct mortality to falcons. The potential also exists for falcons to collide with operating wind turbine towers and blades, or meteorological towers and guy wires. While still slight, the potential for direct adverse impacts to falcons is higher during the operation phase than the construction phase.

Disturbance to falcons may occur during wind energy project construction and operation. Noise and human activity associated with the project could impact nesting falcons (should they move into the action area) if a project were in the vicinity of a nest.

Indirect effects to falcons may occur through the alteration and destruction of suitable habitat. Construction of wind energy projects could remove nesting habitat if large soap tree yucca plants or mesquite trees in grassland were removed. Removal of such potential nest sites may reduce the possibility of nest establishment and reproductive success.

CONCLUSION

After reviewing the current status of the northern aplomado falcon, the environmental baseline for the action area, effects of the proposed action, and the cumulative effects, it is the Service's biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of the northern aplomado falcon. No critical habitat has been designated for this species; therefore, none will be affected.

Potential adverse effects to the falcon as a result of the proposed action are minimized through the implementation of BMPs included in the proposed action.

Currently, no population of falcons occurs in southern New Mexico. Populations exist in coastal south Texas, west Texas, and northern Mexico. Following future reintroductions in southern New Mexico, the potential exists for a viable resident population to occur in historically occupied areas of southern New Mexico, particularly the Chihuahuan Desert. The Service has proposed to designate this reintroduced population as a nonessential experimental population according to section 10(j) of the Endangered Species Act. Thus, the reintroduced population would not be essential to the continued existence of the species.

The BMPs included in the proposed action minimize the potential for adverse effects to falcons currently present in southern New Mexico, which are considered emigrant falcons from the Chihuahuan population, as well as falcons that may be part of the introduced, nonessential experimental population.

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SOUTHWESTERN WILLOW FLYCATCHER

STATUS OF THE SPECIES

Description

The Southwestern Willow Flycatcher (*Empidonax traillii extimus*) is one of 11 flycatchers in the genus *Empidonax* (Order Passeriformes, Family Tyrannidae) breeding in North America. Although the *Empidonax* flycatchers are notoriously difficult to distinguish by sight in the wild, each has unique morphological features, vocalizations, habitats, behaviors and/or other traits that allow biologists to distinguish them.

Distribution and Abundance

The historical breeding range of the Southwestern Willow Flycatcher included southern California, southern Nevada, southern Utah, Arizona, New Mexico, western Texas, southwestern Colorado, and extreme northwestern Mexico (Hubbard 1987, Unitt 1987, Browning 1993). The flycatcher's current range is similar to the historical range, but the quantity of suitable habitat within that range is much reduced from historical levels. The flycatcher occurs from near sea level to over 8500 feet (2600 m), but is primarily found in lower elevation riparian habitats. Throughout its range, the flycatcher's distribution follows that of its riparian habitat; relatively small, isolated, widely dispersed locales in a vast arid region. Marshall (2000) found that 53 percent of Southwestern Willow Flycatchers were in just 10 sites (breeding groups) rangewide, while the other 47 percent were distributed among 99 small sites of ten or fewer territories. In some parts of its northern range, questions of range boundaries between it and other willow flycatcher subspecies exist, including possible intergradations between subspecies.

The historic breeding range of the flycatcher is considered to have been primarily from the Rio Grande Valley westward, including the Rio Grande, Chama, Zuni, San Francisco, and Gila watersheds (Bailey 1928, Ligon 1961, Hubbard 1987); breeding was unconfirmed in the San Juan and Pecos drainages (Hubbard 1987). Contemporary surveys documented that flycatchers persist in the Rio Grande, Chama, Zuni, San Francisco, and Gila watersheds and that small breeding populations also occur in the San Juan drainage and along Coyote Creek in the Canadian River drainage, but breeding remains unconfirmed in the Pecos watershed (Maynard 1995, Cooper 1996, Cooper 1997, Williams and Leal 1998). The Gila Valley was identified by Hubbard (1987) as a stronghold for the taxon, and recent surveys have confirmed that area contains one of the largest known flycatcher populations (Skaggs 1996, Stoleson and Finch 1999). The subspecific identity (*E. t. extimus*. vs. *E. t. adastus*) of willow flycatchers in northern New Mexico has been problematical (Hubbard 1987, Unitt 1987, Maynard 1995, Travis 1996), but recent genetic research supports affiliation with *E.t. extimus* (Paxton 2000).

When the Southwestern Willow Flycatcher was listed as endangered in 1995, approximately 350 territories were known to exist (Sogge et al. 2001). As of the 2001 breeding season, the minimum known number of Southwestern Willow Flycatchers was 986 territories. Though much suitable habitat remains to be surveyed, the rate of discovery of new nesting pairs has recently leveled off (Sogge et al. 2001). A coarse estimate is that an additional 200 to 300 nesting pairs may remain undiscovered, yielding an estimated total population of 1,200 to 1,300 pairs/territories. Unitt (1987) estimated that the total flycatcher population may be 500 to 1000 pairs; thus, nearly a decade of intense survey efforts have found little more than slightly above the upper end of Unitt's estimate. The surveys of the 1990s have been valuable in developing a rangewide population estimate, but cannot identify a rangewide trend over that period. However, some local trends may be evident, as discussed below.

Habitat

The flycatcher breeds in different types of dense riparian habitats, across a large elevational and geographic area. Although other willow flycatcher subspecies in cooler, less arid regions may breed more commonly in shrubby habitats away from water (McCabe 1991), the Southwestern Willow Flycatcher usually breeds in patchy to dense riparian habitats along streams or other wetlands, near or adjacent to surface water or underlain by saturated soil. Common tree and shrub species comprising nesting habitat include willows (*Salix* spp.), seepwillow (aka mulefat; *Baccharis* spp.), boxelder (*Acer negundo*), stinging nettle (*Urtica* spp.), blackberry (*Rubus* spp.), cottonwood (*Populus* spp.), arrowweed (*Tessaria sericea*), tamarisk (aka saltcedar; *Tamarix ramosissima*), and Russian olive (*Eleagnus angustifolia*) (Sogge et al. 1993). Habitat characteristics such as plant species composition, size and shape of habitat patch, canopy structure, vegetation height, and vegetation density vary across the subspecies' range. However, general unifying characteristics of flycatcher habitat can be identified. Regardless of the plant species composition or height, occupied sites usually consist of dense vegetation in the patch interior, or an aggregate of dense patches interspersed with openings. In most cases this dense vegetation occurs within the first 10-13 feet (3-4 m) above ground. These dense patches are often interspersed with small openings, open water, or shorter/sparser vegetation, creating a mosaic that is not uniformly dense. In almost all cases, slow-moving or still surface water and/or saturated soil is present at or near breeding sites during wet or non-drought years.

Thickets of trees and shrubs used for nesting range in height from 6 to 98 feet (2 to 30 m). Lower-stature thickets (6-13 feet or 2-4 m) tend to be found at higher elevation sites, with tall stature habitats at middle and lower elevation riparian forests. Nest sites typically have dense foliage from the ground level up to approximately 13 feet (4 m) above ground, although dense foliage may exist only at the shrub level, or as a low dense canopy. Nest sites typically have a dense canopy, but nests may be placed in a tree at the edge of a habitat patch, with sparse canopy overhead. The diversity of nest site plant species may be low (e.g., monocultures of willow or tamarisk) or comparatively high. Native, non-native, and exotic plants are discussed below.

Occupied sites dominated by native plants vary from single-species, single-layer patches to multi-species, multi-layered strata with complex canopy and subcanopy structure. Site characteristics differ substantially with elevation. Low to mid-elevation sites range from single plant species to mixtures of native broadleaf trees and shrubs including willows, cottonwood, boxelder, ash (*Fraxinus* sp.), alder (*Alnus* sp.), blackberry, and nettle. Average canopy height can be as short as 13 ft (4 m) or as high as 98 feet (30 m). High-elevation nest sites dominated by native plants are more similar to each other than low elevation native sites. Most known high elevation (6,230 feet or >1,900 m) breeding sites are comprised completely of native trees and shrubs, and are dominated by a single species of willow, such as coyote willow (*Salix exigua*) or Geyer's willow (*S. geyeriana*). However, Russian olive is a major habitat component at some high elevation breeding sites in New Mexico. Average canopy height is generally only 10-23 feet (3 to 7 m). Patch structure is characterized by a single vegetative layer with no distinct overstory or understory. There is usually dense branch and twig structure in the lower 6.5 ft (2 m), with high live foliage density from the ground to the canopy. Tree and shrub vegetation is often associated with sedges, rushes, nettles and other herbaceous wetland plants. These willow patches are usually found in mountain meadows, and are often associated with stretches of stream or river that include beaver dams and pooled water.

Southwestern Willow Flycatchers also breed in sites comprising dense mixtures of native trees and shrubs mixed with exotic/introduced species such as tamarisk or Russian olive. The exotics are often primarily in the understory, but may be a component of the overstory. At several sites, tamarisk provides a dense understory below an upper canopy of gallery willows or cottonwoods, forming a habitat that is structurally similar to the cottonwood-willow habitats in which flycatchers historically nested. A particular site may be dominated primarily by natives or exotics, or be a more-or-less equal mixture. The native and exotic components may be dispersed throughout the habitat or concentrated in distinct, separate clumps within a larger matrix. Generally, these habitats are found below 3,940 feet (1,200 m) elevation.

Southwestern Willow Flycatchers also nest in some riparian habitats dominated by exotics, primarily tamarisk and Russian olive. Most such exotic habitats range below 3,940 ft (1,200 m) elevation, and are nearly monotypic, dense stands of tamarisk or Russian olive that form a nearly continuous, closed canopy with no distinct overstory layer. Canopy height generally averages 16 - 33 feet (5 to 10 m), with canopy density uniformly high. The lower 6.5 feet (2 m) of vegetation is often comprised of dense, often dead, branches. However, live foliage density may be relatively low from 6.5 feet (0 to 2 m) above ground, but increases higher in the canopy. The flycatcher does not nest in all of the exotic species that can dominate riparian systems. For example, flycatchers rarely use giant reed (*Arundo donax*) and are not known to use tree of heaven (*Ailanthus altissima*).

Population Dynamics

The total number of southwestern willow flycatchers is small, with an estimated 1100-1200 territories rangewide. These territories are distributed in a large number of very small breeding groups, and only a small number of relatively large breeding groups. These isolated breeding groups are vulnerable to local extirpation from floods, fire, severe weather, disease, and shifts in birth/death rates and sex ratios. Marshall and Stoleson (2000) noted that even moderate variation in stochastic factors that might be sustained by larger populations can reduce a small population below a threshold level from which it cannot recover. The persistence of small populations depends in part on immigration from nearby populations, at least in some years (Stacey and Taper 1992). The small, isolated nature of current southwestern willow flycatcher populations exacerbates the risk of local extirpation by reducing the likelihood of immigration among populations." The vulnerability of the few relatively large populations makes the above threats particularly acute. In recent years, several of the few larger populations have been impacted by fire (San Pedro River) and inundation by impounded water (Lake Mead, Lake Isabella). Also, the flycatcher appears to be a quasi-colonial species (McCabe 1991). At its few large breeding sites, many territories are often packed into relatively small areas, with significant levels of polygyny, extra-pair copulation, and pair re-shuffling (Paxton et al. 1997, Netter et al. 1998, Paradzick et al. 1999). These may be significant factors in maintaining genetic interchange. The presence of a threshold "colony size" may be an important catalyst for successful breeding sites to function.

The Recovery Plan for the flycatcher estimated population persistence over time within an existing network of occupied willow flycatcher sites through an incidence function analysis (Hanski 1994, Lamberson et al. 2000). Results of the model predicted that the greatest stability occurred when flycatcher sites have 10 - 25 territories. Once a threshold of about 25 territories/site is reached, the benefit of increasing the number of birds diminishes. Instead, metapopulation persistence (stability) is more likely to increase by adding more sites rather than adding more territories to existing sites. In addition to maximizing the colonization potential of sites within the metapopulations, this risk-spreading strategy reduces the likelihood that catastrophic events (e.g. fire, flood, disease) will negatively impact all sites. The goal of the Recovery Plan is to assure long-term persistence of the species throughout its range, rather than maximize the number of birds or achieve historical pre-European settlement population levels.

Reasons for Listing

The Southwestern Willow Flycatcher was listed on February 27, 1995, as endangered throughout its range in the U.S. This included Arizona, New Mexico, California, Colorado, Nevada, Texas, and Utah. The primary cause of the flycatcher's decline is loss and modification of habitat. The FWS proposed critical habitat for the flycatcher on October 12, 2004, and final critical habitat was designated on October 19, 2005.

Riparian ecosystems have declined from reductions in water flow, interruptions in natural hydrological events and cycles, physical modifications to streams, modification of native plant communities by invasion of exotic species, and direct removal of riparian vegetation. Wintering habitat has also been lost and modified for this and other Neotropical migratory birds (Finch 1991, Sherry and Holmes 1993). The major mechanisms resulting in loss and modification of habitat involve water management and land use practices, and are discussed below.

ENVIRONMENTAL BASELINE

Of the nine states considered in this programmatic consultation, the species occurs in Colorado, New Mexico, Nevada, and Utah. However, no known flycatcher nests or habitat is present within the action area within these states. Nevertheless, flycatchers could migrate through this area. No critical habitat has been proposed within the action area.

EFFECTS OF THE ACTION

Because no flycatcher habitat is known to occur within the action area, none will be disturbed. Even though potential wind energy projects could be located in areas distant from Southwestern Willow Flycatcher habitat, migrating birds could collide with wind turbine blades or power lines. Passerines have been known to have collisions with towers and guy wires. However, a conservation measure proposed by the BLM is that the use of guy wires will be avoided. This will reduce the likelihood that flycatchers will collide with associated wind energy equipment. Although the risk is low, take of a migrating Southwestern Willow Flycatcher in the form of death could occur if a flycatcher were to collide with wind energy equipment.

CONCLUSION

After reviewing the status of the Southwestern Willow Flycatcher, the environmental baseline for the action area, effects of the proposed action, and the cumulative effects, it is our biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of the Southwestern Willow Flycatcher. No critical habitat has been proposed within the action area; therefore, none will be affected. We base our conclusion on the following reasons: (1) the risk of flycatchers colliding with energy equipment within the proposed action is extremely low, (2) the proposed action is outside of flycatcher nesting habitat, (3) surveys will be completed prior to construction, and (4) the use of guy wires will be avoided.

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MEXICAN SPOTTED OWL

STATUS OF THE SPECIES

Description

The Mexican Spotted Owl (*Strix occidentalis lucida*) is mottled in appearance with irregular white and brown spots on its abdomen, back, and head. *Strix occidentalis* translates as "owl of the west" and *lucida* means "light" or "bright." Unlike most owls, spotted owls have dark eyes. Several thin white bands mark an otherwise brown tail. Adult male and female spotted owls are mostly similar in plumage characteristics, but the sexes can be readily distinguished by voice. Juveniles, subadults, and adults can be distinguished by plumage characteristics (Forsman 1981, Moen et al. 1991). Juvenile spotted owls (hatchling to approximately five months) have a downy appearance. Although the spotted owl is often referred to as a medium-sized owl, it ranks among the largest owls in North America. Of the 19 species of owls that occur in North America, only 4 are larger than the Spotted Owl (Johnsgard 1988). Like many other owls, spotted owls exhibit reversed sexual dimorphism (i.e., females are larger than males).

Distribution and Abundance

The Mexican Spotted Owl occurs throughout the southwestern United States and Mexico in forested mountains and canyonlands (Gutierrez et al. 1995, Ward et al. 1995). It occurs from the four corner states of Utah, Colorado, Arizona, New Mexico, south into small portions of Texas, south through Mexico. While this owl occupies a broad geographic area, it does not occur uniformly throughout its range (U.S. Fish and Wildlife 1995). Instead, Mexican spotted owls occupancy corresponds to isolated mountain systems and canyons within the southwest. Their distribution reflects the availability of forested mountain and canyon habitats (Ganey et al.1998). The assumption has been made that the current owl distribution mimics its historical extent, with a few exceptions however (U.S. Fish and Wildlife 1995). The owl has not been reported recently along major riparian corridors in Arizona and New Mexico, or in historically documented areas of southern Mexico (U.S. Fish and Wildlife 1995). Riparian communities and previously occupied localities in the southwestern United States and southern Mexico have undergone significant habitat alteration since the historical sightings (U.S. Fish and Wildlife

1993).

In the U.S., the majority of owls are found on National Forest System lands. However, surveys have found an increasing number of owls on National Park Service lands within the United States. This increase is a product of more surveys being completed within National Parks (e.g., several parks within southern Utah, Grand Canyon in Arizona, Dinosaur National Park in Colorado, and Guadalupe National Park in west Texas).

The U.S. range of the Mexican Spotted Owl has been divided into six recovery units (RU), pursuant to the 1995 Recovery Plan. These include Colorado Plateau, Southern Rocky Mountains - Colorado, Southern Rocky Mountains - New Mexico, Upper Gila Mountains, Basin and Range - West, and Basin and Range - East. The RUs were identified based on physiographic provinces, biotic regimes, perceived threats to owls or their habitat, administrative boundaries, and known patterns of owl distribution (U.S. Fish and Wildlife Service 1995).

Habitat

Mexican spotted owls nest, roost, forage, and disperse in a diverse array of biotic communities. Mixed-conifer forests are commonly used throughout most of the range (Johnson and Johnson 1985, Skaggs and Raitt 1988, Ganey et al. 1998, Ganey and Balda 1989). In general, these forests are dominated by Douglas-fir and/or white fir, with codominant species including southwestern white pine, limber pine, and ponderosa pine (Brown et al. 1980). The understory often contains the above coniferous species as well as broadleaved species such as Gambel oak, maples, boxelder, and New Mexico locust. In areas within Arizona and New Mexico, forests used for roosting and nesting often contain mature or old-growth stands with complex structure (U.S. Fish and Wildlife Service 1995:26). The complex structure is often difficult to describe; typically it is uneven-aged, multistoried, and has high canopy closure (U.S. Fish and Wildlife Service 1995:27). Several hypotheses have been proposed to explain why spotted owls nest in closed-canopy forests (Gutierrez et al.1995). Barrows (1981) suggested that spotted owls are relatively intolerant of high temperatures and roost and nest in shady forests because they provide favorable microclimate conditions.

Migrating owls have been known to move distances ranging from 5 to 50 km from their nesting areas in the winter. Wintering areas of two owls from the San Francisco Peaks could not be located despite an aerial search covering thousands of square kilometers. This suggests that some owls may move long distances. Also presently unknown is how and why migrating owls select particular wintering areas. For example, the two migrating owls from the Bar-M Canyon study area moved 50 km to winter in pinyon-juniper woodlands. These owls were members of a mated pair, but did not migrate together or even in the same year. They occupied adjacent but non-overlapping winter ranges. Much of the area between these wintering areas and their breeding area consisted of pinyon-juniper woodland that was, at least superficially, similar to the wintering areas occupied.

At present, there is little information on the specific habitat features that migrating spotted owls use in wintering areas. Further, owls use these areas at a time of year when they are unlikely to vocalize (Ganey 1990), making it difficult to locate such areas through calling surveys. This leaves us with no objective means to identify and protect such areas. The types of lowland areas in which wintering owls have been observed cover vast areas, and we have no evidence that suitable wintering areas are limiting. Thus, we see little evidence that specific protective measures for wintering areas or habitats used by migrating spotted owls would be useful or appropriate at this time.

Radio-marked juveniles dispersed in September and October in all study areas, with most dispersing in September. They are capable of moving long distances, but many successful dispersers occupy territories near their natal territory. Distance from the natal site to the last observed location for radio-marked juveniles ranged from <1 to >92 km. They move through a wide variety of habitats during the dispersal period, many of which differ greatly from typical breeding habitat and have no formal protective measures under U.S. Fish and Wildlife Service (1995).

Mexican spotted owls consume a variety of prey throughout their range but commonly eat small- and medium-sized rodents such as woodrats, mice (peromyscid), and voles (microtine). Spotted owls also consume bats, birds, reptiles, and arthropods (Ward 2001).

The Mexican Spotted Owl was listed as a threatened species in 1993 (U.S. Fish and Wildlife Service 1993). The primary reasons for listing were the threat of even-aged timber management and threat of catastrophic wildfire. On August 31, 2004, critical habitat for the Mexican spotted owl was designated. The Service appointed the Mexican Spotted Owl Recovery Team in 1993, and it produced the Recovery Plan for the Mexican Spotted Owl in 1995 (U.S. Fish and Wildlife Service 1995).

ENVIRONMENTAL BASELINE

In the nine states considered in the programmatic biological assessment, the Mexican Spotted Owl occurs in Colorado, Utah, and New Mexico. In Colorado and Utah, the project areas are outside of the range of the spotted owl. In New Mexico, the action area is within the Basin and Range-East Recovery Unit. Owls primarily occur on the Lincoln and Cibola National Forests within this RU. The largest concentration of Mexican Spotted Owls in the Recovery Unit occurs in the Sacramento Mountains on the Lincoln NF. Owls also occur in the Guadalupe Mountains of the Lincoln National Park. No owls are known to occur within the action area. However, juvenile owls have been known to disperse across areas that are not considered habitat. No critical habitat has been designated within the action area.

EFFECTS OF THE ACTION

Because no owls are known to nest within the action area, disturbance from wind energy projects and habitat alteration is not likely to impact the owl. According to the BLM, surveys are required as part of the ROW approval process, and these should ensure that nesting habitat is not disturbed. In addition, the BLM is not proposing energy projects within Mexican spotted owl habitat. However, there is a slight possibility that a dispersing juvenile or adult owl moving through the area during the winter could collide with construction equipment or elevated power lines. In addition, the possibility for mortality of migrating or dispersing Mexican Spotted Owls is greater during wind project operation than during construction. However, the risk is low

CONCLUSION

After reviewing the status of the Mexican Spotted Owl, the environmental baseline for the action area, effects of the proposed action, and cumulative effects, it is our biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of the Mexican Spotted Owl. No critical habitat has been designated for this species within the action area; therefore, none will be affected. We base our conclusion on the following reasons: (1) the risk of owls colliding with energy equipment within the proposed action area is extremely low, (2) the proposed action is outside of spotted owl nesting and roosting habitat, and (3) surveys will be completed prior to construction to ensure that owl habitat is not altered and any potential owls nearby are not disturbed.

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BALD EAGLE

STATUS OF THE SPECIES

This section summarizes the best available information regarding the current rangewide status of the bald eagle. Additional information regarding the bald eagle is contained in the administrative record for this consultation and from the sources of information cited for this species.

The bald eagle *(Haliaeetus leucocephalus*) is a large bird of prey that historically ranged and nested throughout North America except extreme northern Alaska and Canada and central and southern Mexico. Present-day breeding occurs primarily in northern California, Alaska, Oregon, Washington, Minnesota, Wisconsin, Michigan, Maine, the Chesapeake Bay area, Florida, the tri-state corner of Idaho, Montana, and Wyoming, and in parts of Canada. The U.S. Fish and Wildlife Service (USFWS) estimated that the breeding population exceeded 5,748 occupied breeding areas in 1998 (USFWS 1999).

The female bald eagle usually weighs 10 to 14 pounds and is larger than the male, which weighs 8 to 10 pounds. The bald eagles in the northern portions of their range are significantly larger than those from the southern portions. The wings span 6 to 7 feet. The fledgling bald eagle is generally dark brown except for the underwing linings which are primarily white. Between fledging and adulthood, the bald eagle's appearance changes with feather replacement each summer. Young dark bald eagles may be confused with the golden eagle, *Aquila chrysaetos.* The bald eagle's distinctive white head and tail are not apparent until the bird fully matures, at 4 to 5 years of age (64 FR 36454).

The bald eagle occurs in association with aquatic ecosystems, frequenting estuaries, lakes, reservoirs, major river systems, and some seacoast habitats. In coastal areas, the bald eagle primarily nests in forested areas near the ocean, along rivers, and at estuaries, lakes, and reservoirs (Isaacs and Anthony 2001). Inland, bald eagles inhabit primarily riparian habitats in cottonwood groves along streams and rivers, and in coniferous forests. Bald eagles primarily feed on fish, but also on small mammals and carrion. In areas where water is scarce, bald eagles are found nesting away from water sources and will often feed on carrion: road kill, hunting gut piles, and winter kill. They are also known to be kleptoparasitic, stealing prey from other raptors and corvids (Ehrlich et al 1988).

Generally, suitable habitat for bald eagles includes those areas that provide an adequate food base of fish, waterfowl, and/or carrion, with large trees for perches and nest sites. In winter, bald eagles often congregate at specific wintering sites that are generally close to open water and offer good perch trees and night roosts (60 FR 36000).

Nests are most often constructed in the tops of large trees (Howell 1937, Murphy 1965) but can occur on cliffs or on the ground in treeless areas (Troyer and Hensel 1965). Nest sites are usually in large trees along shorelines in relatively remote areas that are free of disturbance (USFWS 1999). Besides the distance to nearest water, other features that influence nest location can include: diversity, abundance, and vulnerability of prey base; and absence of human development and disturbance (Buehler 2000). Inland, mature cottonwood groves found along streams and rivers are typically used as bald eagle nesting habitat. Nest locations usually provide proximity to a food source, good visibility from the nest, and a clear flight path to the nest (Herrick 1924).

One of the most important characteristics of bald eagle nesting habitat is an open forest structure (Anthony et al. 1982). The use of dominant nest trees in forest stands with openings and edges is widespread. One breeding territory in Ohio was occupied for nearly a century (Herrick 1924). Often several alternate nests are built by one pair in a breeding territory, and in any given year, a new nest may be built or an old nest may be reoccupied (Greater Yellowstone Bald Eagle Working Group (GYBEWG) 1996).

For the purposes of this Biological opinion, Bald eagle nesting habitat is defined as any mature stand of conifer or cottonwood trees in association with rivers, streams, reservoirs, lakes or any significant body of water with the availability of a concentrated food source. Furthermore, the Service defines a recently active bald eagle nest as a nest which has been active within the past 5 years.

The USFWS listed the bald eagle south of the 40th parallel as endangered under the Endangered Species Preservation Act of 1966, on March 11, 1967 (U.S. Fish and Wildlife Service 1967). On February 14, 1978, listing was extended to eagles in all of the 48 conterminous states, with eagles in Oregon, Washington, Minnesota, Michigan, and Wisconsin listed as threatened and the remainder as endangered (U.S. Fish and Wildlife Service 1978). The species was reclassified to threatened throughout the 48 States on July 12, 1995 (60 FR 36000). No critical habitat has been designated for this species. The bald eagle has been proposed for delisting (64 FR 36453, July 6, 1999).

Life history and Population dynamics

Bald eagles are long-lived. The longest living bald eagle known in the wild was reported near Haines, Alaska as 28 years old (Schempf 1997). Bald eagles from Arizona are known to have exceeded 12 years of age (Hunt *et al.* 1992). In captivity, bald eagles may live 40 or more years.

Once bald eagles mate, the bond is long-term, though documentation is limited. Variations in pair bonding are known to occur. If one mate dies or disappears, the other will accept a new partner. Bald eagles usually nest in trees near water, but are known to nest on cliffs and (although rarely) on the ground. Nest trees must be sturdy and open to support a nest that is often

5 feet wide and 3 feet deep. Adults tend to use the same breeding areas year after year, and often the same nest, though a breeding area may include one or more alternate nests. A 35-year old nest at Vermilion, Ohio, measured 81⁄2 feet across at the top and 12 feet deep before it blew down in 1925 (Herrick 1932).

Bald eagle pairs begin courtship about a month before egg-laying. In the south, courtship occurs as early as September, and in the north, as late as May. The nesting season lasts about 6 months. Incubation lasts approximately 35 days and fledging takes place at 11 to 12 weeks of age. Parental care may extend 4 to 11 weeks after fledging (Wood, Collopy, and Sekerak 1998).

As they leave their breeding areas, some bald eagles stay in the general vicinity while most migrate for several months and hundreds of miles to their wintering grounds. Young eagles may wander randomly for years before returning to nest in natal areas. Eagles seek wintering areas offering an abundant and readily available food supply with suitable night roosts. Night roosts typically offer isolation and thermal protection from winds. Carrion and easily scavenged prey provides important sources of winter food in terrestrial habitats far from open water.

Winter roosts:

Wintering bald eagles occur throughout the United States but are most abundant in the West and Midwest (USFWS 1983) along major river systems and large bodies of water in the mid-western states, Chesapeake Bay region, Pacific Northwestern states, and states of the intermountain west, including Wyoming, Utah, Colorado, New Mexico, and Arizona. On their winter range, bald eagles may roost singly or in small groups but larger communal roosts are important and may predominate in many areas (Platt 1976). A communal roost is defined as an area usually less than 10 acres in size that contains 6 or more bald eagles on any given night. Critical roost sites are defined as exhibiting traditional use for 5 or more years, and containing 15 or more eagles per night for 14 or more nights per season (U.S. Fish and Wildlife Service 1983). Communal roosts may have developed as information centers in response to the distribution of foods (Fitzner and Hanson 1978, Steenhof 1976, Ward and Zehavi 1973). By congregating with other birds, an individual eagle may enhance its chances of finding unevenly distributed food resources. Communal roosts usually are located in stands of mature old growth conifers or cottonwoods, and roosts may be several miles from feeding areas. Communal roosting also may facilitate pair bonding (Steenhof 1976).

An abundant, readily available food supply in conjunction with one or more suitable night roost sites is the primary characteristic of occupied bald eagle winter habitat. The majority of wintering bald eagles are found near open water where they feed on fish and waterfowl, often taking those that are dead, crippled, or otherwise vulnerable (Stalmaster and Associates 1990, Lingle and Krapu 1986, USFWS 1983). Freedom from human disturbance is an important component of wintering habitat (Fitzner and Hanson 1979, Detrich 1978). When suitable habitat conditions exist, particularly in combination with a lack of human disturbance, wintering bald eagles will also forage in terrestrial habitats capturing small and medium sized mammals (e.g., prairie dogs and rabbits) or scavenging carrion or roadkill, and winter mortalities of big game or livestock (USFWS 1983). Lingle and Krapu (1986) found eagles consumed at least 50 species of fish, birds, and mammals along the North Platte and Platte Rivers during the winters of 1978-1979 and 1979-1980.

Inclement weather is also a major impetus for communal roosting. Roosts are usually located on the leeward sides of mountains, woodlots, or in protected canyons. Communal night roosts are used more often during days of winds greater than 17 km/hr (Steenhof et al. 1980) or during periods of inclement weather (Anderson and Patterson 1988). Platt (1976) observed that the most protected stand on the wintering site was consistently used as a roost during severe weather. Large, live trees in sheltered areas provide a more favorable thermal environment and help minimize the energy stress encountered by wintering eagles. Freedom from human disturbance also is important in communal roost site selection (Buehler et al. 1991, USFWS 1986, USBR 1981, Steenhof et al. 1980). Continued human disturbance of a night roost may cause eagles to abandon an area (USFWS 1983).

Anderson and Patterson (1988) characterized bald eagle winter roosts in Wyoming. Twenty-three roosts were located, which contained from one to 24 eagles. Roosts were located on slopes with northeasterly aspects and typically in forest stands with high densities of conifers and snags. These forest stands had larger and more open trees than the surrounding forest.

The number of eagles using a roost and times of arrival to and departure from the roost are influenced by temperature, precipitation, and wind conditions. During moderate weather, eagles usually leave the roost at dawn, and may ride thermal currents in the vicinity of the roost for up to a half hour before departing for feeding areas (USBR 1981). Eagles have been observed to fly over 15 miles from their feeding areas to roosting sites (Swisher 1964).

Hunting:

The bald eagle typically hunts from perches or while soaring over suitable prey habitat. Prey is often taken on the wing, including snatching fish from surface waters, snaring waterfowl in the air, or pouncing on small mammals. When it is available, carrion is also eaten. General foraging habitats include nearly all upland and aquatic habitats that support sufficient prey species. Suitable general foraging habitats can include grasslands, shrublands, streams, rivers, lakes, and reservoirs. Concentrated foraging habitats are typically habitats that support high densities of prey species and can often be a reliable source of prey for wintering bald eagles. Concentrated foraging habitats can include big game crucial winter ranges, ice-free water bodies that support fish and waterfowl during the winter, cattle and sheep stockyard operations, and big game roadkill.

STATUS AND DISTRIBUTION

An estimated one-quarter to one-half million bald eagles occurred on the North American continent when Europeans first arrived. Initial eagle population declines probably began in the late 1800s and coincided with declines in numbers of waterfowl, shorebirds, and other prey species. Nesting habitat was modified or destroyed, and killing of bald eagles was also prevalent. These factors reduced bald eagle numbers until the 1940s when bald eagles were protected under the Bald Eagle Protection Act (16 U.S.C. 668). The Bald Eagle Protection Act slowed the decline of bald eagle populations by prohibiting numerous activities that adversely affected eagles and increasing public awareness of their plight. The widespread use of dichloro-diphenyl-trichloroethane (DDT) and other organochlorine compounds in the 1940s for mosquito control and as a general insecticide caused additional declines in bald eagle populations. DDT accumulated in individual birds following ingestion of contaminated food. DDT breaks down into dichlorophenyl-dichloroethylene (DDE) and accumulates in the fatty tissues of adult females, leading to impaired calcium release necessary for eggshell formation. Thinner eggshells led to reproductive failure, which is considered a primary cause of declines in the bald eagle population. DDT was banned in the United States in 1972 (60 FR 36000). Bald eagles have increased in number and expanded in range since the banning of DDT and other persistent organochlorine compounds, and due to habitat protection and additional recovery efforts. The breeding population exceeded 5,748 occupied breeding areas in 1998 (64 FR 36454).

Human disturbance of bald eagles is a continuing threat that may increase as numbers of bald eagles increase and human development continues to expand into rural areas (64 FR 36454). Threats to bald eagles include entanglement in monofilament fishing line and fish tackle; overgrazing and related degradation of riparian vegetation; malicious and accidental harassment including shooting, disturbance from off-road vehicles and recreational activities (especially watercraft), and low-level aircraft overflights; alteration of aquatic and riparian systems for water distribution systems and maintenance of existing water development features such as dams or diversion structures; collisions with transmission lines; poisoning; and electrocution (Beatty and Driscoll 1999; Stalmaster 1987).

ENVIRONMENTAL BASELINE

Regulations implementing the Act (50 CFR 402.02) define the environmental baseline as the past and present impacts of all Federal, State, or private actions and other human activities in the action area, the anticipated impacts of all proposed State or Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation process.

The action area is defined at 50 CFR 402 to mean "all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action".

STATUS OF THE SPECIES WITHIN THE ACTION AREA

Colorado:

Colorado is a popular wintering area for bald eagles (Harmata 1984). In recent years the

Colorado Division of Wildlife (CDOW) midwinter bald eagle counts have averaged approximately 800 to 900 eagles (J. Craig, CDOW, pers. comm.). Over the last 10 years, counts have ranged from a high of 1,235 in 1994 to a low of 595 in 2001. In 2004, an estimated 849 bald eagles were found to be wintering in Colorado along standardized survey routes. Colorado's bald eagle surveys take place on a series of 34 standardized routes along the state's major river drainages and associated reservoirs, including the South Platte, Colorado, White, Yampa and Arkansas rivers, and Chatfield and Bonny reservoirs. As many as 40 miscellaneous routes are also surveyed. Biologists use low-flying airplanes to track raptors sitting in trees along waterways.

In 1997, 29 nesting pairs of bald eagles were located in the state (J. Craig, pers. comm.). However, the maintenance of suitable wintering grounds is more important to the overall stability and recovery of the species than is the contribution made by nesting pairs.

Montana:

Montana operates under the Pacific Bald Eagle Recovery Plan (USFWS 1986) that uses the zone approach to differentiate subpopulations and habitats important to bald eagle recovery in the Pacific recovery area. The management zone approach is central to the recovery process because establishment of well-distributed bald eagle populations and habitats is essential for recovery of the species in the recovery area. There are seven bald eagle management zones in Montana.

Bald eagles occur year-round in Montana and occur in all 49 latilongs (Bergeron et al. 1992, Montana Bird Distribution Committee (MBDC) 1996). Currently, about 75% of nesting pairs in Montana can be found in the western third of the state, west of the Rocky Mountain Front (Montana Bald Eagle Working Group (MBEWG) 1994). Most breeding areas are associated with large montane rivers, lakes, impoundments and coniferous and cottonwood (*Populus spp.*) forests. The remaining 25% of pairs are scattered throughout the eastern two thirds of the state along major prairie rivers. Most prairie breeding areas are associated with the Yellowstone River, but a number of bald eagles nest along the Bighorn, Tongue and lower Missouri Rivers (MBEWG 1994). Wintering and migration habitat is distributed throughout Montana.

In 1978 there were only 12 breeding areas for bald eagles known in Montana (Servheen 1978). In the autumn of 1995, 222 current or historical breeding areas were known in Montana (MBEWG 1995). By the end of 2001, there were 297 known bald eagle nesting territories in the state and 220 of the 261 territories surveyed were active (MBEWG 2001). Out of the 220 active nests, 188 were successful, producing 347 young. In 2003, 309 bald eagle nest territories were recorded, with 216 active nests out of 260 checked (MBEWG 2003).

Nevada:

There are two established nesting territories in western Nevada with recently confirmed reports of successful reproduction (Marlette Lake and Lahontan Reservoir). There are four other locations in Nevada where bald eagle nesting attempts were documented between 1985 and 2005 (Carson River, Salmon Falls Creek, Stillwater NWR, and Ruby Valley). However, in all four of these cases the nests failed and no young were produced. There is also a historic nest account from 1866 of bald eagles nesting at Anaho Island (now Anaho Island NWR) in Pyramid Lake. But there are no modern records of bald eagles nesting at the Anaho Island site. Bald eagles will likely continue to expand nesting efforts in Nevada as their numbers in adjacent states increase and displaced/dispersing individuals seek new territories.

Bald eagle numbers are greatest in Nevada during the wintering/migration period. There are at least 20 sites in Nevada (lakes, reservoirs, rivers) where bald eagles have been documented during the wintering period. These include Lake Tahoe, Salmon Falls Creek, Antelope Valley, Ruby Lake NWR, Pahranagat NWR, Kirch Wildlife Mgmt. Area, Stillwater NWR, Lahontan Reservoir, Carson Valley/Mud Lake, and Lake Mead. More recently bald eagles have been documented wintering in the Carson Valley in an agricultural area with many cattle ranches. They feed on the afterbirth from livestock typical in January and February.

New Mexico:

Bald eagles migrating from the north are found in a variety of habitats throughout the southwest recovery region. Wintering bald eagles often congregate at specific sites that are generally close to open water and offer good perch trees and night roosts (USFWS 1995). Bald eagles wintering in New Mexico are often found in upland habitats and can be considered an uncommon to locally common migrant and wintering species, with multiple reports from all 33 Counties (S. Williams, Ph.D., New Mexico Department of Game and Fish, personal communication, October 20, 2005).

In New Mexico, the bald eagle is a winter migrant from the northern border southward to the Gila, lower Rio Grande, middle Pecos, and Canadian valleys. Key habitat areas include winter roost and concentration areas, such as Navajo Lake, the Chama Valley, Cochiti Lake, the northeastern lakes, the lower Canadian Valley, Sumner Lake, Elephant Butte Lake, Caballo Lake, and the upper Gila Basin. These sites have large numbers of waterfowl from November to March and fisheries supported by reservoirs that provide the prey base to support foraging eagles. Winter and migrant populations seem to have increased in New Mexico, apparently as the result of reservoir construction and the expansion of fish and waterfowl populations.

The U.S. Army Corps of Engineers has conducted two annual aerial winter surveys for bald eagles in the Middle Rio Grande (Albuquerque to Rio Chama confluence) and Rio Chama from 1988 through 1996. The mean annual number of bald eagle sightings from 1988 to 1996 is 64.

Bald eagles occur around Navajo Reservoir and along the San Juan River, primarily as winter residents. Winter concentration areas occur around Navajo Reservoir and some of its tributaries. Winter concentration areas have been designated along the Piedra, San Juan, and Pine Reservoir arms in Colorado, and in several areas around the Reservoir in New Mexico. Food sources include fish and carrion. Night roost sites consisting of undisturbed cottonwood groves or ponderosa pine groves, from which eagles disperse daily for feeding, are important factors in maintaining wintering populations.

Bald eagles usually overwinter on the Pecos River from October to April. The density depends on prey, suitable perch and roost sites, weather conditions, and sometimes, lack of human disturbance. Individuals are generally found from the headwaters of the Pecos River to just downstream from Fort Sumner. A small number of bald eagles may be found wintering throughout all National Forests in New Mexico. One known winter roost area, Monument Spring, is monitored each year (Geo-Marine, Inc. 2001) on the Lincoln NF. In 2003, 11 bald eagles were located along the Capitan Mountains of the Lincoln NF. The exact location of the roost(s) is unknown at this time (U.S. Forest Service 2004).

Historic evidence to document bald eagles nesting in New Mexico is lacking, although unverified reports suggest one or two pairs may have nested in southwestern New Mexico prior to 1928. In the mid-1980s, a pair established a territory in Colfax County in an area where bald eagles concentrated in winter, and in 1987 an active nest was discovered nearby which produced two fledglings that year. In 1988, an active nest was discovered in Sierra County, also in an area of wintering eagle concentration; the nest fledged one young that year. Through 1999, those two nests together fledged a minimum of 31 young, with that in Colfax County being one of the more productive nests in North America. Additional nesting activity was recorded elsewhere after the mid-1980s, always in areas of wintering concentrations, including in San Juan, Rio Arriba, Quay, and Sierra counties, but in each instance eagles built nests only to abandon the effort prior to egg laying; such “practice” nests are not uncommon among inexperienced adults. In 1998, two additional nests were discovered in Colfax County, and each fledged young in both 1998 and 1999 (five young total) (S.O. Williams 2000).

The occurrence of breeding bald eagles in the state of New Mexico is very limited. In 2001, the New Mexico Game and Fish Department (NMGFD) reported that four bald eagle nest sites occurred in New Mexico, all on private lands. Two active nests have been monitored since the 1980’s. Fledging success has been good at both locations (U.S. Forest Service 2001). In May of 2005, a bald eagle nest was discovered at Quemado Lake on the Gila National Forest (W. Murphy, Forest U.S. Fish and Wildlife Service, 2005, unpubl. data).

Oregon:

Bald eagles breed in 32 of 36 counties in Oregon; all counties except for Sherman, Gilliam, Morrow, and Malheur (Isaacs and Anthony 2001). The bald eagle is a fairly common breeder at Upper Klamath Lake, along the Columbia River below Portland, and at Crane Prairie and Wickiup Reservoir. It is found throughout the state during the non-breeding season.

There is variation locally in the number of eagles and the timing of peak abundance due to weather and food supply. They are very common in winter and early spring in the Klamath (Keister et al. 1987) and Harney (Isaacs and Anthony 1987) basins, the Columbia River estuary (Garrett et al. 1988), and Lake Billy Chinook (Concannon 1998); common in winter and early spring at Hells Canyon, Oxbow, and Brownlee reservoirs, and Wallowa and Grande Ronde reservoirs (Isaacs et al. 1992), Crooked River Valley above Prineville Reservoir (Isaacs et al. 1993), south end of the Willamette Valley (Isaacs unpubl. data), John Day Reservoir above Service Creek (Isaacs et al. 1996), Columbia River in the Umatilla NWR area (Isaacs unpubl. data), Goose Lake Valley (Isaacs unpubl. data), Summer Lake and Chewaucan River downstream of Paisley (R.L. Madigan p.c.), and at Sauvie Island (Isaacs unpubl. data); common in fall at Wickiup Reservoir (Isaacs unpubl. data, G.J. Niehuser p.c.) and Odell Lake (Crescent RD 1998).

Less is known about migrant and winter bald eagle populations in Oregon. Midwinter Bald Eagle Surveys (MBES) conducted in early Jan 1979-1983 (Opp 1979, 1980b, Oreg. Dept. of Fish and Wildl. unpubl. data) and 1988-2001 indicate January populations as stable or increasing (1988-1992 average 593; 1997-2001 average 692; Isaacs unpubl. data). MBES do not occur when eagles are most abundant; local populations have 2-3 times more eagles in February and early March (Keister et al. 1987, Isaacs and Anthony 1987, Garrett et al. 1988, Isaacs et al. 1992, 1993, 1996), indicating minimum population at peak abundance of 1400-2100.

Utah:

Occupied breeding territories in Utah continue to increase. In 2005, there were eight known nesting pairs in widely scattered locations throughout Utah: one in Dagget County, one in Davis County, one in Duchesne County, one in Emery County, three in Grand County, one in Wayne County. Four of these nesting pairs have been active since 1983, 1988, 1991, and 1996 respectively. Nesting success by two of these four pairs has been erratic as determined through annual monitoring (UDWR 2005). One of the four pairs has had stable nesting success, and the last has enjoyed exceptional nesting success: twenty-five young have been produced over a ten-year period (UDWR 2005). The remaining four pairs (of the eight known to exist in the state) have only recently been found: one in 2004 and three in 2005 (UDWR 2005). It is strongly suspected that other nesting pairs exist in Utah at the present time (UDWR 2005). Historically, breeding occurred in at least five other counties including Tooele, Utah, Wasatch, Summit, and Wayne.

In addition to breeding bald eagles, Utah provides habitat for a large wintering population of bald eagles, which occupy the State between October and April each year. Approximately 1,200 bald eagles winter in Utah each year.

Washington:

In Washington, bald eagles are most common along saltwater, lakes, and rivers in the western portion of the state and along the Columbia River east of the Cascade Mountains (Stinson et.al, 2001). Resident, breeding eagles are found throughout the state near large bodies of water. Most nesting habitat in Washington is located in the San Juan Islands and on the Olympic Peninsula coastline (Stinson et.al, 2001). The primary wintering range of bald eagles in Washington is Puget Sound and its major rivers. Most eagles wintering in Washington occur along the Skagit, Nooksack, and Sauk River Basins (USFWS 1986).

Wyoming:

The bald eagle is considered a year-round resident along the Snake River, with an influx of winter residents and migrants during the spring and fall. The Wyoming portion of the Greater Yellowstone Ecosystem contained only 17 nesting territories in 1986 (USFWS 1986). By 1999, the number of nesting territories in the Wyoming portion of the Greater Yellowstone Ecosystem had risen to 59. All 59 known bald eagle nesting territories were surveyed by the Wyoming Game and Fish Department in 1999 and 58 of those territories were occupied. Of those 58 nests occupied in 1999, 33 nests were successful with 47 young fledged (WGFD 2000).

Within the Snake River BLM planning area, there are currently 15 known bald eagle breeding territories. Fourteen of the known bald eagle territories within the planning area are located on private land. One territory is located on BLM-owned land. Thirteen of the fifteen known territories contained active breeding pairs in 2003. Of the thirteen active breeding pairs in 2003, seven fledged young while six failed to fledge any young. Of the 7 successful nesting attempts, 9 young bald eagles were fledged. (Susan Patla, Personal Communication, Nongame biologist WGFD).

FACTORS AFFECTING THE SPECIES' ENVIRONMENT WITHIN THE ACTION AREA

The southwestern bald eagle population is exposed to hazards from the regionally increasing human population (USFWS 2003). These include extensive loss and modification of riparian breeding and foraging habitat through clearing of vegetation, changes in groundwater levels, groundwater pumping, surface water diversion, alteration of natural hydrologic regimes, changes in water quality, and alteration of prey base by exotic aquatic species (USFWS 2003). In addition, recreation is often concentrated along rivers and streams (USFWS 2003). Some of the continuing threats and disturbances to bald eagles include entanglement in monofilament fishing line and fish tackle; overgrazing and related degradation of riparian vegetation; malicious and accidental harassment including shooting, disturbance from off-road vehicles and recreational activities (especially watercraft), and low-level aircraft overflights; alteration of aquatic and riparian systems for water distribution systems and maintenance of existing water development features such as dams or diversion structures; collisions with transmission lines; poisoning; and electrocution (Beatty and Driscoll 1999; Stalmaster 1987).

Collisions with vehicles may also pose a threat to eagles foraging on roadkills. Electrocutions and collisions due to power lines are another cause of eagle mortality. Current research is helping to establish guidelines to create safer utility lines utilizing anti-perch devices.

As early as 1922, researchers noted the electrocution of raptors. However, not until the 1970's did researchers become aware of the magnitude of the problem. Franson et al. (as cited in Avian Power Line Interaction Committee (APLIC) 1996) summarized that 12 percent of the known bald eagle mortalities were the result of electrocution. Electrocution deaths of bald eagles have been documented across the country (APLIC 1996). Between 1986 and 1996, electric utility company records from across the western United States and Canada showed that 118 bald eagles and an additional 358 unidentified eagles were electrocuted (Harness 2002). In predominantly treeless areas, power poles may be the only perches available to bald eagles.

EFFECTS OF THE ACTION

Construction components of the Proposed Action have the potential to remove snags or live trees that the bald eagle could use for perching, roosting, or nesting.

Effects from construction of wind power projects to the bald eagle and its suitable habitat are bulleted below.

- Short-term Direct Effects:
  - Visual or auditory disturbance or displacement of individuals from low-flying aircraft, vehicles, heavy equipment, and humans during construction and maintenance actions, affecting foraging, roosting, or reproductive behavior;
  - Nest abandonment or mortality of young or eggs, resulting in the loss of one year's recruitment; and
  - Illness or mortality due to inadvertent chemical contamination of terrestrial species or aquatic habitats and species (special status species or prey species) during pesticide and herbicide applications.
- Short-term Indirect Effects:
  - Changes in food or prey quality and quantity or foraging habitats.
- Long-term Effects:
  - Loss of nesting, roosting, and foraging habitat

Effects from operations of wind power projects to the bald eagle and its suitable habitat are bulleted below.

- Short-term Direct Effects:
  - Mortality or injury of adults, or young from collisions with wind turbines, associated power lines, or from vehicles or equipment used during maintenance operations;
  - Illness or mortality due to inadvertent chemical contamination of terrestrial species or aquatic habitats and species (special status species or prey species) during pesticide and herbicide applications.
- Long-term Effects:
  - Avoidance of area due to turbine noise or light disturbance
  - Loss of foraging habitat

CONCLUSIONS

The conclusions of this biological opinion are based on full implementation of the project as described in the "Description of the Proposed Action" section of this document, including the resource protection measures that were incorporated into the project design. We understand the measure (F.3) adopted to establish buffer zones for raptors to include compliance with applicable standards adopted by the Service to avoid disturbance of eagles.

After reviewing the status of the bald eagle, the environmental baseline for the action area, effects of the proposed action, and the cumulative effects, it is our biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of the bald eagle. No critical habitat has been designated for this species; therefore, none will be affected. We base our conclusion on the following:

The BMPs and mitigating measures adopted at the programmatic level and applied to projects carried out within this program will effectively minimize the likelihood of any adverse effects to the species.

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PIPING PLOVER

Unless otherwise noted, the following information is from the “Revised Recovery Plan for Piping Plovers (*Charadrius melodus*) Breeding on the Great Lakes and Northern Great Plains”

(U.S. Fish and Wildlife Service 1994). The piping plover was federally listed in 1985, as threatened in the northern Great Plains and Atlantic Coast and endangered in the drainages of the Great Lakes.

The Service designated critical habitat for the northern Great Plains breeding population of the piping plover in 2002 (67 FR 57638, dated September 11, 2002). The designation includes 19 critical habitat units including prairie alkali wetlands and inland and reservoir lakes, totaling approximately 183,422 acres (74,228.4 hectares) of which two are in Montana (Sheridan and McCone counties), and 5 critical habitat units found along portions of 4 rivers in the States of Minnesota, Montana (Bowdoin and Charles M. Russell National Wildlife Refuges), Nebraska, North Dakota, and South Dakota, totaling approximately 1,207.5 river miles (1,943.3 kilometers). There was no critical habitat designated in Colorado, New Mexico or Wyoming.

Species Description

The piping plover is one of six North American species of belted plovers in the Family *Charadriidae*. It is a small plover (body length of 6.7 inches, wing lengths 4.3 to 4.7 inches) with sandy colored upper parts, white beneath, and orange-colored legs. During the breeding season, both sexes acquire a black forehead, a single black breast band, and an orange bill. In the non-breeding season, the black markings and orange bill color are lost. Juveniles appear similar to adult non-breeding plumage. Adult plumage is acquired in the second year. The piping plover takes its name from the beautiful vocalizations made during courtship displays.

Originally described as a race of the Old World common ringed plover (*Charadrius hiaticula*), the piping plover was first considered a separate species by Ord (1824). The American Ornithologists Union (AOU) Checklist listed the species as *Charadrius melodus* in 1931. Scientists have debated for years the validity of the designation of two subspecies, *C. m. melodus* (Atlantic birds), and *C. m. circumcintus* (inland birds), which the AOU adopted in 1957 (U.S. Fish and Wildlife Service 2000). In 1998, the AOU returned to the single-species designation after genetics were reported similar between the groups (Haig 1988, AOU 1998).

Life History

Piping plovers arrive on breeding grounds from mid-April through mid-May. Courtship and pair formation occurs on the breeding grounds. Piping plovers nest on sparsely vegetated sandbars, aggregate mining spoil piles, and reservoir shorelines. Piping plovers also nest on shorelines of alkali lakes in the prairie pothole region of the United States and Canada. Nests are initiated in early to mid-May, incubation lasts 25 to 31 days, with eggs hatching from late May through mid-June. The precocial chicks generally remain on the nesting territory, slowly expanding their movements until fledging around 21 days after hatching. Adults may leave the breeding grounds before their young, as early as mid-July. The young leave a few weeks later, and are usually gone by late August (U.S. Fish and Wildlife Service 1988).

The piping plover defends a relatively large breeding territory, resulting in low breeding densities. The piping plover is mainly monogamous though sequential polyandry has been occasionally observed (Haig 1992). The nest is a simple scrape on the ground, lined with pebbles. Normal clutch size is four, and no more than one brood of young is raised in a breeding season. Both adults share in incubation and care of the young (U.S. Fish and Wildlife Service 1988).

Piping plovers forage visually for invertebrates in shallow water and associated moist substrates. Corn and Armbruster (1993) emphasize the importance of river channel habitat for foraging. Although the food base is similar taxonomically, invertebrate catch rates and densities are higher on river channel sites than on spoil piles. Invertebrate catch rates also increase more dramatically over the course of the summer on riverine sites than on sand pit sites.

Food availability can be critical for piping plovers. Chick mortality is correlated with reduced growth rates (Cairns 1982), potentially a result of reduced food availability. During the breeding season, energy demands on shorebirds are typically higher than energy intake rates (Ashkenazie and Safriel 1979), and even on the best of foraging habitats, breeding shorebirds may not be able to forage efficiently enough to meet those demands (Evans 1976). In areas where invertebrate densities are not high, such as the riverine habitats occupied by piping plovers on the northern Great Plains, reductions in invertebrate density, and thus in feeding efficiency can reduce shorebirds' abilities to minimize the energy deficit they are likely to face during the breeding season (Goss-Custard 1977a and 1977b, Connors et al. 1981).

Population Dynamics

Most nesting sites on the prairies that have been monitored for 10 years or more have experienced a decline; with an overall decrease of 13 percent between 1987 and 1991 (Haig and Plissner 1992). Ryan et al. (1993) developed a stochastic population growth model, using empirical demographic data and indicated that the Great Plains piping plover population is declining 7 percent annually. Unchecked, this decline would result in extirpation of the species in approximately 80 years.

Status and Distribution

The historical breeding distribution of the northern Great Plains population of the piping plover included beaches and sandbars of the prairie rivers and alkali wetlands from Alberta, Canada, south to the Texas Panhandle and east to Iowa (U.S. Fish and Wildlife Service 1988). Currently, piping plovers are widely distributed in small populations across their breeding range. The majority of adults are found in the Great Plains while the number of birds and breeding sites on the Great Lakes remains small.

The current breeding range for the Great Plains extends from alkali wetlands in southeastern Alberta through southern Saskatchewan and Manitoba to Lake of the Woods in southwestern Ontario and northwestern Minnesota, south along major prairie rivers including the Yellowstone, Missouri, Niobrara, North Platte, Platte, and Loup Rivers, reservoirs in southeastern Colorado, and alkali wetlands in northeastern Montana, North Dakota, and South Dakota. Occasional breeding has occurred in Oklahoma and northern Saskatchewan.

The piping plover winters along Gulf Coast beaches and mud flats from Florida to northern Mexico. Large numbers of piping plovers are observed wintering along the Texas coast. Band return data indicate that most Great Plains piping plovers winter along the Gulf Coast, though a few have been observed along the Atlantic coast (U.S. Fish and Wildlife Service 1988).

Plover nesting in Colorado was first observed in 1949 and a few reports of non-nesting birds occurred during the 1950-1960s, but there are no reports of nesting between 1949 and 1989 (U.S. Fish and Wildlife Service 2001). During the 1990s, nesting has been observed on various reservoirs in the vicinity of the Arkansas River (Plissner and Haig 1997, Nelson and Carter 1991, Nelson 1997, Nelson 1999.

In 1990, the CDOW contracted with the Colorado Bird Observatory to conduct piping plover inventories and habitat surveys on the eastern plains of Colorado (CDOW 1994). Additional surveys were conducted in the South Platte River drainage in northeast Colorado and the Arkansas River drainage in southeast Colorado; however, nesting activity was found only in southeast Colorado.

In 2003, six plover nests were found at John Martin Reservoir and one each at Adobe Creek Reservoir and Neenoshe Reservoir. Twenty young fledged in 2003, out of the original 32 eggs. Plover use of the individual reservoirs listed above varies from year to year due to water levels (Nelson 2003). Due to water availability and water allocations, some of the reservoirs may not be replenished during periods of drought. Changes in the availability of water and exposure of sand bar flats vary from year to year, as does the availability of nesting habitat. Consequently, use varies among the various reservoirs.

<u>Threats</u>
The piping plover was listed because of a substantial decline in the species and its habitat, shrinkage of the breeding range, and continued threats to the species, its habitat and range. Threats to the species in the Great Plains include the historic and continued enormous loss of appropriate sandy beaches and other littoral habitats due to recreational and commercial development; damming and channelization of rivers; withdrawal of water for irrigation and other purposes; the drainage and modification of Great Plains wetlands; predation; trampling by large confined herds of cattle; human disturbance from recreational use of rivers, alkali wetlands, and bare, alluvial islands.

STATUS OF THE SPECIES WITHIN THE ACTION AREA

Within the action area, piping plovers may occur on BLM lands in Colorado, Montana, New Mexico, and Wyoming. The piping plover uses the margins of alkali lakes, beaches, and sparsely vegetated sand bars of major rivers as nesting habitat (Haig and Elliot-Smith 2004). Migration and wintering habitat consists of similar habitats as well as mud flats and alkali flats. Human disturbance at beaches and water regulation policies in rivers are the primary current threats to the plover (U.S. Fish and Wildlife Service 2004).

Colorado

Much of the information for Colorado was taken from the *Colorado Bureau of Land Management Statewide Programmatic Biological Assessment for Least Tern (Sterna antillarum), Piping Plover (Charadrius melodus), Preble's Meadow Jumping Mouse (Zapus hudsonius preblei), and Arkansas Darter (Etheostoma cragini)* (Real West Natural Resource Consulting 2004).

Nearly 3,000 acres of BLM surface ownership includes piping plover foraging and nesting habitat in Colorado. These areas include land under and surrounding Neegronde, Neenoshe, Neeskah, and Neesopah reservoirs. Plover nesting and foraging areas around Adobe Creek Reservoir and John Martin Reservoir do not include BLM lands. There are no federal mineral estate lands, either with or without BLM surface ownership, on any foraging or nesting habitat.

Montana

The most westerly breeding piping plovers in the United States occur in Montana on sandflats above the west end of Fort Peck Dam, on the shorelines of the Big Dry Arm of Fort Peck Reservoir, on the saline wetlands near Dagmar and Medicine Lake National Wildlife Refuge, and in Pondera County in the western part of the state (U.S. Fish and Wildlife Service 1994).

More recently, piping plovers are known to breed primarily in three distinct areas: in the extreme northeast portion of the state (Northeast Montana Wetland Management District), Nelson Reservoir and Bowdoin National Wildlife Refuge, and along the Missouri River including Fort Peck Reservoir (Montana Piping Plover Work Group 2003). Plovers have also sporadically reproduced at Alkali Lake in Pondera County, which is the extreme western edge of their distribution. There are no winter records of plovers in the state.

New Mexico

The piping plover has been reported in New Mexico on only seven occasions, most recently in April 2001. In New Mexico, the piping plover is known as a rare spring (April) migrant, having been verified at Springer Lake (Colfax County), Bosque del Apache National Wildlife Refuge (Socorro County), on Fort Bliss in Texas and New Mexico, and on the south shore of the Santa Rosa Reservoir (Guadalupe County) in 1995 (New Mexico Department of Game and Fish, 1988; Williams 2001).

Wyoming

Only limited observational information is available in Wyoming. Piping plovers have been reported in the south-eastern and eastern parts of the state, but only one observation of potential breeding has been reported (near Dubois, Wyoming).

EFFECTS OF THE ACTION

Effects from **construction** of wind power projects to the piping plover and its suitable habitat are bulleted below.

- Short-term Direct Effects:
  - Mortality or injury of migrating adults or young from collisions with construction equipment or elevated power lines.
  - Nest abandonment or mortality of young or eggs.
- Short-term Indirect Effects:
  - Changes in food quality and quantity or foraging habitats.
- Long-term Effects:
  - Loss of nesting and foraging habitat.

Effects from **operations** of wind power projects to plovers and their suitable habitat are bulleted below.

- Short-term Direct Effects:
  - Mortality or injury of adults, or young from collisions with wind turbines, associated power lines, or from vehicles or equipment used during maintenance operations.
- Long-term Effects:
  - Avoidance of area due to turbine noise or light disturbance
  - Loss of foraging habitat

CONCLUSIONS

After reviewing the status of the piping plover, the environmental baseline for the action area, effects of the proposed action, and the cumulative effects, it is our biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of the piping plover. There is no designated critical habitat for piping plovers in or near wind energy ROW application areas; therefore, none will be affected. We base our conclusion on the following:

The BMPs and mitigating measures adopted at the programmatic level and applied to projects carried out within this program will effectively minimize the likelihood of any adverse effects to the species.

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LEAST TERN

Unless otherwise noted, the following information is from the "Recovery Plan for the Interior Population of the Least Tern (*Sterna antillarum*)" (U.S. Fish and Wildlife Service 1990a). The interior population of the least tern was federally listed as an endangered species in 1985. Critical habitat has not been designated for the interior population of the least tern.

Species Description

Least terns are the smallest members of the subfamily Sterninae and family Laridae of the order Charadriiformes, measuring only 8 to 9 inches in total length with a wingspan of 19 to 21 inches. The species is characterized by a black crown, white forehead, dark gray wings and back, and black outer primaries. Sexes are alike except for the intensity of bill and leg color, which varies from orange in males to orange/yellows in females, and a slight difference in bill size. Most bills are tipped in black (U.S. Fish and Wildlife Service 1990).

Life History

Least terns nest in colonies from as small as a single pair of birds to 100-plus pairs. Least terns arrive at breeding areas from late April to early June (U.S. Fish and Wildlife Service 2003) and spend 4 to 5 months at their nesting sites. Breeding pairs are monogamous, and courtship and pair formation occur on the breeding grounds. Courtship involves aerial displays, courtship feeding, posturing, parading, and copulation. Courtship occurs at the nest site or at some distance from the nest site (Tomkins 1959) and includes the fish flight, an aerial display involving pursuit and maneuvers culminating in a fish transfer on the ground between the two displaying birds. Other courtship behaviors include nest scraping, copulation, and a variety of postures and vocalizations (U.S. Fish and Wildlife Service 2003).

Least terns nest in areas with similar habitat attributes throughout their North American breeding range. Least terns choose areas with little vegetative cover (Dirks 1990, Ziewitz et al. 1992) and homogenous substrates (Adolf 1998) that are close to stable food sources (Faanes 1983, Dugger 1997, Adolf 1998). Beaches, sand and gravel spoil piles, sandbars, and peninsulas are the principal breeding habitats for all least tern populations or subspecies. The interior least tern also nests along the shores of reservoirs (Chase and Loeffler 1978, Neck and Riskind 1981, Boyd 1987, Schwalbach 1988).

Least terns forage almost exclusively upon small, narrow bodied, schooling fish (Atwood and Kelly 1984, Wilson et al.1993, Schweitzer and Leslie 1996). Least terns are viewed as opportunistic feeders, exploiting any fish within a certain size range. Foraging habitat for least terns includes side channels, sloughs, tributaries, and shallow-water habitats adjacent to sand islands and the main channel. To successfully reproduce, productive foraging habitat must be located within a short distance of a colony (Dugger 1997).

Population Dynamics

During 1997 there were an estimated 5,412 least terns widely scattered across the interior of the United States. An analysis of least tern population data for 1986-1995 for the entire range indicated an overall positive trend. The trend for the entire population was influenced by a relatively strong positive trend on the lower Mississippi River where more than half of the interior least terns nest.

Nesting habitats are ephemeral in quality and abundance, and productivity varies from year to year. Populations could probably be maintained if fledgling success was high periodically (Mertz 1971, Caswell 1982, Kirsch and Sidle 1999). Population trends for least terns are more sensitive to variations in adult and fledged juvenile survival than to fledging success (Kirsch 1992). However, statistics on adult and fledged juvenile survival are not known (Kirsch and Sidle 1999).

Status and Distribution

The interior least tern historically bred along the Mississippi, Red, and Rio Grande River systems and Rivers of central Texas. The breeding range extended from Texas to Montana and from eastern Colorado and New Mexico to southern Indiana. Formerly distributed throughout riverine habitats across the Great Plains, the least tern now occupies scattered remnants of its former range. Where they still occupy riverine breeding habitat of the Missouri River and its tributaries, the Arkansas and Red River systems, and the Rio Grande, they are generally limited to segments that are not affected by impoundments or channelization.

The winter distribution of the least tern is poorly understood (Thompson et al. 1997). On the Pacific coast, they have been observed in coastal areas of southern Mexico. On the Atlantic coast, they are regularly observed wintering along the eastern coast of Mexico, Central, and South America as far south as Argentina.

Of the potential wind energy development sites identified, least terns are found in various locations in Montana, Wyoming, Colorado and New Mexico – although information is limited. In Colorado, least terns are found in the south-eastern plains of the Arkansas River Valley. The species nests on barren beaches adjacent to large prairie reservoirs, almost totally restricted to Bent and Kiowa Counties. Migrants or nonbreeding individuals also occur at other reservoirs on the southeastern plains. Casual nonbreeding summer visitors may occur on the northeastern plains and casual to very rare spring and fall migrants may occur on the northeastern plains. Over the past 14 years, least terns have been located at the following Colorado reservoirs (Nelson 2003a): John Martin Reservoir, Adobe Creek Reservoir, Great Plains Reservoir System (Neenoshe, Neegronde, Neesopah, and Neeskah Reservoirs), Upper Queen Reservoir, Timber Lake, and Veehof Reservoir.

In 1990, the Colorado Division of Wildlife (CDOW) contracted with the Colorado Bird Observatory to conduct tern inventories and habitat surveys on the eastern plains of Colorado (CDOW 1994). Additional surveys were conducted in the South Platte River drainage in northeast Colorado and the Arkansas River drainage in southeast Colorado; however, nesting activity was found only in southeast Colorado.

In 2003, 33 least tern nests were found in Colorado (Nelson 2003b), representing a minimum of 24 nesting pairs. The nests were located at John Martin Reservoir, Alkali Island in Neenoshe Reservoir, and Adobe Creek Reservoir. In the past five years, the number of nesting terns at John Martin Reservoir has surpassed the number at all other reservoirs (Nelson 2003a) and that reservoir has taken on increasing importance to the recovery of the species.

Changes in the availability of water and exposure of sand bar flats vary from year to year (given annual variations in water availability and allocations), as does the availability of nesting habitat. Consequently, use varies among the various reservoirs.

There have only been a few observations of least terns in the south-eastern and eastern parts of Wyoming, but there is no indication of breeding. In all likelihood, least terns are simply migrating through the state. We currently have no recent information for Montana and New Mexico.

Threats

Historically, the least tern was hunted for the commercial use of its feathers to decorate ladies' hats (U.S. Fish and Wildlife Service 1990b). Since the early 1900s, habitat alteration and destruction in the form of river channelization and the construction of reservoirs for hydropower, flood control, and irrigation has had detrimental effects on the species' habitat. Channelization, irrigation, and the construction of reservoirs also have contributed to the elimination of much of the riverine nesting habitat for least terns.

Storage and supply of water for irrigation, power generation, and navigation has altered the natural hydrograph to which the interior least tern's breeding season was historically adapted. High flow periods may now extend into the normal nesting period, thereby reducing the quality of existing nest sites and forcing interior least terns to initiate nests in poor quality locations. Extreme fluctuations can flood existing nests, inundate potential nesting areas, or dewater feeding areas.

STATUS OF THE SPECIES WITHIN THE ACTION AREA

Within the action area, least terns may occur on BLM lands in Colorado, Montana, New Mexico, and Wyoming. Interior populations of the least tern nest on sand bars, islands, and sparsely vegetated beaches associated with major rivers and lakes. Migration tends to follow major river systems but inland populations will migrate cross country (Thompson *et. al*. 1997). Water diversion and altered hydrology are primary factors contributing to habitat loss for inland populations.

Colorado

Much of the information for Colorado was taken from the *Colorado Bureau of Land Management Statewide Programmatic Biological Assessment for Least Tern (Sterna antillarum), Piping Plover (Charadrius melodus), Preble's Meadow Jumping Mouse (Zapus hudsonius preblei), and Arkansas Darter (Etheostoma cragini)* (Real West Natural Resource Consulting 2004).

Approximately 3,200 acres of BLM surface ownership includes least tern foraging and nesting habitat in Colorado. These areas include land under and surrounding Neegronde, Neenoshe, Neeskah, and Neesopah reservoirs. In the case of BLM lands, foraging and nesting acreages are the same because the areas serve both as nesting and foraging sites. Total habitat in the entire Royal Gorge Resource Area, regardless of landownership, is 15,811 acres for nesting and 20,621 acres for foraging. Nesting habitat also serves as foraging habitat, but the reverse is not necessarily true; some habitat is used for foraging but is not suitable for nesting.

Neenoshe Reservoir is the largest of the reservoirs with 3,696 surface acres. Warm water fish species are present at all four reservoirs and facilities for anglers are available at the three reservoirs, Neenoshe, Neegronde, and Neesopah, which also have public access.

The nesting and foraging areas around Adobe Creek Reservoir and John Martin Reservoir do not include BLM lands. There are no federal mineral estate lands, either with or without BLM surface ownership, on any foraging or nesting habitat.

Montana

Breeding terns were more recently recorded on the Yellowstone River and on the Missouri River between Fort Peck Reservoir and North Dakota (U.S. Fish and Wildlife Service 1990a). A few least terns have been recorded on islands and shoreline within the Fort Peck Reservoir (Charles M. Russell National Wildlife Refuge). These locations are the westernmost nesting sites of the interior least tern.

New Mexico

In New Mexico, interior least terns were first recorded nesting at Bitter Lake National Wildlife

Refuge in 1949, and terns have nested there annually since then. Population counts over the period have been variable, ranging as high as 60 birds in 1961, but typically 20 to 30 individuals during a breeding season. In 1986 and 1987, interior least terns nested off the refuge on City of Roswell property, but these were impacted by human disturbance, nest flooding, environmental contamination, and predation (New Mexico Department of Game and Fish, 1988, 1994). Interior least terns may also breed occasionally at Bottomless Lake State Park and Wade's Bog (Hubbard 1985). The species occurs in migration in Eddy County and as a vagrant elsewhere, including Espanola, Sumner Lake (DeBaca County), Bosque del Apache National Wildlife Refuge (Socorro County), and near Glenwood, Las Cruces, and Alamogordo.

On June 9, 2004, 5 pairs of interior least terns were observed in a backwater area of Brantley Reservoir on the Pecos River (Eddy County, New Mexico). The nearest documented nesting was at Bitter Lake National Wildlife Refuge, 60 miles north of Brantley Reservoir. It is unknown whether interior least terns have used areas around Brantley Reservoir for nesting in previously years. At least 14 adults were observed, with an estimated 7 nests on the lakeshore. Six juvenile terns were observed near the nesting area in late August (Bureau of Reclamation 2005; J. Montgomery, Ph.D., Fish and Wildlife Service permittee, electronic mail message, August 23, 2004).

In 2005, interior least terns did not nest at Brantley Reservoir, possibly due to potential nesting areas being under water, vegetated, or impacted by human disturbance. However, approximately six to eight adults, and up to five immature (one-year-old) terns occupied the reservoir area until August. The 2005 nesting season was the most successful year at Bitter Lake National Wildlife Refuge since the mid-1980s (when observers began monitoring nesting on a regular basis) and probably back to 1937 when the refuge was established. Fourteen pairs fledged 23 juveniles (J. Montgomery, Ph.D., Fish and Wildlife Service permittee, electronic mail message, September 7, 2005).

Wyoming

Only limited observational information is available in Wyoming. Least terns have been reported in the southeastern and eastern parts of the state, and no breeding information is available.

EFFECTS OF THE ACTION

The following effects are considered likely:

Effects from construction of wind power projects to the least tern and its suitable habitat are bulleted below.

- Short-term Direct Effects:
    - Mortality or injury of migrating adults or young from collisions with construction equipment or elevated power lines.
    - Nest abandonment or mortality of young or eggs.

- Short-term Indirect Effects:
  - Changes in food quality and quantity or foraging habitats.
- Long-term Effects:
  - Loss of nesting and foraging habitat.

Effects from operations of wind power projects to terns and their suitable habitat are bulleted below.

- Short-term Direct Effects:
  - Mortality or injury of adults, or young from collisions with wind turbines, associated power lines, or from vehicles or equipment used during maintenance operations.
- Long-term Effects:
  - Avoidance of area due to turbine noise or light disturbance
  - Loss of foraging habitat

CONCLUSIONS

After reviewing the status of the species, the environmental baseline for the action area, effects of the proposed action, and the cumulative effects, it is our biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of the least tern. There is no designated critical habitat for least terns; therefore, none will be affected. We base our conclusion on the following:

The BMPs and mitigating measures adopted at the programmatic level and applied to projects carried out within this program will effectively minimize the likelihood of any adverse effects to the species.

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GRIZZLY BEAR

STATUS OF THE SPECIES

The grizzly bear (*Ursus arctos horribilis*) originally inhabited a variety of habitats from the Great Plains to mountainous areas throughout western North America, from central Mexico to the Arctic Ocean. With the advent of Euroamerican colonization in the early nineteenth century, grizzly bear numbers were reduced from over 50,000 to less than 1,000 in North America south of the Canadian border. Today, the grizzly bear occupies less than two percent of its former range south of Canada (U.S. Fish and Wildlife Service 1993). In the conterminous 48 States, only five remaining areas have either remnant or self-perpetuating populations. These remaining populations are principally located in mountainous regions in Washington, Idaho, Wyoming, and Montana and are often associated with National Parks and wilderness areas.

The Grizzly Bear Recovery Plan was approved January 1982 and was revised and approved on September 10, 1993 (Recovery Plan) (U.S. Fish and Wildlife Service 1993). The Recovery Plan details recovery objectives and strategies for the grizzly bear recovery zones in the ecosystems where grizzly bear populations persist. These recovery zones are the Northern Continental Divide Ecosystem (NCDE), Yellowstone Grizzly Bear Ecosystem (YGBE), Cabinet-Yaak Ecosystem (CYE), and Selkirk Ecosystem (SE). The Recovery Plan also includes recovery strategies for the North Cascades ecosystem in Washington, where only a very few bears are believed to remain, and for the Selway-Bitterroot ecosystem of Idaho and Montana where suitable grizzly bear habitat still occurs.

Species Description

Grizzly bears are among the largest terrestrial mammals in North America. South of the United States (U.S.)-Canada border, adult females range from 250 to 350 pounds and adult males range from 400 to 600 pounds. Grizzly bears are relatively long-lived, living 25 years or longer in the wild. Grizzly bears are omnivorous, opportunistic feeders that require foods rich in protein or carbohydrates in excess of maintenance requirements in order to survive seasonal pre-and post-denning requirements. Grizzly bears are homeo-hypothermic hibernators, meaning their body temperature drops no more than 5 degrees Centigrade during winter when deep snow, low food availability, and low ambient air temperatures appear to make winter sleep essential to grizzly bears' survival (Craighead and Craighead 1972a,b). Grizzly bears excavate dens and require environments well-covered with a blanket of snow for up to five months, generally beginning in fall (September to November) and extending until spring (March to April) (Craighead and Craighead 1972b; Pearson 1975).

Listing history

The grizzly bear was federally listed as a threatened species in the lower 48 states on July 28, 1975 (40 FR 31736). The Service identified the following as factors establishing the need to list: (1) present or threatened destruction, modification, or curtailment of habitat or range; (2) overutilization for commercial, sporting, scientific, or educational purposes; and (3) other manmade factors affecting its continued existence. The two primary challenges in grizzly bear conservation are the reduction of human-caused mortality and the conservation of remaining habitat (U.S. Fish and Wildlife Service 1993). On November 17, 2005, the Service proposed to recognize grizzly bears in the Yellowstone region as a distinct population segment (DPS) and to remove this DPS from the list of threatened wildlife.

Life history

The search for energy-rich food appears to be a driving force in grizzly bear behavior, habitat selection, intraspecific and interspecific interactions. Grizzly bears historically used a wide variety of habitats across the North America, from open to forested, temperate through alpine and arctic habitats, once occurring as far south as Mexico. They are highly dependent upon learned food locations within their home ranges. Adequate nutritional quality and quantity are important factors for successful reproduction. Diverse structural stages that support wide varieties of nourishing plants and animals are necessary for meeting the high energy demands of these large animals. Grizzly bears follow phenological vegetative, tuber or fruit development, seek out concentrated food sources including carrion, live prey (fish, mammals, insects), and are easily attracted to human food sources including gardens, grain, compost, bird seed, livestock, game gut piles, bait, and garbage. Bears that lose their natural fear and avoidance of humans, usually as a result of food rewards, become habituated and may become food-conditioned. Grizzly bears defend food and have been known to charge when surprised. Both habituation and food conditioning increase chances of human-caused grizzly bear mortality as a result of real or perceived threats to human safety or property. Nuisance grizzly bear mortalities can be a result of legal management actions, defense of human life or illegal killing.

Adult grizzly bears are individualistic and normally solitary, except females with cubs or during short breeding relationships. They tolerate other grizzly bears at closer distances when food sources are concentrated and siblings may associate for several years following weaning (Murie 1944, 1962; Jonkel and Cowan 1971; Craighead 1976; Egbert and Stokes 1976; Glenn et al. 1976; Herrero 1978). Across their range, home range sizes vary from about 50 square miles or more for females to a few hundred square miles for males. Overlap of home ranges is common. Grizzly bears may have one of the lowest reproductive rates among terrestrial mammals, resulting primarily from the late age at first reproduction, small average litter size, and the long interval between litters. Mating occurs from late May through mid-July. Females in estrus accept more than one adult male (Hornocker 1962), and can produce cubs from different fathers the same year (Craighead et al. 1995). Age of first reproduction and litter size may be nutritionally related (Herrero 1978; Russell et al. 1978). Average age at first reproduction in the lower 48 states for females is 5.5 years and litter size ranges from one to four cubs who stay with the mother up to two years. Males may reach physiological reproductive age at 4.5 years, but may not be behaviorally reproductive due to other dominant males preventing mating.

Natural mortality is known to occur from intra-specific predation, but the degree this occurs in natural populations is not known. Parasites and disease do not appear to be a significant cause of natural mortality (Jonkel and Cowan 1971; Kistchinskii 1972; Mundy and Flook 1973; Rogers and Rogers 1976). As animals highly dependent upon learned habitat, displacement into unknown territory (such as subadult dispersal) may lead to submarginal nutrition, reduced reproduction or greater exposure to adult predatory bears or human food sources (which can lead to human-caused mortality). Starvation and mortality in dens during food shortages have been surmised, but have not been documented as a major mortality factor. Natural mortality in rare, relatively secretive animals such as grizzlies can be extremely difficult to document or quantify.

Human-caused mortality has been slightly better quantified, but recent models speculate that reported mortality may only be up to 50 percent of actual mortality (McLellan et al. 1999). Between 1800 and 1975, grizzly populations in the lower 48 states had declined drastically. Fur trapping, mining, ranching and farming pushed westward, altered habitat and resulted in the direct killing of grizzly bears. Historically, grizzly bears were targeted in predator control programs in the 1930's. Predator control was probably responsible for extirpation in many states that no longer support grizzlies. More recent human-caused mortality in Montana includes legal hunting (canceled since 1991), management control actions, defense of life, vehicle and train collisions, defense of property, mistaken identity by black bear or other big game hunters, poaching and malicious killing. Grizzly bears normally avoid people, possibly as a result of many generations of bear sport hunting and human-caused mortality. Displacement away from human activities has been documented to reduce fitness of grizzly bears, affecting survival in some instances. Avoidance of roads can lead grizzly bears to either avoid essential habitat along roads, or could put them at greater risk of exposure to human-caused mortality if they do not avoid roads.

Status and Distribution

Status of grizzly bears in the YGBE. The 9,209 square mile YGBE recovery zone includes portions of Wyoming, Montana, and Idaho and portions of six National Forests (Beaverhead, Bridger-Teton, Custer, Gallatin, Shoshone, and Targhee), Yellowstone and Grand Teton National Parks, John D. Rockefeller Memorial Parkway, portions of adjacent private and State lands, and lands managed by the Bureau of Land Management.

All recovery parameters for the recovery zone were met in 2003 (Schwartz and Haroldson 2004). Recovery parameters had been met for at least the last 5 years through 2003. The mortality threshold of 5.2 for female bears was slightly exceeded in 2004 with a 6-year running average of 6 human-caused female mortalities (Haroldson and Frey 2005). There were a total of 26 documented grizzly bear mortalities in 2004, of which 19 were known human-caused deaths, five were natural mortalities and two were of undetermined causes (Haroldson and Frey 2005). All other recovery parameters were met in 2004 (Schwartz et al. 2005). The number of females with cubs has surpassed the recovery criterion for a number of years (Haroldson 2005) and bears now occur where they have not been reported for many years. A total of 49 unduplicated females with 96 cubs were documented in the Greater Yellowstone Ecosystem in 2004 (Haroldson 2005). With this, the 6-year running average of females with cubs within the Recovery Zone and a 10-mile perimeter has gradually increased from 15 in 1986 to 40 in 2004. The mean litter size of two in 2004 was consistent with past years (Haroldson 2005).

As in other ecosystems, the exact size of the grizzly bear population in the YGBE is not known. The nature of the species and the rugged terrain it inhabits makes complete population census difficult. Haroldson and Frey (2005) calculated a minimum population estimate within the recovery zone and a 10-mile perimeter of 431 grizzly bears. Population parameters more readily monitored are used as an alternative index to population size (Knight and Eberhardt 1984). Eberhardt and Knight (1996) used several estimators and calculated a minimum total population size of 245 bears, an estimated population size of 390 bears using marked females, and an estimated population size of 344 bears using distinct family groups. Population parameters for the YGBE recovery zone were summarized as follows in the Final Conservation Strategy for the Grizzly Bear in the Yellowstone Area (Interagency Conservation Strategy Team 2003, page 20): "From the mid 1980s, the Yellowstone grizzly population has grown at approximately three to four percent or more per year (Eberhardt et al. 1994, Boyce 1995, Boyce et al. 2001). Boyce (1995) has calculated that the Yellowstone population has a probability of extinction of 0.0004 (4/10,000) – a very low probability. Nevertheless, as Boyce points out, 'Population size alone is not a sufficient criterion for evaluating population viability' and 'even though a population may have increased or decreased over the past 10 to 20 years, this offers no indication that the population will continue on the same trajectory in the future.'" Schwartz et al. (2005) built an array of models for examining trend, and estimated that the population was increasing within the recovery zone, and decreasing outside the recovery zone, suggesting a source-sink dynamic.

The best available information suggests the YGBE grizzly bear population is stable and is likely increasing. The long term conservation of the population continues to depend largely on managing bear-human conflict, which often results in human-caused mortality of grizzly bears. Years in which natural grizzly bear food production and availability are high can result in younger age classes of grizzly bears accustomed to fairly good food availability. A year of drought and poor food production can compel grizzly bears to search widely for food. Such wide ranging movements can bring grizzly bears into closer contact with humans, increasing bear-human conflicts and resultant control/management actions.

Status of grizzly bear populations in CYE and SE. The Cabinet/Yaak Ecosystem in northwestern Montana and northeastern Idaho has over 1,900 square miles of forested and mountainous habitat occupied by grizzly bears. The population in the Cabinet Mountains portion of this area is thought to be less than 15 bears. The Yaak section of the CYE currently supports a minimum of approximately 20 bears. The Yaak population estimate does not include credible reports from the public of grizzly bear observations, which suggest a population estimate of 20 to 30 bears in the Yaak section of the CYE would be conservative (Kasworm et al. 2000). There are grizzly bears to the north of the U.S.-Canada border, and interchanges of radio-collared bears across the border have been documented (U.S. Fish and Wildlife Service 1993).

The Selkirk Ecosystem of northwestern Idaho, northeastern Washington, and southeastern British Columbia includes about 1,080 square miles in the U.S. portion and about 875 square miles in the Canadian portion of the recovery zone. The Selkirk recovery zone is the only defined grizzly bear recovery zone that includes part of Canada because the habitat in the U.S. portion is not of sufficient size to support a minimum population. The habitat is contiguous across the border and radio-collared bears are known to move back and forth across the border. Therefore, the grizzly bears north and south of the border are considered one population (U.S. Fish and Wildlife Service 1993).

Neither the CYE nor the SE grizzly bear populations have ever attained the Recovery Plan criteria for females with cubs. Population trend information is statistically inconclusive, though the point estimate of the rate of increase declined during 1999 to 2004 (Kasworm et al. 2000, Kasworm 2001, Kasworm et al. 2004) in the CYE. The Service determined that the combined SE-CYE grizzly bear recovery zones were warranted endangered but precluded in 1999 and suggested that the two populations might be inter-connected (FR 26725-26733).

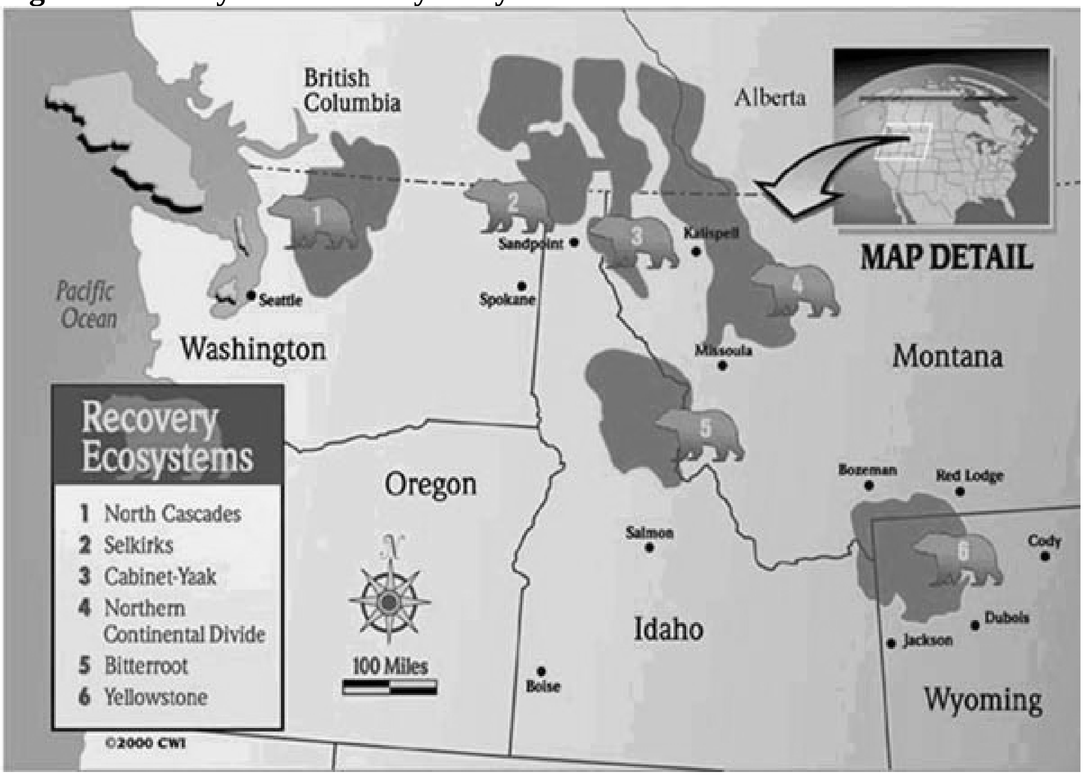
The most recent data indicate that population status is below recovery goals in the CYE for the distribution of females with young in bear management subunits and exceeds the 6-year average of female mortality in the recovery zone (U.S. Fish and Wildlife Service 2004a).

Status of the Selway-Bitterroot and North Cascades ecosystems. Grizzly bear recovery efforts in the Selway-Bitterroot Ecosystem and North Cascades Ecosystem are in the planning stages. In the North Cascades Ecosystem, most of the grizzly bear population occurs north of the Canada-U.S. border, but a few grizzly bears persist south of the border. Though suitable habitat remains, grizzly bears were extirpated from the Selway-Bitterroot Ecosystem decades ago. An environmental impact statement and decision notice addressing the impacts of reintroducing grizzly bears into the Bitterroot Ecosystem in east central Idaho was released in 2000 (65 FR 69623-69643).

Status of grizzly bears in the NCDE. The NCDE extends from the Rocky Mountains of northern Montana into contiguous areas in Alberta and British Columbia, Canada (Figure 1).

The U.S. portion of the NCDE which makes up the NCDE recovery zone (U.S. Fish and Wildlife Service 1993) encompasses over 9,600 square miles and includes parts of five National Forests (Flathead, Kootenai, Helena, Lewis and Clark, and Lolo), four wilderness areas (Bob Marshall, Mission Mountains, Great Bear, and Scapegoat) and one wilderness study area (Deep Creek North) (Figure 2). National Forest System lands encompass 63 percent of the NCDE recovery zone. Additionally, the NCDE recovery zone includes Glacier National Park (GNP), the Flathead Indian Reservation (Salish-Kootenai tribal land), the Blackfeet Indian Reservation, adjacent private and State lands, and lands managed by the Bureau of Land Management.

Figure 1. Grizzly bear recovery ecosystems



Source: Interagency Grizzly Bear Committee http://www.fs.fed.us/r1/wildlife/igbc/

A major issue in grizzly bear recovery is sanitation related human-caused grizzly bear mortality. Towns and settlements are common in low elevations and major valley bottoms within and adjacent to recovery zones. Human generated food sources such as bird feeders, garbage, pet and livestock foods, human foods, gardens, and orchards present powerful attractants for grizzly bears. Grizzly bears attracted to these human-generated food sources become habituated and food conditioned. Such bears often become a threat to human safety and property and are killed illegally or removed through agency nuisance grizzly bear control actions.

Special food storage orders for National Forest lands within the recovery zones have been issued by the Forest Service to reduce the possibility of grizzly bear habituation to human-related food sources.

Sanitation related grizzly bear deaths are among the leading causes of grizzly bear mortality in the NCDE (U.S. Fish and Wildlife 2003a). Data collected since 1980 (Chris Servheen, U.S. Fish and Wildlife Service 2004, in. litt.) (Table 6) demonstrate human site conflicts which include food habituation and garbage resulted in 15.5 percent of total grizzly bear mortality within the NCDE recovery zone.

This figure increases to 22 percent with the addition of grizzly bear mortality resulting from livestock depredation. Illegal and malicious killing of grizzly bears is the second leading cause of death at 13.5 percent. Legal hunting of grizzly bears is the only activity that exceeds human site conflicts as a source of grizzly bear mortality. Legal hunting of grizzly bears ended in Montana in 1991.

Table 6. Causes of known grizzly bear mortalities in the NCDE 1980 to 2002. (Chris Servheen, U.S. Fish and Wildlife Service, in litt. 2004)

Category	Number of Mortalities	Percent of Total Mortalities
Capture	10	2.8
Car	13	3.7
Human fatality	11	3.1
Human site conflict	55	**15.5**
Legal grizzly hunt	81	**22.8**
Livestock depredation	22	**6.2**
Illegal/malicious	48	**13.5**
Mistaken identity	29	8.2
Natural	15	4.2
Self-defense	23	6.5
Trains	29	8.2
Under investigation	8	2.3
Unknown	11	3.1
TOTAL	355	-
DEATHS per YEAR	15.43	-
DEATHS per YEAR without legal hunting	11.91	-

The Recovery Plan states that each individual ecosystem will remain listed until its specific recovery criteria are met. Recovery criteria include a minimum number of females with cubs seen annually, distribution of family groups throughout the recovery zone, and a limit on human-caused mortality. Demographic recovery outlined for the recovery zones includes the following criteria:

NCDE:

- observation of 22 females with cubs of the year (unduplicated sightings), 10 in Glacier National Park and 12 outside the park, over a 6-year average both inside the recovery area and within 10 mile area immediately surrounding the recovery zone, excluding Canada

- twenty-one of the 23 BMUs occupied by females with young from a running 6-year sum of verified observations, and with no two adjacent BMUs unoccupied

- known, human-caused mortality not to exceed four percent of the current population estimate (based on most recent 3-year average of females with young)
- no more than 30 percent of the known, human-caused mortality shall be females
- the mortality limits cannot be exceeded in more than 2 consecutive years for recovery to be achieved
- recovery in the NCDE cannot be achieved without occupancy of the Mission Mountains portion of the NCDE

YGBE:

- Observation of 15 females with cubs over a running 6-year average both inside the recovery zone and within a 10 mile area immediately surrounding the recovery zone
- 16 of 18 BMUs occupied by females with young from a running 6-year sum of verified observations, and with no two adjacent BMUs unoccupied
- Known, human-caused mortality not to exceed 4 percent of population estimate based on the most recent 3-year sum of females with cubs.
- No more than 30 percent of the known, human-caused mortality shall be females
- The mortality limits cannot be exceeded in more than 2 consecutive years for recovery to be achieved

CYE:

- Six females with cubs over a rolling 6-year average both inside the recovery zone and within a 10 mile area immediately surrounding the recovery zone, excluding Canada
- 18 of 22 BMUs occupied by females with young from a running 6-year sum of verified observations
- Known human caused mortality not to exceed 4 percent of the population estimate based on the most recent 3-year sum of females with cubs
- No more than 30 percent of the known human-caused mortality shall be females
- The mortality limits cannot be exceeded during any 2 consecutive years for recovery to be achieved
- Presently, grizzly numbers are so small in this ecosystem that the mortality goal shall be zero known human-caused mortalities

SE:

- Six females with cubs over a running 6-year average both inside the recovery zone and within a 10 mile area immediately surrounding the recovery zone, including Cananda
- 7 of 10 BMUs on the U.S. side occupied by females with young from a running 6-year sum of verified sightings

- Human caused mortality not to exceed 4 percent of the population estimate based on the most recent 3-year sum of females with cubs
- No more than 30 percent of human caused mortality shall be females
- The mortality limits cannot be exceeded during any 2 consecutive years for recovery to be achieved
- Presently, grizzly numbers are so small in this ecosystem that the mortality goal shall be zero known human-caused mortalities

The mortality of grizzly bears is counted towards recovery zone statistics if mortality occurs within a 10-mile area outside the recovery zone boundary. This is a conservative accounting for grizzly bears making their range primarily in the recovery zone, but it includes bears whose range overlaps the recovery zone line.

The purpose of counting females with cubs is to estimate a known minimum number of adult females to demonstrate sufficient reproduction to offset existing levels of mortality (U.S. Fish and Wildlife Service 1993). Years during which the effort to count female grizzly bears is poor or conditions are unfavorable may yield very conservative counts of females with cubs. These conservative counts result in a conservative minimum population estimate, which results in conservative human-caused mortality limits. Due to the varying effort and success in counting females with cubs, neither these annual number of females with cubs counted (Knight and Blanchard 1993) or the human-caused mortality limits/annual tally can be used to estimate trend (U.S. Fish and Wildlife Service 1993).

EFFECTS OF THE ACTION

The following effects are considered likely:

Effects from construction of wind power to grizzly bears and suitable habitat are bulleted below.

- Short-term direct effects of construction phase:
    - Visual or auditory disturbance or displacement of individuals from low-flying aircraft, vehicles, heavy equipment, and humans during construction and maintenance actions

- Short-term indirect effects of construction phase:
    - Changes in food or prey quality and quantity or foraging habitats

- Long-term effects of construction phase:
    - Habituation of individual bears to human food sources associated with improperly stored human or livestock food, garbage, or road-kills resulting in the need for future management actions

Effects from operations of wind power projects to grizzly bears and their suitable habitat are bulleted below.

- Short term direct effects of operation phase:
    - Visual or auditory disturbance or displacement of individuals from vehicles and humans during maintenance actions

- Short term indirect effects of operation phase:
    - Changes in food or prey quality and quantity or foraging habitats

- Long-term effects of operation phase:
    - Habituation of individual bears to human food sources associated with improperly stored human or livestock food, garbage, or road kills, resulting in the need for future management actions
    - Habituation of individual bears to feeding on birds or bats killed or injured by electrocution or collision with infrastructure, resulting in the need for future management actions
    - Long-term loss of habitat due to displacement of bears from suitable habitat adjacent to project roads due to disturbance from vehicles and human presence

CONCLUSIONS

After reviewing the status of grizzly bears, the environmental baseline for the action area, effects of the proposed action, and the cumulative effects, it is our biological opinion that the action, as proposed, is not likely to jeopardize the continued existence of grizzly bears. No critical habitat has been designated for this species; therefore, none will be affected. We base our conclusion on the following:

The BMPs and mitigating measures adopted at the programmatic level and applied to projects carried out within this program will effectively minimize the likelihood of any adverse effects to the species.

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DESERT TORTOISE (MOJAVE POPULATION)

Status of the Mojave Desert Tortoise Rangewide

The desert tortoise (*Gopherus agassizii*) is a large, herbivorous reptile found in portions of California, Arizona, Nevada, and Utah. It also occurs in Sonora and Sinaloa, Mexico. The Mojave population of the desert tortoise includes those animals living north and west of the Colorado River in the Mojave Desert of California, Nevada, Arizona, southwestern Utah, and in the Sonoran Desert in California. Desert tortoises reach 8 to 15 inches in carapace length. Adults have a domed carapace and relatively flat, unhinged plastron. Shell color is brownish, with yellow to tan scute centers. The forelimbs are flattened and adapted for digging and burrowing. Optimal habitat has been characterized as creosote bush scrub in which annual precipitation ranges from 2 to 8 inches, where the diversity of perennial plants is relatively high, and production of ephemerals is high (Luckenbach 1982; Turner 1982; Turner and Brown 1982). Soils must be friable enough for digging of burrows, but firm enough so that burrows do not collapse. Desert tortoises occur from below sea level to an elevation of 7,300 feet, but the most favorable habitat occurs at elevations of approximately 1,000 to 3,000 feet (Luckenbach 1982).

Desert tortoises are most active during the spring and early summer when annual plants are most common. Additional activity occurs during warmer fall months and occasionally after summer rainstorms. Desert tortoises spend the remainder of the year in burrows, escaping the extreme conditions of the desert. In Nevada and Arizona, tortoises are considered to be active from approximately March 15 through October 15.

Desert tortoises are most commonly found within the desert scrub vegetation type, primarily in creosote bush scrub. In addition, they occur in succulent scrub, cheesebush scrub, blackbrush scrub, hopsage scrub, shadscale scrub, microphyll woodland, Mojave saltbush-allscale scrub, and scrub-steppe vegetation types of the desert and semidesert grassland complex (U.S. Fish and Wildlife Service 1994). Within these vegetation types, desert tortoises potentially can survive and reproduce where their basic habitat requirements are met. These requirements include a sufficient amount and quality of forage species; shelter sites for protection from predators and environmental extremes; suitable substrates for burrowing, nesting, and overwintering; various plants for shelter; and adequate area for movement, dispersal, and gene flow. Throughout most of the Mojave Region, tortoises occur most commonly on gently sloping terrain with soils ranging from sandy-gravel and with scattered shrubs, and where there is abundant inter-shrub space for growth of herbaceous plants. Throughout their range, however, tortoises can be found in steeper, rockier areas.

The size of desert tortoise home ranges varies with respect to location and year. Females have long-term home ranges that are approximately half that of the average male, which range from 25 to 200 acres (Berry 1986). Over its lifetime, each desert tortoise may require more than 1.5 square miles of habitat and make forays of more than 7 miles at a time (Berry 1986). In drought years, the ability of tortoises to drink while surface water is available following rains may be crucial for tortoise survival. During droughts, tortoises forage over larger areas, increasing the likelihood of encounters with sources of injury or mortality including humans and other predators. Further information on the range, biology, and ecology of the desert tortoise can be found in Berry and Burge (1984), Burge (1978), Burge and Bradley (1976), Bury *et al*. (1994), Germano *et al*. (1994), Hovik and Hardenbrook (1989), Karl (1981, 1983a, 1983b), Luckenbach (1982), U. S. Fish and Wildlife Service 1994), and Weinstein *et al*. (1987).

On August 4, 1989, the Service published an emergency rule listing the Mojave population of the desert tortoise as endangered (54 FR 42270). On April 2, 1990, the Service determined the Mojave population of the desert tortoise to be threatened (55 FR 12178). Reasons for the determination included significant population declines, loss of habitat from construction projects such as roads, housing and energy developments, and conversion of native habitat to agriculture. Grazing and off-highway-vehicle (OHV) activity have degraded additional habitat. Also cited as threatening the desert tortoise's continuing existence were illegal collection by humans for pets or consumption, upper respiratory tract disease (URTD), predation on juvenile desert tortoises by common ravens (*Corvus corax*), coyotes (*Canis latrans*), and kit foxes (*Vulpes macrotis*), fire, and collisions with vehicles on paved and unpaved roads.

On June 28, 1994, the Service approved the final Desert Tortoise (Mojave Population) Recovery Plan (Recovery Plan) (U.S. Fish and Wildlife Service 1994). The Recovery Plan divides the range of the desert tortoise into 6 recovery units and recommends establishment of 14 Desert Wildlife Management Areas (DWMAs) throughout the recovery units. Within each DWMA, the Recovery Plan recommends implementation of reserve-level protection of desert tortoise populations and habitat, while maintaining and protecting other sensitive species and ecosystem functions. The design of DWMAs should follow accepted concepts of reserve design. As part of the actions needed to accomplish recovery, the Recovery Plan recommends that land management within all DWMAs should restrict human activities that negatively impact desert tortoises (U.S. Fish and Wildlife Service 1994). The DWMAs/Areas of Critical Environmental Concern (ACECs) have been designated by the Bureau of Land Management (BLM) through development or modification of their land-use plans in Arizona, Nevada, Utah, and parts of California.

The California Desert Conservation Area Plan (BLM 1980) is the primary plan that guides the overall management of desert tortoise habitat in California. Land-use planning activities are underway in California to complete designation of DWMAs/(ACECs). Desert tortoise habitat management in Arizona is covered primarily by the Mojave Amendment to BLM's Arizona Strip Resource Management Plan, which was prepared to implement the Recovery Plan. BLM's Arizona Strip Field Office designated 167,065 acres of desert tortoise habitat as ACECs. In Nevada, BLM's Las Vegas, Ely, and Battle Mountain field offices manage desert tortoise habitat; 941,800 acres of desert tortoise habitat were designated as ACECs by the Las Vegas and Ely field offices. No desert tortoise critical habitat or proposed ACECs occur within the jurisdiction of the Battle Mountain Field Office. The regulation of activities within critical habitat through section 7 consultation is based on recommendations in the Recovery Plan (U.S. Fish and Wildlife Service 1994).

Long-term monitoring of desert tortoise populations is a high priority recovery task as identified in the Recovery Plan. From 1995 to 1998, pilot field studies and workshops were conducted to develop a monitoring program for desert tortoise. In 1998, the Desert Tortoise Management Oversight Group identified line distance sampling as the appropriate method to determine rangewide desert tortoise population densities and trends. Monitoring of populations using this method is underway across the range of the desert tortoise. Successful rangewide monitoring will enable managers to evaluate the overall effectiveness of recovery actions and population responses to these actions, thus guiding recovery of the Mojave desert tortoise. Rangewide tortoise population monitoring began in 2001 and is conducted annually.

Changing ecological condition as a result of natural events or human-caused activities may stress individuals and result in a more severe clinical expression of URTD (Brown *et al*. 2002). For example, the proliferation of non-native plants within the range of the tortoise has had far-reaching impacts on tortoise populations. Tortoises have been found to prefer native vegetation over non-natives (Tracy *et al*. 2004). Non-native annual plants in desert tortoise critical habitat in the western Mojave Desert were found to compose over 60 percent of the annual biomass (Brooks 1998). The reduction in quantity and quality of forage may stress tortoises and make them more susceptible to drought- and disease-related mortality (Brown *et al*. 1994). Malnutrition has been associated with several disease outbreaks in both humans and turtles (Borysenko and Lewis 1979). What is currently known with certainty about disease in the desert tortoise relates entirely to individual tortoises and not populations; virtually nothing is known about the demographic consequences of disease (Tracy *et al*. 2004).

<u>a. Desert Tortoise Recovery Plan Assessment and Recommendations</u>

The General Accounting Office (GAO) Report, *Endangered Species: Research Strategy and Long-Term Monitoring Needed for the Mojave Desert Tortoise Recovery Program* (GAO 2002), directed the Service to periodically reassess the Recovery Plan to determine whether scientific information developed since its publication could alter implementation actions or allay some of the uncertainties about its recommendations. In response to the GAO report, the Service initiated a review of the existing Recovery Plan in 2003.

In March 2003, the Service impaneled the Desert Tortoise Recovery Plan Assessment Committee (Committee) to assess the Recovery Plan. The Committee was selected to represent several important characteristics with particular emphasis on commitment to solid science. The charge to the Committee was to review the entire Recovery Plan in relation to contemporary knowledge to determine which parts of the Recovery Plan will need updating. The recommendations of the Committee were presented to the Service and Desert Tortoise Management Oversight Group on March 24, 2004. The recommendations will be used as a guide by a recovery team of scientists and stakeholders to modify the 1994 Recovery Plan. A revised recovery plan is anticipated in 2006.

The Committee recognized that the distribution and abundance data indicate trends leading away from recovery goals in some parts of the species' range. These results indicate a need for more aggressive efforts to facilitate recovery. Many of the original prescriptions of the Recovery Plan were never implemented although these prescriptions continue to be appropriate. New prescriptions should be prioritized to assess redundancies and synergies within individual threats.

<u>b. Recovery Units</u>

The Northeastern Mojave Recovery Unit occurs primarily in Nevada, but it also extends into California along the Ivanpah Valley and into extreme southwestern Utah and northwestern Arizona. Vegetation within this unit is characterized by creosote bush scrub, big galleta-scrub steppe, desert needlegrass scrub-steppe, and blackbrush scrub (in higher elevations). Topography is varied, with flats, valleys, alluvial fans, washes, and rocky slopes. Much of the northern portion of the Northeastern Mojave Recovery Unit is characterized as basin and range, with elevations from 2,500 to 12,000 feet. Desert tortoises typically eat summer and winter annuals, cacti, and perennial grasses. Desert tortoises in this recovery unit, the northern portion of which represents the northernmost distribution of the species, are typically found in low densities (about 10 to 20 adults per square mile).

A kernel analysis was conducted in 2003-2004 for the desert tortoise (Tracy *et al*. 2004) as part of the reassessment of the 1994 Recovery Plan. The kernel analyses revealed several areas in which the kernel estimations for live tortoises and carcasses did not overlap. The pattern of non-overlapping kernels that is of greatest concern is those in which there were large areas where the kernels encompassed carcasses but not live animals. These regions represent areas within DWMAs where there were likely recent die-offs or declines in tortoise populations. The kernel analysis indicated large areas in the Piute-Eldorado Valley where there were carcasses but no live tortoises. For this entire area in 2001, there were 103 miles of transects walked, and a total of 6 live and 15 dead tortoises were found, resulting in a live encounter rate of 0.06 tortoises per mile of transect for this area. This encounter rate was among the lowest that year for any of the areas sampled in the range of the Mojave desert tortoise (Tracy *et al*. 2004).

Kernel analysis for the Coyote Springs DWMA showed areas where the distributions of carcasses and living tortoises do not overlap; however, densities of adult tortoises for the region do not show a statistical trend over time. Thus, while there may be a local die-off occurring in the northern portion of this DWMA, this does not appear to influence the overall trend in the region as interpreted by study plot data. Because permanent study plots for this region were discontinued after 1996, if there have been recent declines in numbers they are not reflected in the kernel analysis. Nevertheless, large regions of non-overlapping carcass and live tortoise kernels in the regions were not identified adjacent to the Coyote Springs DWMA. The probability of finding either a live tortoise or a carcass was relatively very low for Beaver Dam Slope and Gold-Butte Pakoon and moderately low for Mormon Mesa/Coyote Springs.

The Eastern Mojave Recovery Unit is situated primarily in California, but also extends into Nevada in the Amargosa, Pahrump, and Piute valleys. In the Eastern Mojave Recovery Unit, desert tortoises are often active in late summer and early autumn in addition to spring because this region receives both winter and summer rains and supports two distinct annual floras on which they can feed. Desert tortoises in the Eastern Mojave Recovery Unit occupy a variety of vegetation types and feed on summer and winter annuals, cacti, perennial grasses, and herbaceous perennials. They den singly in caliche caves, bajadas, and washes. This recovery unit is isolated from the Western Mojave Recovery Unit by the Baker Sink, a low-elevation, extremely hot and arid strip that extends from Death Valley to Bristol Dry Lake. The Baker Sink area is generally not considered suitable for desert tortoises. Desert tortoise densities in the Eastern Mojave Recovery Unit can vary dramatically, ranging from 5 to as many as 350 adults per square mile (U.S. Fish and Wildlife Service 1994).

Ivanpah and Piute–Eldorado valleys contained study plots that were analyzed in the Eastern Mojave Recovery Unit analysis. While there was no overall statistical trend in adult density over time, the 2000 survey at Goffs and the 2002 survey at Shadow Valley indicate low densities of adult tortoises relative to earlier years. Unfortunately, there are no data in the latter years for all five study plots within this recovery unit, and therefore, while there is no statistical trend in adult densities, we cannot conclude that tortoises have not experienced recent declines in this area. The probability of finding a carcass on a distance sampling transect was considerably higher for Ivanpah, Chemehuevi, Fenner, and Piute-Eldorado, which make up the Eastern Mojave Recovery Unit.

The Northern Colorado Recovery Unit is located completely in California. Here desert tortoises are found in the valleys, on bajadas and desert pavements, and to a lesser extent in the broad, well-developed washes. They feed on both summer and winter annuals and den singly in burrows under shrubs, in intershrub spaces, and rarely in washes. The climate is somewhat warmer than in other recovery units, with only 2 to 12 freezing days per year. The tortoises have the California mitochondrial DNA (mtDNA) haplotype and phenotype. Allozyme frequencies differ significantly between this recovery unit and the Western Mojave, indicating some degree of reproductive isolation between the two.

Desert tortoises in the Eastern Colorado Recovery Unit, also located completely in California, occupy well-developed washes, desert pavements, piedmonts, and rocky slopes characterized by relatively species-rich succulent scrub, creosote bush scrub, and blue palo verde-ironwood-smoke tree communities. Winter burrows are generally shorter in length, and activity periods are longer than elsewhere due to mild winters and substantial summer precipitation. The tortoises feed on summer and winter annuals and some cacti; they den singly. They also have the California mtDNA haplotype and shell type.

The Upper Virgin River Recovery Unit encompasses all desert tortoise habitat in Washington County, Utah, except the Beaver Dam Slope, Utah population. The desert tortoise population in the area of St. George, Utah is at the extreme northeastern edge of the species' range and experiences long, cold winters (about 100 freezing days) and mild summers, during which the tortoises are continually active. Here the animals live in a complex topography consisting of canyons, mesas, sand dunes, and sandstone outcrops where the vegetation is a transitional mixture of sagebrush scrub, creosote bush scrub, blackbush scrub, and a psammophytic community. Desert tortoises use sandstone and lava caves instead of burrows, travel to sand dunes for egg-laying, and use still other habitats for foraging. Two or more desert tortoises often use the same burrow. Shell morphology and mtDNA have not been studied in this recovery unit, but allozyme variation is similar to that found in the Northeastern Mojave Recovery Unit.

The Western Mojave Recovery Unit occurs completely in California and is exceptionally heterogeneous and large. It is composed of the Western Mojave, Southern Mojave, and Central Mojave regions, each of which has distinct climatic and vegetational characteristics. The most pronounced difference between the Western Mojave and other recovery units is in timing of rainfall and the resulting vegetation. Most rainfall occurs in fall and winter and produces winter annuals, which are the primary food source of tortoises. Above ground activity occurs primarily in spring, associated with winter annual production. Thus, tortoises are adapted to a regime of winter rains and rare summer storms. Here, desert tortoises occur primarily in valleys, on alluvial fans, bajadas, and rolling hills in saltbrush, creosote bush, and scrub steppe communities. Tortoises dig deep burrows (usually located under shrubs on bajadas) for winter hibernation and summer aestivation. These desert tortoises generally den singly. They have a California mtDNA haplotype and a California shell type.

Distribution: The prescriptions for recovery in the Recovery Plan were for individual populations and assumed that preserving large blocks of habitat and managing threats in that habitat would be principally all that would be necessary to recover the species. However, that original paradigm, and the prescriptions made within that paradigm, may be wrong. Existing data have revealed population crashes that have occurred asynchronously across the range. There are reports that some populations, which have crashed previously, have subsequently increased in population density. Additionally, all known dense populations of desert tortoises have crashed. This suggests that density-dependent mortality occurs in desert tortoise populations, and that population dynamics may be asynchronous.

These characteristics indicate that tortoises may exist in a classic metapopulation structure (Hanski 1999; Levins and Culver 1971; Levins *et al*. 1984), and this should portend profoundly different prescriptions for recovery. In particular, if desert tortoises have historically existed in metapopulations, then connections among habitat patches are a necessary part of conservation prescriptions. Additionally, habitat suitable for tortoises, but without tortoises, should be regarded as equally necessary for recovery. Long-term persistence cannot be determined from tortoise density or tortoise numbers alone, but assessment must include the complexities of metapopulation dynamics and the habitat characteristics that promote metapopulation dynamics including habitat connectivity through inefficient corridors (*i.e.*, partial connectivity), asynchrony of subpopulation dynamics, and several separate habitat patches. Some of the characteristics of proper metapopulation function may already have been obviated by proliferation of highways, and habitat fragmentation due to satellite urbanization. Thus, management may require artificially facilitating metapopulation processes such as movement among patches.

The genetic distinctness of tortoise populations and their pathogens should be assessed to guide all manipulative management actions (*e.g.*, head starting, translocation, habitat restoration, and corridor management). The Committee proposed a revision to the previous delineation of recovery units, or distinct population segments (DPSs) based on new scientific information. The recommended delineations reflect the prevailing concepts of subpopulation “discreteness,” and “significance,” and incorporate morphological, behavioral, genetic, and environmental information. The Committee’s recommendation reduces the number of recovery units from six to five by leaving the original Upper Virgin River and Western Mojave units intact and recombining the four central units into three reconfigured units: Lower Virgin River Desert, Northeastern Mojave Desert (including Amargosa Valley, Ivanpah Valley, and Shadow Valley), and Eastern Mojave and Colorado Desert. These recommended DPSs are based largely on the best resolving biochemical/genetic data of Rainboth *et al*. (1989), Lamb *et al*. (1989), Lamb and Lydehard (1994), and Britten *et al*. (1997). Because these delineations are general and not definitive at this time, more data and analyses are required which may result in additional modification. Although DPSs have been proposed by the Committee, no DPSs have been officially designated by the Service.

The 1994 Recovery Plan conceived desert tortoises to be distributed in large populations that required large areas and large densities to recover. However, existing data are consistent with the possibility that tortoises have evolved to exist in *metapopulations*. Metapopulation theory conceives that tortoises are distributed in metapopulation patches connected with corridors that allow inefficient and asynchronous movements of individuals among the patches. This paradigm conceives that some habitat patches within the range of the desert tortoise will have low population numbers or no tortoises at all, and others will have higher population numbers. Movement among the patches is necessary for persistence of the "system." If desert tortoises evolved to exist in metapopulations, then long-term persistence requires addressing habitat fragmentation caused by highways and satellite urbanization. Ensuring the integrity and function of natural corridors among habitat patches might require active management of tortoise densities in habitat patches and associated corridors.

Land managers and field scientists identified 116 species of alien plants in the Mojave and Colorado Deserts (Brooks and Esque 2002). The proliferation of non-native plant species has also contributed to an increase in fire frequency in tortoise habitat by providing sufficient fuel to carry fires, especially in the intershrub spaces that are mostly devoid of native vegetation (U.S. Fish and Wildlife Service 1994; Brooks 1998; Brown and Minnich 1986). Changes in plant communities caused by alien plants and recurrent fire may negatively affect the desert tortoise by altering habitat structure and species composition of their food plants (Brooks and Esque 2002).

Numerous wildfires occurred in desert tortoise habitat across the range of the desert tortoise this year due to abundant fuel from the proliferation of non-native plant species after a very wet winter. In Nevada, BLM estimates that 400,000 acres of desert tortoise habitat burned, including 30,000 to 35,000 acres of desert tortoise critical habitat. Although the greatest extent of burned habitat occurred in Nevada, desert tortoise habitat also burned in Utah, Arizona, and California. Post wildfire analyses are underway to quantify the number of acres of both critical and non-critical habitat affected by these wildfires. Although tortoises were burned and killed by the wildfires, tortoise mortality estimates are not available at this time.

Disease was identified in the 1994 Recovery Plan as an important threat to the desert tortoise. Disease is a natural phenomenon in wild populations of animals and can contribute to population declines by increasing mortality and reducing reproduction. However, URTD appears to be a complex, multi-factorial disease interacting with other stressors to affect desert tortoises (Brown *et al*. 2002; Tracy *et al*. 2004). The disease occurs mostly in relatively dense desert tortoise populations, as mycoplasmal infections are dependent upon higher densities of the host (Tracy *et al*. 2004).

Reproduction: Desert tortoises possess a combination of life history and reproductive characteristics that affect the ability of populations to survive external threats. Tortoises grow slowly, require 15 to 20 years to reach sexual maturity, and have low reproductive rates during a long period of reproductive potential (Turner *et al*. 1984; Bury 1987; Tracy *et al*. 2004). At Yucca Mountain, Nye County Nevada (Northeastern Mojave Recovery Unit), Mueller *et al*. (1998) estimated that the mean age of first reproduction was 19 to 20 years; clutch size (1 to 10 eggs) and annual fecundity (0 to 16 eggs) were related to female size but annual clutch frequency

(0 to 2) was not. Further, Mueller suggested that body condition during July to October may determine the number of eggs a tortoise can produce the following spring. McLuckie and Friedell (2002) determined that the Beaver Dam Slope desert tortoise population, within the Northeastern Mojave Recovery Unit, had a lower clutch frequency (1.33 ± 0.14) per reproductive female and fewer reproductive females (14 out of 21) when compared with other Mojave desert tortoise populations. In the 1990's, Beaver Dam Slope experienced dramatic population declines due primarily to disease and habitat degradation and alteration (U.S. Fish and Wildlife Service 1994). The number of eggs that a female desert tortoise can produce in a season is dependent on a variety of factors including environment, habitat, availability of forage and drinking water, and physiological condition (Henen 1997; McLuckie and Fridell 2002).

Numbers: Data collected on 1-square-mile permanent study plots indicate that tortoise populations have declined both in numbers of tortoises found during surveys and in densities of live tortoises at most sites since the plots were first established 20-30 years ago (Berry *et al*. 2002). Declines of 50 to 96 percent have occurred regardless of initial tortoise densities. Increases in the occurrence of shell-skeletal remains have been found to correspond with declines in numbers and densities of live tortoises with the exception of certain plots where poaching has been documented (Berry 2003).

Results of desert tortoise surveys at three survey plots in Arizona indicate that all three sites have experienced significant die-offs. Six live tortoises were located in a 2001 survey of the Beaver Dam Slope Exclosure Plot (Walker and Woodman 2002). Three had definitive signs of URTD, and two of those also had lesions indicative of cutaneous dyskeratosis. Previous surveys of this plot detected 31 live tortoises in 1996, 20 live tortoises in 1989, and 19 live tortoises in 1980. The 2001 survey report indicated that it is likely that there is no longer a reproductively viable population of tortoises on this study plot. Thirty-seven live tortoises were located in a 2002 survey of the Littlefield Plot (Young *et al*. 2002). None had definitive signs of URTD. Twenty-three tortoises had lesions indicative of cutaneous dyskeratosis. Previous surveys of this plot detected 80 live tortoises in 1998 and 46 live tortoises in 1993. The survey report indicated that the site might be in the middle of a die-off due to the high number of carcasses found since the site was last surveyed in 1998. Nine live tortoises were located during the mark phase of a 2003 survey of the Virgin Slope Plot (Goodlett and Woodman 2003). The surveyors determined that the confidence intervals of the population estimate would be excessively wide and not lead to an accurate population estimate, so the recapture phase was not conducted. One tortoise had definitive signs of URTD. Seven tortoises had lesions indicative of cutaneous dyskeratosis. Previous surveys of this plot detected 41 live tortoises in 1997 and 15 live tortoises in 1992. The survey report indicated that the site may be at the end of a die-off that began around 1996-1997.

The Western Mojave has experienced marked population declines as indicated in the Recovery Plan and continues today. Spatial analyses of the Western Mojave show areas with increased probabilities of encountering dead rather than live animals, areas where kernel estimates for carcasses exist in the absence of live animals, and extensive regions where there are clusters of carcasses where there are no clusters of live animals. Collectively, these analyses point generally toward the same areas within the Western Mojave, namely the northern portion of the

Fremont-Kramer DWMA and the northwestern part of the Superior–Cronese DWMA. Together, these independent analyses, based on different combinations of data, all suggest the same conclusion for the Western Mojave. Data are not currently available with sufficient detail for most of the range of the desert tortoise with the exception of the Western Mojave (Tracy *et al*. 2004).

Declines in tortoise abundance appear to correspond with increased incidence of disease in tortoise populations. The Goffs permanent study plot in Ivanpah Valley, California, suffered 92 to 96 percent decreases in tortoise density between 1994 and 2000 (Berry 2003). The high prevalence of disease in Goffs tortoises likely contributed to this decline (Christopher *et al*. 2003). Upper respiratory tract disease has not yet been detected at permanent study plots in the Sonoran Desert of California, but is prevalent at study plots across the rest of the species' range (Berry 2003) and has been shown to be a contributing factor in population declines in the western Mojave Desert (Brown *et al*. 1999; Christopher *et al*. 2003). High mortality rates at permanent study plots in the northeastern and eastern Mojave and Sonoran Deserts appear to be associated with incidence of shell diseases in tortoises (Jacobson *et al*. 1994). Low levels of shell diseases were detected in many populations when the plots were first established, but were found to increase during the 1980s and 1990s (Jacobson *et al*. 1994; Christopher *et al*. 2003). A herpesvirus has recently been discovered in desert tortoises, but little is known about its effects on tortoise populations at this time (Berry *et al*. 2002; Origgi *et al*. 2002).

The kernel analysis of the Eastern Colorado Recovery Unit shows that the distributions of the living tortoises and carcasses overlap for most of the region. The Chuckwalla Bench study plot occurs outside the study area, which creates a problem in evaluating what may be occurring in that area of the recovery unit. However, the few transects walked in that portion of the DWMA yielded no observations of live or dead tortoises. This illustrates our concern for drawing conclusions from areas represented by too few study plots and leaves us with guarded concern for this region. The percentage of transects with live animals was relatively high for most DWMAs within the Eastern Colorado Recovery Unit. In addition, the ratio of carcasses to live animals was low within this recovery unit relative to others.

The status and trends of desert tortoise populations are difficult to determine based only upon assessment of tortoise density due largely to their overall low abundance, subterranean sheltering behavior, and cryptic nature of the species. Thus, monitoring and recovery should include a comprehensive assessment of the status and trends of threats and habitats as well as population distribution and abundance.

For more information on desert tortoise or expanded discussions on recovery units and recommended DPSs, please refer to the Recovery Plan (U.S. Fish and Wildlife Service 1994) and report prepared by the Committee (Tracy *et al*. 2004).

c. Critical Habitat - Rangewide

On February 8, 1994, the Service designated approximately 6.45 million acres of critical habitat for the Mojave population of the desert tortoise in portions of California (4.75 million acres), Nevada (1.22 million acres), Arizona (339 thousand acres), and Utah (129 thousand acres) (59 FR 5820-5846, also see corrections in 59 FR 9032-9036), which became effective on March 10, 1994. Desert tortoise critical habitat was designated by the Service to identify the key biological and physical needs of the desert tortoise and key areas for recovery, and focuses conservation actions on those areas. Desert tortoise critical habitat is composed of specific geographic areas that contain the primary constituent elements of critical habitat, consisting of the biological and physical attributes essential to the species' conservation within those areas, such as space, food, water, nutrition, cover, shelter, reproductive sites, and special habitats. The specific primary constituent elements of desert tortoise critical habitat are: sufficient space to support viable populations within each of the six recovery units, and to provide for movement, dispersal, and gene flow; sufficient quality and quantity of forage species and the proper soil conditions to provide for the growth of these species; suitable substrates for burrowing, nesting, and overwintering; burrows, caliche caves, and other shelter sites; sufficient vegetation for shelter from temperature extremes and predators; and habitat protected from disturbance and human-caused mortality.

Critical habitat units were based on recommendations for DWMAs outlined in the *Draft Recovery Plan for the Desert Tortoise (Mojave Population)* (U.S. Fish and Wildlife Service 1993). These DWMAs are also identified as "desert tortoise ACECs" by BLM. Because the critical habitat boundaries were drawn to optimize reserve design, the critical habitat unit may contain both "suitable" and "unsuitable" habitat. Suitable habitat can be generally defined as areas that provide the primary constituent elements.

ENVIRONMENTAL BASELINE

The action area for the effects analysis, as determined by the BLM, is a 10,000-acre area centered over each of the nine ROWs (*i.e.*, 90,000 acres total) on BLM-administered land where industry has existing or pending wind energy ROW applications as shown on the February 25, 2003, map provided with the BA. This area would have an approximate 2-mile radius from the center point of the ROW and may substantially exceed the actual action area for one or more project sites.

A general overview of the environmental baseline for the desert tortoise is described in the Las Vegas Resource Management Plan (RMP) (BLM 1998) and the programmatic biological opinion (File No. 1-5-98-F-053) issued to the BLM on June 18, 1998, for implementation of proposed actions described in the RMP, including ROWs. These documents are herein incorporated by reference.

Since the Mojave population of the desert tortoise was first listed under the Act in 1989, three regional-level HCPs have been implemented for development of desert tortoise habitat in Clark County, Nevada. At that time, approximately 89 percent of Clark County consisted of public lands administered by the Federal government, thereby providing little opportunity for mitigation for the loss of desert tortoise habitat under an HCP on non-Federal lands. Alternatively, funds are collected under HCPs and spent to implement conservation and recovery actions on Federal lands as mitigation for impacts that occur on non-Federal lands. BLM-managed lands are included in these areas where mitigation funds are used to promote recovery of the desert tortoise. A description of these HCPs is provided below:

(1) On May 23, 1991, the Service issued a biological opinion (File No. 1-5-91-FW-40) on the issuance of incidental take permit PRT-756260 under section 10(a)(1)(B) of the Act. The Service concluded that incidental take of 3,710 desert tortoises on up to 22,352 acres of habitat within the Las Vegas Valley and Boulder City in Clark County, Nevada, was not likely to jeopardize the continued existence of the desert tortoise. The permit application was accompanied by the *Short-Term Habitat Conservation Plan for the Desert Tortoise in the Las Vegas Valley, Clark County, Nevada* (Regional Environmental Consultants [RECON] 1991) (Short-term HCP) and an implementation agreement that identified specific measures to minimize and mitigate the effects of the action on desert tortoises.

On July 29, 1994, the Service issued a non-jeopardy biological opinion (File No. 1-5-94-FW-237) on the issuance of an amendment to incidental take permit PRT-756260 to extend the expiration date of the existing permit by 1 year (to July 31, 1995) and include an additional disturbance of 8,000 acres of desert tortoise habitat within the existing permit area. The amendment did not authorize an increase in the number of desert tortoises allowed to be taken under the existing permit. Additional measures to minimize and mitigate the effects of the amendment were also identified. Approximately 1,300 desert tortoises were taken under the authority of PRT-756260, as amended. In addition, during the Short-term HCP, as amended, approximately 541,000 acres of desert tortoise habitat have been conserved in Clark County on lands administered by BLM and National Park Service.

(2) On July 11, 1995, the Service issued an incidental take permit (PRT-801045) to Clark County, Nevada, including cities within the county and the Nevada Department of Transportation (NDOT), under the authority of Section 10(a)(1)(B) of the Act. The permit became effective August 1, 1995, and allowed the "incidental take" of desert tortoises for a period of 30 years on 111,000 acres of non-Federal land in Clark County, and approximately 2,900 acres associated with NDOT activities in Clark, Lincoln, Esmeralda, Mineral, and Nye counties, Nevada. The Clark County Desert Conservation Plan (DCP) served as the permittees' habitat conservation plan and detailed their proposed measures to minimize, monitor, and mitigate the effects of the proposed take on the desert tortoise (RECON 1995). The permitees and NDOT imposed and paid a fee of $550 per acre of habitat disturbance to fund these measures. The permittees expended approximately $1.65 million per year to minimize and mitigate the potential loss of desert tortoise habitat. The majority of these funds were used to implement minimization and mitigation measures, such as increased law enforcement; construction of highway barriers; road designation, signing, closure, and rehabilitation; and tortoise inventory and monitoring within the lands managed for tortoise recovery (e.g., ACECs). The benefit to

the species, as provided by the DCP, substantially minimized and mitigated those effects that occurred through development within the permit area and aided in recovery of the desert tortoise.

(3) On November 22, 2000, the Service issued an incidental take permit (TE-034927) to Clark County, Nevada, including cities within the county and NDOT which supersedes the DCP permit. In the biological/conference opinion (File No. 1-5-00-FW-575), the Service determined that issuance of the incidental take permit to Clark County would not jeopardize the listed desert tortoise or southwestern willow flycatcher (*Empidonax traillii extimus*), or any of the 76 species that are not listed or not proposed for listing under the Act that are covered under the incidental take permit. Under the special terms and conditions of the permit, take of avian species, with the exception of Peregrine falcon (*Falco peregrinus anatum*) and phainopepla (*Phainopepla nitens*), would not be authorized until acquisition of private lands in desert riparian habitats in southern Nevada has occurred. The incidental take permit allows incidental take of covered species for a period of 30 years on 145,000 acres of non-Federal land in Clark County, and within NDOT rights-of-way, south of the 38th parallel in Nevada. The Clark County Multiple Species Habitat Conservation Plan and Environmental Impact Statement (MSHCP) (RECON 2000), serves as the permittees' habitat conservation plan and details their proposed measures to minimize, mitigate, and monitor the effects of covered activities on the 78 species.

As partial mitigation under the DCP, carried forward in the MSHCP, the County purchased a conservation easement from the City of Boulder City in 1994. The term of the Boulder City Conservation Easement (BCCE) is for 50 years and will be retained in a natural condition with the purpose for recovery of the desert tortoise and conservation of other species in the area. Certain uses shall be prohibited within the BCCE including motor vehicle activity off designated roads, livestock grazing, and any activity that is inconsistent with the purposes of the BCCE. Much of the BCCE is also designated desert tortoise critical habitat. Within the boundary of the BCCE, Boulder City reserved the Solar Energy Zone for energy development projects in addition to adjacent energy generation facilities described previously.

EFFECTS OF THE PROPOSED ACTION ON THE LISTED SPECIES

Modification of the land use plans to adopt the proposed wind energy development program is anticipated to result in construction of the wind energy structures and access roads that impact the Mojave desert tortoise. The BA for the proposed project identifies 3,470 acres as being potentially disturbed on a long-term basis in Nevada. Within these acres, there are nine potential ROWs in Nevada that could potentially affect about 1,000 acres of tortoise habitat. Potential habitat impacts include removal of shrubs and other plants, disturbance the topsoil and surface soil crusts, soil compaction, increased erosion, and providing conditions that promote establishment of weeds and alien plants. Once established, alien plants, which are less nutritious for desert tortoises, may displace native plant species. BLM proposes to minimize desert tortoise habitat impacts by (1) requiring the use of existing roads and restricting vehicular travel to these roads, where applicable, (2) disturbing the minimal amount of habitat, (3) locating activities in previously disturbed areas, (4) implementing a habitat restoration plan, (5) cleaning vehicles and equipment before entry to project sites, (6) using certified weed-free materials, (7) monitoring access roads and utility transmission corridors for invasive (alien) plants, (8) salvaging topsoil for use in rehabilitation projects, and (9) incorporating project-specific measures when specific actions are proposed that may affect the desert tortoise.

Individual desert tortoises may be adversely affected during construction and operation of the wind energy structures. Desert tortoises may be buried in their burrows; killed or injured by project vehicles; trapped or injured by falling into open holes or trenches; killed or injured by vegetation mowing; adversely affected if they contact spilled, hazardous materials; or captured and displaced out of harm's way. Additional harassment may occur from increased levels of human activity; noise, and ground vibrations produced by vehicles and heavy equipment (Bondello 1976; Bondello *et al*. 1979); harassment or capture of tortoises by workers for use as pets; and the presence of pets which may kill, injure, or harass desert tortoises. Ground vibrations can cause desert tortoises to emerge from their burrows; slapping the ground several times within a few feet of a desert tortoise burrow entrance will often cause a desert tortoise to emerge (Medica *et al*. 1986). The use of explosives may also cause tortoises to emerge from their burrows and may collapse nearby burrows.

The BLM proposes to minimize most of the above effects by implementing the following measures: (1) identify and avoid important tortoise areas; (2) conduct [desert tortoise] surveys; (3) avoid important habitats or periods of courtship, breeding, and nesting; (4) prohibit pets from construction sites; (5) report wildlife problems to the BLM; (6) use noise reduction devices; (7) use explosives only outside safe distances from tortoises and their burrows; (8) apply erosion-control measures; (9) incorporate project-specific measures when specific actions are proposed that may affect the desert tortoise; and (10) inform workers to adhere to speed limits, not to disturb wildlife, or bring pets to project sites.

Trash accumulation at project sites may attract and concentrate predators such as ravens, coyotes, raptors, and kit fox, which may result in increased predation of desert tortoises. Natural predation in undisturbed, healthy ecosystems is generally not an issue of concern. However, predation rates may be altered when natural habitats are disturbed or modified. Common raven populations in the California deserts have increased 10-fold from 1968 to 1992 in response to expanding human use of the desert (Boarman and Berry 1995). Because ravens make frequent use of food, water, and nest site subsidies provided by humans, their population increases can be tied to this increase in food and water sources, such as landfills and septic ponds (Boarman 1992; U.S. Fish and Wildlife Service 1994). Ravens may be attracted to landfills or project sites if trash is accessible by scavengers (Boarman and Berry 1995; BLM 1990). Considering that ravens were very scarce in this area prior to 1940, it is assumed that the current level of raven predation on juvenile desert tortoises is an unnatural occurrence (BLM 1990). Raptors are also known desert tortoise predators which may be attracted to project area structures. BLM proposes to minimize most of these effects by: (1) implementing a litter-control program, (2) incorporating measures to reduce raptor use of the project site into project designs, including anti-perching and anti-nesting measures, (3) and incorporating project-specific measures when specific actions are proposed that may affect the desert tortoise.

New access roads and upgraded existing access roads may provide public access or increase public access into desert tortoise habitat. Increased public use may increase illegal collection of tortoises for pets and vandalism, and those habitat disturbances described above that may occur as a result of project activities.

EFFECTS OF THE ACTION ON DESERT TORTOISE CRITICAL HABITAT

Because the BLM proposes to avoid ACECs, and ACECs approximate desert tortoise critical habitat in the action area, no adverse effects to tortoise critical habitat are anticipated.

Cumulative Effects

Cumulative effects include the effects of future State, local, or private actions that are reasonably certain to occur in the action area considered in this biological opinion. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the Act.

The purpose of this project is to meet the demand for electrical power in the southwestern United States, which will likely contribute to an increase in development of private lands within the action area. As a result, desert tortoise habitat will likely continue to be destroyed and degraded. Within desert tortoise-occupied habitat, these effects are expected to be minimized and mitigated through habitat conservation plans currently in effect or under development.

CONCLUSION

After reviewing the current status of the desert tortoise, the environmental baseline for the action area, the effects of BLM’s proposed land use plan amendments to adopt wind energy development program, and the cumulative effects, it is the Service’s biological opinion that the project, as proposed and analyzed, is not likely to jeopardize the continued existence of the threatened desert tortoise (Mojave population) because, at worst, only about 1,000 out of about 3.5 million acres of tortoise habitat in Nevada could be adversely affected. Given the proposed minimization measures we anticipate that less than 1,000 acres of tortoise habitat will be adversely affected.

No effects to tortoise critical habitat are anticipated.

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CALIFORNIA CONDOR

On October 6, 1996, the Service announced its intention to reintroduce California condors into northern Arizona and southern Utah, and designate the released birds as a nonessential, experimental population (NEP) under Section 10(j) of the Endangered Species Act (61 FR 54043). On October 29, 1996, six California condors were released at the Vermilion Cliffs in Coconino County of northern Arizona. Since then, additional birds have been released. The designated experimental population area (ExPA) includes remote federal (BLM, USFS, and NPS) and Native American Reservation lands, and some private lands in northern Arizona, southern Utah and southeastern Nevada (61 FR 54043). The primary release site and current nesting sites occur at Grand Canyon National Park and Vermillion Cliffs, Arizona.

The California condor may occur throughout southern Utah in a variety of habitats in the Richfield, Moab, and Southern Utah Support Centers' FMP planning areas. Although most of the time the condors will occur within the designated ExPA, condors can have also been observed north of the ExPA boundary. Condors have been documented in Utah as far north as Flaming Gorge Reservoir; regular sightings occur in southern Utah, particularly in the vicinity of Zion National Park/Kolob Canyons.

Condors from a nonessential experimental population, when they occur on BLM land, are subject to the conferral requirement of section 7(a)(4) of the Act. Conferral is required for any Federal agency action likely to jeopardize such a species. Inasmuch as the basis for designating an experimental population to be nonessential is a determination that complete loss of the population would not be likely to appreciably reduce the likelihood of survival of the species in the wild, the Service does not believe that any effect to the nonessential experimental population of the California condor could be likely to jeopardize the species, and that consequently conferral is not required in this instance.