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Vegetation Treatments on Bureau of Land Management Lands in 17 Western States

Bureau of Land Management

Draft Biological Assessment

Vegetation Treatments on Bureau of Land Management Lands in 17 Western States

Prepared by

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CHAPTER 1

INTRODUCTION

Background

This programmatic Biological Assessment (BA) analyzes the potential effects to federally-listed threatened and endangered species, and species proposed for listing, and their habitats as a result of vegetation treatments proposed by the U. S. Department of the Interior (USDI) Bureau of Land Management (BLM). With an overall goal of improving ecosystem health, the BLM proposes to treat up to 6 million acres of land in the 17 western United States, including Alaska, on an annual basis (Map 1-1). A *Vegetation Treatments on Bureau of Land Management Lands in 17 Western States Programmatic Environmental Impact Statement* (PEIS) and a *Vegetation Treatments Using Herbicides on Bureau of Land Management Lands in 17 Western States Programmatic Environmental Report* (PER) are currently being prepared for this treatment program.

The BLM last assessed its program-wide vegetation treatment methods during the late 1980s and early 1990s. Environmental Impact Statements and Records of Decision (RODs) were prepared that covered vegetation treatments in 14 western states in the continental U.S. (USDI BLM 1985, 1987, 1988, 1991, 1992). However, BAs were not prepared in conjunction with these earlier impact analyses.

At the time earlier EISs were completed, the BLM was proposing to treat only about 16% of the total acreage that would be treated under the program that is now being proposed. Because the impacts under the new program are likely to be much greater than those assessed in earlier EISs, the BLM is preparing a new PEIS for the proposed increase in use of herbicides on BLM-administered lands (public lands). Other proposed treatment activities (i.e., fire, and mechanical, manual, and biological control) are being addressed in a PER, since the use of these techniques has been affirmed in previous EISs.

This BA is prepared in accordance with Section 7 of the federal Endangered Species Act (ESA; the Act) of 1973, as amended (19 U.S.C. 1536 [c], 50 CFR 402.14[c]). The ESA requires that federal agencies "insure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any listed species or result in the destruction or adverse modification of critical habitat of such species." The purpose of the Act is to provide a means for conserving the ecosystems upon which threatened and endangered species depend, and to provide a program for protecting these species.

The BA also complies with several other rules and regulations that govern threatened and endangered species. These include the ***Fish and Wildlife Conservation Act of 1980,*** which encourages federal agencies to conserve and promote the conservation of non-game fish and wildlife species and their habitats. ***Executive Order 13186, Responsibilities of Federal Agencies to Protect Migratory Birds***, requires that federal agencies that have, or are likely to have, a measurable negative effect on migratory bird populations develop a Memorandum of Understanding (MOU) with the USFWS that shall promote the conservation of migratory bird populations. If the U.S. Fish and Wildlife Service (USFWS) determines that migratory birds could be harmed by BLM vegetation treatment actions, the two agencies would develop a site-specific assessment and mitigation to prevent harm to these birds. The ***Bald Eagle Protection Act***, passed by Congress in 1940, prohibits the take, possession, sale, purchase, barter, or offer to sell, purchase, or barter, export or import of the bald eagle at any time or in any manner. In 1962, Congress amended the Eagle Act to cover golden eagles. The ***Sikes Act*** authorizes the USDI to plan, develop, maintain, and coordinate programs with state agencies for the conservation and rehabilitation of wildlife, fish, and game on public lands. Agency-wide guidance in the protection and management of species of concern, and consultation requirements, is given in ***BLM Manual 6840 (Special Status Species)***.

The purpose of this BA is to:

- Evaluate the effects of the proposed action on listed species, species proposed for listing, and/or their critical habitat, that are known to be or could be present within the project area.
- Determine the need for consultation and conference with the National Oceanic and Atmospheric Administration Fisheries Service (NOAA Fisheries) and the USFWS.
- Meet the requirements of the ESA and the National Environmental Policy Act (NEPA, 42 U.S.C. 4321 *et seq*., implemented at 40 Code of Federal Regulations [CFR] parts 1500-1508).
- Ensure that the BLM recovers or maintains populations of listed species or species proposed for listing that occur on public lands by outlining mitigation and standard operating procedures (SOPs) for groups of species that react similarly to the vegetation treatments proposed in this document.

An important overriding assumption of the BA is that each site-specific action that could occur under the proposed action will be analyzed as required by NEPA and the ESA, and that there will be compliance with all federal laws during implementation of the project. Since the PEIS and PER are programmatic in nature, it does not authorize a specific commitment of resources. Therefore, any proposed site-specific activity will require a site-specific NEPA analysis and consultation between the local BLM field office and NOAA Fisheries and USFWS. The procedures that the BLM field office would follow during consultation are summarized in Chapter 3.

The BLM, NOAA Fisheries, and USFWS met in November 2001 to discuss the procedures for preparing a consultation agreement for the PEIS. A memorandum outlining these procedures was finalized in May 2002. This memorandum identified activities that would occur during informal consultation and information that the BLM would provide to NOAA Fisheries and USFWS as part of the initiation package to begin the formal consultation process. The initiation package would include the BA, as well as ecological risk assessments (ERAs) that address the risks to threatened and endangered species, and species proposed for listing (collectively known throughout this document as TEP species), from the herbicides that the BLM now uses, or proposes to use, to treat vegetation. The memorandum also stated that formal consultation would begin with the release of the draft PEIS to the public.

During January through March 2002, the BLM held 19 public scoping meetings in the western U.S., including Alaska, and in Washington, D.C. During this period, the public commented on a wide range of issues related to the proposed vegetation treatment activities, including the potential effects of treatments on threatened and endangered species, and species proposed for listing. These comments were summarized in a *Scoping Comment Summary Report for the Vegetation Treatments EIS* in May 2002 (ENSR 2002a).

Beginning in spring 2002, the BLM also participated in an Ad Hoc Interagency Team to address the effects of invasive vegetation and noxious weed treatments on humans, plants, and animals. This team consisted of the BLM, U.S. Environmental Protection Agency (USEPA), NOAA Fisheries, and USFWS. Information gained by the agency team was used to prepare this BA.

In May 2002, the BLM began the process of developing the assessment procedures that would be followed while conducting ERAs. This process involved close coordination with NOAA Fisheries, the USFWS, and the USEPA; representatives of these agencies participated in weekly telephone calls with the BLM and its contractor who prepared the ERAs. These agencies also provided information they felt was necessary to meet their requirements for consultation under the ESA, and reviewed draft work products prepared by the BLM contractor. In November 2002, the BLM submitted a draft *Vegetation Treatments Programmatic EIS Ecological Risk Assessment Protocol* (ENSR 2002b) to the USEPA, NOAA Fisheries, and USFWS, and requested they review the document. The BLM also requested that the agencies provide comments on the document, indicating issues that must be addressed in the ERA protocol to ensure that the ERA would meet the requirements of NOAA Fisheries and the USFWS for consultation under the ESA, as applicable to treatments involving the use of herbicides.

The USEPA provided comments to the BLM in mid-December 2002. NOAA Fisheries provided comments to the BLM in early March 2003. These comments were used in the development of the final ERA protocol (ENSR 2004). Risk assessments for 10 chemicals were completed in May 2005. Information from the ERAs is included in this BA, including information on likely risks to TEP species, and on SOPs that should be followed to minimize these risks.

With the release of the Draft PEIS to the public in November 2005, the BLM initiated formal consultation with the USFWS and NOAA Fisheries as required under the ESA. The consultation process is described in more detail in Chapter 3.

The ESA defines an endangered species as a species that is in danger of extinction throughout all or a major portion of its range. A threatened species is defined as any species that is likely to become an endangered species within the foreseeable future throughout all or a major portion of its range. Critical habitat is a specific area or type of area that is considered to be essential for the survival of a species, as designated by the USFWS under the ESA.

The species addressed in this BA were identified by Endangered Species Coordinators at BLM offices serving each of the 17 states included in the project area. Species included on these lists are TEP species that are known to be located or could potentially be located on public lands, or that could be affected by activities occurring on public lands. In addition, listed species with designated critical habitat that occurs on public lands or that could be affected by activities occurring on public lands have been included as well. A total of 314 species or subspecies of plants and animals (including populations that are treated separately) are addressed in this BA, 307 of which are federally listed, and 7 of which are proposed for listing. Critical habitat has been designated for 119 of these species, as indicated in Table 1-1. The species and information presented in Table 1-1 is current as of September 2005. It is important to recognize that because this document is programmatic and addresses species over such a wide geographic range, information on species, listing status, and critical habitat is likely to change over time such that Table 1-1 will become less accurate with time. However, this BA will still be able to provide guidance for local BLM offices, since effects analyses are done largely by group of species, rather than individual species.

Document Organization

This BA contains four main parts: a description of the treatment methods proposed for use on public lands throughout the western U.S.; procedures that the BLM field offices will follow during consultation to ensure compliance with the ESA and mitigation identified in the PEIS and PER; background information on all plant and animal species that occur or are likely to occur within the project area that are federally-listed as threatened or endangered, or that are proposed for federal listing under the ESA as of April 2005; and a discussion of the potential effects of the proposed action on these species.

Chapter 2 of this BA provides a description of the proposed action, with detailed information about the methods that will be used to treat vegetation on pubic lands in the western continental U.S. and Alaska. These treatment methods include activities used by the BLM to improve ecosystem health on public lands by reducing levels of fuels and controlling weeds. The BLM has proposed to treat a total of 6 million acres annually by prescribed fire, herbicides, biological control, manual methods, and mechanical methods.

Chapter 3 identifies the procedures that the BLM field offices will follow to ensure that field offices comply with the requirements of the ESA, guidance provided in the PEIS, PER, and the *Vegetation Treatments Programmatic EIS Ecological Risk Assessment Protocol* (ENSR 2004), and guidance provided in BLM Manual 6840 (Special Status Species Management) and BLM Handbook H-1601-1 (Land Use Planning Handbook). Descriptions of these procedures are presented to make sure that each field office follows similar procedures for ensuring that actions taken by the BLM are consistent with the conservation needs of TEP species, and that these actions do not contribute to the need to list any special status species under provision of the ESA.

Chapters 4 through 6 include background information and an analysis of the effects of the proposed action on the species covered by this BA. Species are divided into three main categories: plants (Chapter 4), fish and other aquatic animal species (Chapter 5), and terrestrial animals (Chapter 6). The intention of these divisions is to separate species into broad, biological groups, because of the large number of species considered in this document, and to facilitate logical analysis. In the first part of each section, background information on species abundance and distribution, habitat requirements, reproductive biology and life history, and current status and presence/absence of designated critical habitat is provided. Potential beneficial, direct, indirect, interdependent, and interrelated threats to the species that are unrelated to the proposed action, and that may result in cumulative effect as a result of the proposed action, are also presented (for a more detailed discussion of types of effects, see USFWS and National Marine Fisheries Service 1998). These effects are defined as follows:

- Beneficial – Effects of an action that are wholly positive, without any adverse effects, on a listed species or designated critical habitat. Determination that an action will have beneficial effects is a "may effect" situation.
- Direct – The direct or immediate effects of the project on the species or its habitat. Direct effects result from the agency action including the effects of interrelated actions and interdependent actions.
- Indirect – Effects caused by or resulting from the proposed action, are later in time, and are reasonably certain to occur. Indirect effects may occur outside of the area directly affected by the action.
- Interdependent – Effects that result from an activity that has no independent utility apart from the action on consideration.
- Interrelated – Effects that result from an activity that is part of the proposed action and depends on the proposed action for its justification.
- Cumulative – Include the effects of future state, tribal, local, or private actions that are reasonably certain to occur in the action area considered in this biological assessment. 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 ESA.

For presentation of background material, species are organizing using Bailey's Ecoregion Divisions (Bailey 1995). These divisions allow species to be separated based on geography and broad habitat types, and are the same as those used for much of the analysis in the PEIS and PER. In the second part of each section, the potential effects of the proposed action on the species discussed are presented. In many cases, the effects on a logical grouping of species are described, with grouping systems described in the beginning of each of the three sections. For clarity, the effects of each of the five individual treatment methods are considered separately.

In addition, information on essential fish habitat (EFH) is provided in Appendix A. In 1976, Congress passed into law what is currently known as the Magnuson-Stevens Fishery Conservation and Management Act (MSA). This law authorized the U.S. to manage its fishery resources to a distance of 200 miles off the coast. Under this law, all federal agencies are required to consult with NOAA Fisheries on all actions or proposed actions that are permitted, funded, or undertaken by the agency and that may adversely affect EFH. Essential fish habitat is defined by Congress as "waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity." For the purpose of interpreting the definition of EFH habitat, "waters" include aquatic areas and their associated physical, chemical, and biological properties; "substrate" includes sediment underlying the waters; "necessary" refers to the habitat required to support a sustainable fishery and to manage the species' contribution to a healthy ecosystem; and "spawning, breeding, feeding, or growth to maturity" covers all habitat types utilized by a species throughout its life cycle.

Because of the vast area covered by this project, and the large number of species to be considered by this BA, it was not feasible to include precise information about where listed species or critical habitat are located on public lands, and how those populations are currently managed. Rather, this BA assumes that all TEP species known to occur or that potentially occur on public lands are present in areas where all treatment methods could be utilized.

TABLE 1-1
Species Addressed in This Biological Assessment

Scientific Name	Common Name	Status	State[1]	Critical Habitat	Critical Habitat on BLM Lands	USFWS Recovery Plan
Plants						
Acanthomintha ilicifolia	San Diego thorn-mint	T[2]	CA	No	--	No
Agave arizonica	Arizona agave	E	AZ	No	--	No
Allium munzii	Munz's onion	E	CA	Proposed	--	No
Ambrosia pumila	San Diego ambrosia	E	CA	No	--	No
Amsonia kearneyana	Kearney's blue-star	E	AZ	No	--	Yes
Arabis mcdonaldiana	McDonald's rock-cress	E	CA, OR	No	--	No
Arctomecon humilis	Dwarf bear-poppy	E	UT	No	--	No
Arctostaphylos morroensis	Morro manzanita	T	CA	No	--	Yes
Arctostaphylos myrtifolia	Ione manzanita	T	CA	No	--	No
Arenaria paludicola	Marsh sandwort	E	OR	No	--	Yes
Argemone pleiacantha ssp. *pinnatisecta*	Sacramento prickly poppy	E	NM	No	--	Yes
Asclepias welshii	Welsh's milkweed	T	AZ, UT	Yes	1,600 acres (UT)	Yes
Astragalus albens	Cushenbury milk-vetch	E	CA	Yes	850 acres	Yes
Astragalus ampullarioides	Shivwitz milk-vetch	E	UT	No	--	No
Astragalus applegatei	Applegate's milk-vetch	E	OR	No	--	Yes
Astragalus brauntonii	Braunton's milk-vetch	E	CA	No	--	Yes
Astragalus desereticus	Deseret milk-vetch	T	UT	No	--	No
Astragalus holmgreniorum	Holmgren milk-vetch	E	AZ, UT	No	--	No
Astragalus humillimus	Mancos milk-vetch	E	CO, NM	No	--	Yes
Astragalus jaegerianus	Lane Mountain milk-vetch	E	CA	No	--	No
Astragalus lentiginosus var. *coachellae*	Coachella Valley milk-vetch	E	CA	Proposed	1,000 acres	No
Astragalus lentiginosus var. *piscinensis*	Fish Slough milk-vetch	T	CA	Proposed	--	Yes
Astragalus magdalenae var. *peirsonii*	Peirson's milk-vetch	T	CA	Yes	19,899 acres	No
Astragalus montii	Heliotrope milk-vetch	T	UT	Yes	None	Yes
Astragalus osterhoutii	Osterhout milk-vetch	E	CO	No	--	Yes
Astragalus phoenix	Ash Meadows milk-vetch	T	NV	Yes	None	Yes
Astragalus tricarinatus	Triple-ribbed milk-vetch	E	CA	No	--	No
Atriplex coronata var. *notatior*	San Jacinto Valley crownscale	E	CA	No	--	No
Baccharis vanessae	Encinitis baccharis	T	CA	No	--	No
Berberis nevinii	Nevin's barberry	E	CA	No	--	No
Brodiaea filifolia	Thread-leaved brodiaea	T	CA	Proposed	--	No
Calystegia stebbinsii	Stebbins' morning-glory	E	CA	No	--	Yes
Camissonia benitensis	San Benito evening-primrose	T	CA	No	--	No
Carex specuicola	Navajo sedge	T	UT	Yes	None	No
Castilleja campestris ssp. *succulenta*	Fleshy owl's-clover	T	CA	Yes	NA	No
Caulanthus californicus	California jewelflower	E	CA	No	--	Yes
Ceanothus roderickii	Pine Hill ceanothus	E	CA	No	--	Yes

TABLE 1-1 (Cont.)
Species Addressed in This Biological Assessment

Scientific Name	Common Name	Status	State[1]	Critical Habitat	Critical Habitat on BLM Lands	USFWS Recovery Plan
Plants (Cont.)						
Centaurium namophilum	Spring-loving centaury	T	CA, NV	Yes	None	Yes
Chamaesyce hooveri	Hoover's spurge	T	CA	Yes	NA	Yes
Chlorogalum purpureum var. *purpureum*	Purple amole	T	CA	Yes	None	No
Chorizanthe howellii	Howell's spineflower	E	CA	No	--	Yes
Chorizanthe orcuttiana	Orcutt's spineflower	E	CA	No	--	No
Chorizanthe pungens var.*pungens*	Monterey spineflower	T	CA	Yes	None	Yes
Cirsium fontinale var. *obispoense*	Chorro Creek bog thistle	E	CA	No	--	Yes
Cirsium loncholepis	La Graciosa thistle	E	CA	Yes	None	No
Clarkia springvillensis	Springville clarkia	T	CA	No	--	No
Coryphantha robbinsorum	Cochise pincushion cactus	T	AZ	No	--	Yes
Coryphantha scheeri var. *robustispina*	Pima pineapple cactus	E	AZ	No	--	No
Coryphantha sneedii var. *leei*	Lee pincushion cactus	T	NM	No	--	No
Coryphantha sneedii var. *sneedii*	Sneed pincushion cactus	E	NM	No	--	No
Cycladenia humilis var. *jonesii*	Jones cycladenia	T	CA, AZ, UT	No	--	No
Deinandra (= hemizonia) *conjugens*	Otay tarplant	T	CA	Yes	None	Yes
Dodecahema leptoceras	Slender-horned spineflower	E	CA	No	--	No
Dudleya cymosa ssp. *Marcescens*	Marcescent dudleyea	T	CA	No	--	Yes
Echinocactus horizonthalonius var. *nicholli*	Nichol's Turk's head cactus	E	AZ	No	--	No
Echinocereus fendleri var. *kuenzleri*	Kuenzler hedgehog cactus	E	NM	No	--	No
Echinocereus triglochidiatus var. *arizonicus*	Arizona hedgehog cactus	E	AZ	No	--	No
Enceliopsis nudicaulis var. *corrugata*	Ash Meadows sunray	T	NV	Yes	None	Yes
Eremalche kernensis	Kern mallow	E	CA	No	--	Yes
Eriastrum densifolium ssp. *Sanctorum*	Santa Ana River woolly-star	E	CA	No	--	No
Erigeron decumbens var. *decumbens*	Willamette daisy	E	OR	No	--	No
Erigeron maguirei	Maguire daisy	T	UT	No	--	Yes
Erigeron parishii	Parish's daisy	T	CA	Yes	960 acres	Yes
Erigeron rhizomatus	Zuni fleabane	T	AZ, NM	No	--	No
Eriodictyon altissimum	Indian Knob mountain balm	E	CA	No	--	Yes
Eriodictyon capitatum	Lompoc yerba santa	E	CA	Yes	None	No
Eriogonum apricum	Ione buckwheat	E	CA	No	--	No
Eriogonum gypsophilum	Gypsum wild-buckwheat	T	NM	Yes	None	No
Eriogonum ovalifolium var. *vineum*	Cushenbury buckwheat	E	CA	Yes	430 acres	Yes
Eriogonum ovalifolium var. *williamsiae*	Steamboat buckwheat	E	NV	No	--	Yes
Eriogonum pelinophilum	Clay-loving wild buckwheat	E	CO	Yes	None	No
Erysimum menziesii	Menzies' wallflower	E	CA	No	--	Yes

TABLE 1-1 (Cont.)
Species Addressed in This Biological Assessment

Scientific Name	Common Name	Status	State[1]	Critical Habitat	Critical Habitat on BLM Lands	USFWS Recovery Plan
Plants (Cont.)						
Eutrema penlandii	Penland alpine fen mustard	T	CO	No	--	No
Fremontodendron californicum ssp. *decumbens*	Pine Hill flannelbush	E	CA	No	--	Yes
Fremontodendron mexicanum	Mexican flannelbush	E	CA	No	--	No
Fritillaria gentneri	Gentner's fritillary	E	OR	No	--	Yes
Galium californicum ssp. *sierrae*	El Dorado bedstraw	E	CA	No	--	Yes
Gaura neomexicana var. *coloradensis*	Colorado butterfly plant	T	CO, WY	Yes	None	No
Gilia tenuiflora ssp. *arenaria*	Monterey gilia	E	CA	No	--	Yes
Grindelia fraxino-pratensis	Ash Meadows gumplant	T	CA, NV	Yes	340 acres (CA)	Yes
Hackelia venusta	Showy stickseed	E	OR	No	--	No
Hedeoma todsenii	Todsen's pennyroyal	E	NM	Yes	None	Yes
Helianthus paradoxus	Pecos sunflower	T	NM	No	--	Yes
Howellia aquatilis	Water howellia	T	CA, ID, MT, OR	No	--	Yes
Ivesia kingii var.eremica	Ash Meadows ivesia	T	NV	Yes	None	Yes
Lasthenia conjugens	Contra Costa goldfields	E	CA	Yes	--	Yes
Layia carnosa	Beach layia	E	CA	No	--	Yes
Lembertia congdonii	San Joaquin woolly-threads	E	CA	No	--	Yes
Lepidium barnebyanum	Barneby ridge-cress	E	UT	No	--	Yes
Lepidium papilliferum	Slickspot peppergrass	PT	ID	No	--	No
Lesquerella congesta	Dudley Bluffs bladderpod	T	CO	No	--	Yes
Lesquerella tumulosa	Kodachrome bladderpod	E	UT	No	--	No
Lilaeopsis schaffneriana var. *recurva*	Huachuca water-umbel	E	AZ	Yes	34 miles	No
Lilium occidentale	Western lily	E	OR	No	--	Yes
Limnanthes floccosa californica	Butte County meadowfoam	E	CA	Yes	--	No
Limnanthes floccosa grandiflora	Large-flowered woolly meadowfoam	E	OR	No	--	No
Lomatium bradshawii	Bradshaw's desert-parsley	E	OR	No	--	Yes
Lomatium cookii	Cook's lomatium	E	OR	No	--	No
Lupinus sulphureus ssp. *kincaidii*	Kincaid's lupine	T	OR	No	--	No
Mentzelia leucophylla	Ash Meadows blazingstar	T	NV	Yes	NA	Yes
Mirabilis macfarlanei	Macfarlane's four-o'clock	T	ID, OR	No	--	Yes
Nitrophila mohavensis	Amargosa niterwort	E	CA, NV	Yes	1,200 acres (CA)	Yes
Opuntia treleasei	Bakersfield cactus	E	CA	No	--	Yes
Orcuttia californica	California orcutt grass	E	CA	No	--	Yes
Orcuttia inaequalis	San Joaquin Valley orcutt grass	T	CA	Yes	NA	Yes
Orcuttia pilosa	Hairy orcutt grass	E	CA	Yes	None	Yes
Orcuttia tenuis	Slender orcutt grass	T	CA	Yes	NA	Yes

TABLE 1-1 (Cont.)
Species Addressed in This Biological Assessment

Scientific Name	Common Name	Status	State[1]	Critical Habitat	Critical Habitat on BLM Lands	USFWS Recovery Plan
Plants (Cont.)						
Oxytheca parishii var. *goodmaniana*	Cushenbury oxytheca	E	CA	Yes	85 acres	No
Pediocactus bradyi	Brady pincushion cactus	E	AZ	No	--	No
Pediocactus despainii	San Rafael cactus	E	NM, UT	No	--	Yes
Pediocactus knowltonii	Knowlton cactus	E	CO, NM	No	--	No
Pediocactus peeblesianus var. *peeblesianus*	Peebles Navajo cactus	E	AZ	No	--	No
Pediocactus sileri	Siler pincushion cactus	T	AZ, UT	No	--	No
Pediocactus winkleri	Winkler cactus	T	UT	No	--	Yes
Penstemon haydenii	Blowout penstemon	E	WY	No	--	Yes
Penstemon penlandii	Penland beardtongue	E	CO	No	--	Yes
Phacelia argillacea	Clay phacelia	E	UT	No	--	No
Phacelia formosula	North Park phacelia	E	CO	No	--	No
Phlox hirsuta	Yreka phlox	E	CA	No	--	Yes
Physaria obcordata	Dudley Bluffs (Piceance) twinpod	T	CO, UT	No	--	Yes
Plagiobothrys hirtus	Rough popcornflower	E	OR	No	--	Yes
Plantanthera praeclara	Western prairie fringed orchid	T	MT, WY	No	--	Yes
Pogogyne nudiuscula	Otay mesa-mint	E	CA	No	--	Yes
Polystichum aleuticum	Aleutian shield fern	E	AK	No	--	Yes
Primula maguirei	Maguire primrose	T	UT	No	--	Yes
Pseudobahia bahiifolia	Hartweg's golden sunburst	E	CA	No	--	No
Pseudobahia peirsonii	San Joaquin adobe sunburst	T	CA	No	--	No
Purshia subintegra	Arizona cliff-rose	E	AZ	No	--	Yes
Ranunculus aestivalis	Autumn buttercup	E	UT	No	--	Yes
Schoenocrambe argillacea	Clay reed-mustard	T	NM, UT	No	--	Yes
Schoenocrambe barnebyi	Barneby reed-mustard	E	ID, UT	No	--	Yes
Schoenocrambe suffrutescens	Shrubby reed-mustard	E	UT	No	--	Yes
Sclerocactus glaucus	Uinta Basin hookless cactus	T	CO, UT	No	--	Yes
Sclerocactus mesae-verdae	Mesa Verde cactus	T	CO, NM, UT	No	--	No
Sclerocactus wrightiae	Wright fishhook cactus	E	UT	No	--	Yes
Senecio layneae	Layne's butterweed	T	CA	No	--	Yes
Sidalcea keckii	Keck's checker-mallow	E	CA	Yes	None	No
Sidalcea nelsoniana	Nelson's checker-mallow	T	OR	No	--	Yes
Sidalcea oregana var. *calva*	Wenatchee Mountains checker-mallow	E	OR	Yes	NA	Yes
Silene spaldingii	Spalding's catchfly	T	ID, MT, OR	No	--	No
Spiranthes delitescens	Canelo Hills ladies'-tresses	E	AZ	No	--	No

TABLE 1-1 (Cont.)
Species Addressed in This Biological Assessment

Scientific Name	Common Name	Status	State[1]	Critical Habitat	Critical Habitat on BLM Lands	USFWS Recovery Plan
Plants (Cont.)						
Spiranthes diluvialis	Ute ladies'-tresses	T	CO, ID, MT, NV, OR, UT, WY	No	--	Yes
Stephanomeria malheurensis	Malheur wire-lettuce	E	OR	Yes	160,000 acres	Yes
Streptanthus albidus ssp. *albidus*	Metcalf Canyon jewelflower	E	CA	No	--	Yes
Thelypodium howellii spectabilis	Howell's spectacular thelypody	T	OR	No	--	Yes
Townsendia aprica	Last Chance townsendia	T	UT	No	--	Yes
Tuctoria greenei	Greene's tuctoria	E	CA	Yes	None	Yes
Verbena californica	Red Hills vervain	T	CA	No	--	No
Yermo xanthocephalus	Desert yellowhead	T	WY	Yes	NA	No
Mollusks						
Assiminea pecos	Pecos assiminea snail	PE	NM	Proposed	--	No
Fontelicella idahoensis	Idaho springsnail	E	ID	No	--	Yes
Helminthoglypta walkeriana	Morro shoulderband snail	E	CA	Yes	None	Yes
Lanx sp.	Banbury Springs limpet	E	ID	No	--	Yes
Oxyloma haydeni kanabensis	Kanab ambersnail	E	AZ, UT	Proposed	--	Yes
Physa natricina	Snake River physa snail	E	ID	No	--	Yes
Pyrgulopsis bruneauensis	Bruneau Hot springsnail	E	ID	No	--	Yes
Pyrgulopsis neomexicana	Socorro springsnail	E	NM	No	--	Yes
Pyrgulopsis roswellensis	Roswell springsnail	PE	NM	Proposed	--	No
Taylorconcha serpenticola	Bliss Rapids snail	T	ID	No	--	Yes
Tryonia alamosae	Alamosa springsnail	E	NM	No	--	Yes
Tryonia kosteri	Koster's Tryonia	PE	NV	Proposed	--	No
Valvata utahensis	Utah valvata snail	E	ID, UT	No	--	Yes
Arthropods						
Ambrysus amargosus	Ash Meadows naucorid	T	NV	Yes	None	Yes
Boloria acrocnema	Uncompahgre fritillary butterfly	E	CO	No	--	Yes
Branchinecta conservatio	Conservancy fairy shrimp	E	CA	Yes	None	Yes
Branchinecta longiantenna	Longhorn fairy shrimp	E	CA	Yes	None	Yes
Branchinecta lynchi	Vernal pool fairy shrimp	T	CA, OR	Yes	344 acres (OR/WA)	Yes
Desmocerus californicus dimorphus	Valley elderberry longhorn beetle	T	CA	Yes	None	No
Euphydryas editha quino	Quino checkerspot butterfly	E	CA	Yes	None	Yes
Euproserpinus euterpe	Kern primrose sphinx moth	T	CA	No	--	No
Gammarus desperatus	Noel's amphipod	PE	NM	Proposed	--	No
Hesperia leonardus montana	Pawnee montane skipper	T	CO	No	--	Yes
Icaricia icarioides fenderi	Fender's blue butterfly	E	OR	No	--	No
Lepidurus packardi	Vernal pool tadpole shrimp	E	CA	Yes	15,808 acres	Yes

TABLE 1-1 (Cont.)
Species Addressed in This Biological Assessment

Scientific Name	Common Name	Status	State[1]	Critical Habitat	Critical Habitat on BLM Lands	USFWS Recovery Plan
Arthropods (Cont.)						
Nicrophorus americanus	American burying beetle	E	MT, WY	No	--	Yes
Pseudocopaeodes eunus obscurus	Carson wandering skipper	E	CA, NV	No	--	No
Speyeria zerene hippolyta	Oregon silverspot butterfly	T	OR	Yes	NA	Yes
Thermosphaeroma thermophilus	Socorro isopod	E	NM	No	--	No
Fishes						
Acipenser transmontanus	White sturgeon (Kootenia River population)	E	ID, MT	Yes	None	Yes
Catostomus microps	Modoc sucker	E	CA	Yes	None	No
Catostomus warnerensis	Warner sucker	T	CA, NV, OR	Yes	18 miles (OR/WA)	Yes
Chasmistes brevirostris	Shortnose sucker	E	CA, OR	Proposed	50 miles (OR)	Yes
Chasmistes cujus	Cui-ui	E	NV	No	--	Yes
Chasmistes liorus	June sucker	E	UT	Yes	None	Yes
Crenichthys baileyi baileyi	White River springfish	E	NV	Yes	None	Yes
Crenichthys baileyi grandis	Hiko White River springfish	E	NV	Yes	None	Yes
Crenichthys nevadae	Railroad Valley springfish	T	NV	Yes	None	Yes
Cyprinella formosa	Beautiful shiner	T	AZ, NM	Yes	None	Yes
Cyprinodon diabolis	Devil's Hole pupfish	E	NV	No	--	Yes
Cyprinodon macularius	Desert pupfish	E	AZ, CA	Yes	770 acres (CA)	Yes
Cyprinodon nevadensis mionectes	Ash Meadows Amargosa pupfish	E	NV	Yes	None	Yes
Cyprinodon nevadensis pectoralis	Warm Springs pupfish	E	NV	No	--	Yes
Cyprinodon radiosus	Owens pupfish	E	CA	No	--	Yes
Deltistes luxatus	Lost River sucker	E	CA, OR	No	30 acres (OR)	Yes
Empetrichthys latos	Pahrump poolfish	E	NV	No	--	No
Eremichthys acros	Desert dace	T	NV	Yes	9 acres	Yes
Gambusia nobilis	Pecos gambusia	E	NM	No	--	No
Gasterosteus aculeatus williamsoni	Unarmored threespine stickleback	E	CA	No	--	No
Gila bicolor mohavensis	Mojave tui chub	E	CA	No	--	No
Gila bicolor snyderi	Owens tui chub	E	CA	Yes	None	Yes
Gila bicolor ssp.	Hutton tui chub	T	OR	No	--	Yes
Gila bicolor vaccaceps	Cowhead Lake tui chub	PE	CA	--	--	--
Gila boraxobius	Borax Lake chub	E	OR	Yes	320 acres	No
Gila cypha	Humpback chub	E	AZ, CO, UT, WY	Yes	160 miles (UT)	Yes
Gila elegans	Bonytail chub	E	AZ, CA, CO, NV, UT, WY	Yes	50 miles (AZ); 160 miles (UT)	Yes
Gila intermedia	Gila chub	PE	AZ, NM	Proposed	--	--
Gila robusta jordani	Pahranagat roundtail chub	E	NV	No	--	Yes

TABLE 1-1 (Cont.)
Species Addressed in This Biological Assessment

Scientific Name	Common Name	Status	State[1]	Critical Habitat	Critical Habitat on BLM Lands	USFWS Recovery Plan
Fishes (Cont.)						
Gila seminuda (=robusta)	Virgin River chub	E	AZ, NV, UT	Yes	2,200 acres (AZ); 7,000 acres (UT)	Yes
Hybognathus amarus	Rio Grande silvery minnow	E	NM	Yes	None	Yes
Lepidomeda albivallis	White River spinedace	E	NV	Yes	None	Yes
Lepidomeda mollispinis pratensis	Big Spring spinedace	T	NV	Yes	None	Yes
Lepidomeda vittata	Little Colorado spinedace	T	AZ	Yes	0.25 miles	Yes
Meda fulgida	Spikedace	T	AZ, NM	Yes	72 miles (AZ); 13 miles (NM)	Yes
Moapa coriacea	Moapa dace	E	NV	No	--	Yes
Notropis girardi	Arkansas River shiner	T	NM	Yes	2 miles	No
Notropis simus pecosensis	Pecos bluntnose shiner	T	NM	Yes	64 miles	Yes
Oncorhynchus clarki henshawi	Lahontan cutthroat trout	T	CA, CO, NV, OR, UT	No	--	Yes
Oncorhynchus clarki stomias	Greenback cutthroat trout	T	CO	No	--	Yes
Oncorhynchus gilae	Gila trout	E	AZ, NM	No	--	Yes
Oncorhynchus keta	Chum salmon					
	Columbia River ESU	T	OR	Yes	NA	--
Oncorhynchus kisutch	Coho salmon					
	Central California Coast ESU	T	CA, OR	Yes	220,570 acres (OR)	--
	Southern Oregon/Northern California Coasts ESU	T	CA, OR	Yes	22,000 acres (CA); 329,000 acres (OR/WA)	--
	Oregon Coast ESU	T	OR	No*		--
Oncorhynchus mykiss	Steelhead					
	Southern California ESU	E	CA	Yes	NA	--
	South Central California Coast ESU	T	CA	Yes	NA	--
	California Central Valley ESU	T	CA	Yes	NA	--
	Northern California ESU	T	CA	Yes	NA	--
	Central California Coast ESU	T	CA	Yes	NA	--
	Snake River Basin ESU	T	ID, OR	Yes	36,547 acres; 501 miles	--
	Upper Willamette River ESU	T	OR	Yes	36,547 acres; 501 miles	--
	Upper Columbia River ESU	E	OR	Yes	36,547 acres; 501 miles	--

TABLE 1-1 (Cont.)
Species Addressed in This Biological Assessment

Scientific Name	Common Name	Status	State[1]	Critical Habitat	Critical Habitat on BLM Lands	USFWS Recovery Plan
Fishes (Cont.)						
Oncorhynchus mykiss (Cont.)	Lower Columbia River ESU	T	OR	Yes	36,547 acres; 501 miles	--
	Middle Columbia River ESU	T	OR	Yes	36,547 acres; 501 miles	--
Oncorhynchus nerka	Sockeye salmon					
	Snake River, Idaho ESU	E	ID, OR	Yes	632,600 acres (ID)	--
Oncorhynchus tshawytscha	Chinook salmon					
	California Coastal ESU	T	CA	Yes	NA	--
	Central Valley Spring-run ESU	T	CA	Yes	NA	--
	Sacramento River Winter-run ESU	E	CA, OR	Yes	None	--
	Snake River Fall-run ESU	T	ID, OR	Yes	632,910 acres (ID)	--
	Snake River Spring/Summer-run ESU	T	ID, OR	Yes	631,720 acres (ID); 20 miles (OR/WA)	--
	Lower Columbia River ESU	T	OR	Yes	20 miles (OR/WA)	--
	Upper Willamette River ESU	T	OR	Yes	NA	--
	Upper Columbia River Spring-run ESU	T	OR	Yes	NA	--
Oregonichythys crameri	Oregon chub	E	OR	No	--	Yes
Plagopterus argentissimus	Woundfin	E	AZ, NV, NM, UT	Yes	2,200 acres (AZ); 7,000 acres (UT)	Yes
Poeciliopsis occidentalis	Gila topminnow (incl. Yaqui)	E	AZ, NM	No	--	Yes
Ptychocheilus lucius	Colorado pikeminnow	E	AZ, CA, CO, NM, UT, WY	Yes	200 acres (CO); 350 miles (UT)	Yes
Rhinichthys osculus lethoporus	Independence Valley speckled dace	E	NV	No	--	Yes
Rhinichthys osculus nevadensis	Ash Meadows speckled dace	E	NV	Yes	None	Yes
Rhinichthys osculus oligoporus	Clover Valley speckled dace	E	NV	No	--	Yes
Rhinichthys osculus ssp.	Foskett speckled dace	T	OR	No	--	Yes
Rhinichthys osculus thermalis	Kendall Warm Springs dace	E	WY	No	--	No
Salvelinus confluentus	Bull trout	T	ID, MT, NV, OR	Yes	3,310 acres (ID); 111,765,151 acres (OR/WA)	Yes
Scaphirhynchus albus	Pallid sturgeon	E	CO, MT, WY	No	--	Yes

TABLE 1-1 (Cont.)
Species Addressed in This Biological Assessment

Scientific Name	Common Name	Status	State[1]	Critical Habitat	Critical Habitat on BLM Lands	USFWS Recovery Plan
Fishes (Cont.)						
Tiaroga cobitis	Loach minnow	T	AZ, NM	Yes	72 miles (AZ); 45 acres, 13 miles (NM)	Yes
Xyrauchen texanus	Razorback sucker	E	AZ, CA, CO, NM, NV, UT, WY	Yes	120 miles (AZ); 350 miles (UT)	Yes
Amphibians						
Ambystoma californiense	California tiger salamander	T	CA	No	--	No
Ambystoma tigrinum stebbinsi	Sonora tiger salamander	E	AZ	No	--	Yes
Batrachoseps aridus	Desert slender salamander	E	CA	No	--	No
Bufo baxteri (= *hemiophyrs*)	Wyoming toad	E	WY	No	--	Yes
Bufo californicus (= *microscaphus*)	Arroyo toad	E	CA	Yes	None	Yes
Rana aurora draytonii	California red-legged frog	T	CA	Yes	None	Yes
Rana chiricahuensis	Chiricahua leopard frog	T	AZ, NM	No	--	No
Reptiles						
Crotalus willardi obscurus	New Mexican ridge-nosed rattlesnake	T	AZ, NM	Yes	None	No
Gambelia silus	Blunt-nosed leopard lizard	E	CA	No	--	Yes
Gopherus agassizii	Desert tortoise (Mojave population)	T	AZ, CA, NV, UT	Yes	288,800 acres (AZ); 3,327,400 acres (CA); 1,085,000 acres (NV); 95,000 acres (UT)	Yes
Thamnophis gigas	Giant garter snake	T	CA	No	--	Yes
Uma inornata	Coachella Valley fringe-toed lizard	T	CA	Yes	12,000 acres	No
Birds						
Brachyramphus marmoratus marmoratus	Marbled murrelet	T	AK, CA, OR	Yes	92,000 acres (CA); 483,754 acres (OR/WA)	Yes
Charadrius alexandrinus nivosus	Western snowy plover (Pacific population)	T	CA, OR	Yes	274 acres (OR/WA)	Yes
Charadrius melodus	Piping plover	T	CO, MT, NM, WY	Yes	15 acres (MT)	Yes
Empidonax traillii extimus	Southwestern willow flycatcher	E	AZ, CA, CO, NV, NM, UT	Yes	54 miles (AZ); 400 ac, 9 miles (NM)	Yes
Falco femoralis septentrionalis	Northern aplomado falcon	E	AZ, NM	No	--	Yes
Glaucidium brasilianum cactorum	Cactus ferruginous pygmy-owl	E	AZ	Yes	91,000 ac	Yes
Grus americana	Whooping crane	E (XN)	CO, ID, MT, WY	Yes	--	Yes

TABLE 1-1 (Cont.)
Species Addressed in This Biological Assessment

Scientific Name	Common Name	Status	State[1]	Critical Habitat	Critical Habitat on BLM Lands	USFWS Recovery Plan
Birds (Cont.)						
Gymnogyps californianus	California condor	E, XN	E=CA XN=UT, AZ	Yes	None	Yes
Haliaeetus leucocephalus	Bald eagle	T	All (not listed in AK)	No	5,500,000 acres (ID); 200 acres (OR/WA)	No
Pelecanus occidentalis	Brown pelican	E	AZ, CA, OR	No	--	No
Phoebastria (=Diomedea) albatrus	Short-tailed albatross	E	AK, CA	No	--	No
Pipilo crissalis eremophilus	Inyo California towhee	T	CA	Yes	2,306 acres	Yes
Polioptila californica californica	Coastal California gnatcatcher	T	CA	Yes	None	No
Polystricta stelleri	Steller's eider	T	AK	Yes	5,000,000 acres	Yes
Rallus longirostris yumanensis	Yuma clapper rail	E	AZ, CA, NV	No	--	Yes
Somateria fischeri	Spectacled eider	T	AK	Yes	5,000,000 acres	Yes
Sterna antillarum	Least (interior) tern	E	CO, MT, NM, WY	No	--	Yes
Strix occidentalis caurina	Northern spotted owl	T	CA, OR	Yes	94,000 acres (CA); 1,061,648 acres (OR/WA)	No
Strix occidentalis lucida	Mexican spotted owl	T	AZ, CA, CO, NM, UT	Yes	10,700 acres (AZ); 149,894 acres (CO); 2,500 acres (NM); 1,646,388 acres (UT)	Yes
Vireo bellii pusillus	Least Bell's vireo	E	CA	Yes	None	Yes
Mammals						
Antilocapra americana sonoriensis	Sonoran pronghorn	E	AZ	No	--	Yes
Brachylagus idahoensis	Pygmy rabbit	E	OR	No	--	No
Canis lupus	Gray wolf	E, T, XN	E = AZ, CO, NM, WY T = ID, NV, MT, OR, UT XN = ID, NM	Yes	None	Yes
Cynomys parvidens	Utah prairie dog	T	UT	No	--	Yes
Dipodomys heermanni morroensis	Morro Bay kangaroo rat	E	CA	Yes	None	No
Dipodomys ingens	Giant kangaroo rat	E	CA	No	--	Yes
Dipodomys nitratoides exilis	Fresno kangaroo rat	E	CA	Yes	None	Yes
Dipodomys nitratoides nitratoides	Tipton kangaroo rat	E	CA	No	--	Yes
Dipodomys stephensi (incl. *D. cascus*)	Stephens' kangaroo rat	E	CA	No	--	Yes
Felis pardalis	Ocelot	E	AZ	No	--	Yes

TABLE 1-1 (Cont.)
Species Addressed in This Biological Assessment

Scientific Name	Common Name	Status	State[1]	Critical Habitat	Critical Habitat on BLM Lands	USFWS Recovery Plan
Mammals (Cont.)						
Leptonycteris curosoae yerbuensis	Lesser long-nosed bat	E	AZ, NM	No	--	Yes
Leptonycyteris nivalis	Mexican long-nosed bat	E	NM	No	--	Yes
Lynx canadensis	Canada lynx	T	AK, CO, ID, MT, OR, UT, WY	No	7,019 acres (OR/WA)	No
Microtus californicus scirpensis	Amargosa vole	E	CA	Yes	2,440 acres	Yes
Microtus mexicanus hualpaiensis	Hualapai Mexican vole	E	AZ	No	--	Yes
Mustela nigripes	Black-footed ferret	E, XN	E = AZ, CO, MT, UT, WY XN = AZ, CO, MT, UT, WY	No	--	No
Neotoma fuscipes riparia	Riparian (San Joaquin Valley) woodrat	E	CA	No	--	Yes
Odocoileus virginianus leucurus	Columbian white-tailed deer	E	OR	No	--	Yes
Ovis canadensis	Bighorn sheep (Peninsular Ranges)	E	CA	Yes	226,026 acres	Yes
Ovis canadensis californiana	Bighorn sheep (Sierra Nevada population)	E	CA	No	--	Yes
Panthera onca	Jaguar	E	AZ, NM	No	--	Yes
Rangifer tarandus caribou	Woodland caribou	E	OR	No	--	Yes
Sorex ornatus relictus	Buena Vista Lake ornate shrew	E	CA	Yes	None	Yes
Spermophilus brunneus brunneus	Northern Idaho ground squirrel	T	ID	No	--	No
Ursus arctos horribilis	Grizzly bear	T	ID, MT, OR, WY	No	--	Yes
Vulpes macrotis mutica	San Joaquin kit fox	E	CA	No	--	Yes
Zapus hudsonius preblei	Preble's meadow jumping mouse	T	CO, WY	Yes	None	No

[1] MT may include or refer to North Dakota and/or South Dakota; NM may include or refer to Texas and/or Kansas; OR may include or refer to Washington; and WY may include or refer to Nebraska.
[2] E = federally listed as endangered; T = federally listed as threatened; PE = proposed for listing as endangered; PT = proposed for listing as threatened; and XN = experimental, non-essential population.
NA = Due to incomplete information, recent litsing, or recent change in the status of critical habitat, number of acres of critical habitat on BLM-administed lands is unknown at this time.

The BA also assumes that all five treatment methods could be used where TEP species are found. This document assesses the potential impacts to all 314 proposed or listed species of all treatment methods, and identifies management activities (i.e., mitigation) required to avoid adverse impacts to these species.

This programmatic BA analyzes the potential overall effect of the BLM vegetation treatment program on the TEP species listed in Table 1-1 and discussed in detail in Chapters 4 through 6. When the BLM decides to implement a vegetation treatment program, local BLM offices will still be required, under NEPA, to prepare site (or project) specific analyses of TEP species potentially affected by the project and to consult with the Services. These analyses, which will cover considerably fewer species than this all-encompassing document, are expected to be more detailed in scope than this programmatic document.

Map 1-1

CHAPTER 2

PROPOSED ACTION

The primary objectives of the proposed vegetation treatment program are to manage hazardous fuels, control noxious weeds, and restore fish, wildlife, and rare plant habitat on public lands. Vegetation would be managed on approximately 6 million acres in 17 western states, including Alaska, using five primary treatment methods. About 2 million of these acres would be treated using fire, with mechanical and manual treatments occurring on approximately 2.5 million acres, herbicide treatments occurring on approximately 1 million acres, and biological control occurring on the remaining acres.

The BLM is seeking to expand its vegetation treatment program from current levels in order to promote conservation and improve public land health by slowing the rapid spread of invasive plants and noxious weeds, reducing hazardous fuel levels, and restoring fire-adapted ecosystems over many acres. The BLM also hopes to reduce economic losses to public and private property resulting from wildfire and invasive plant and noxious weed infestations, and provide NEPA documentation for vegetation treatments in Alaska.

The BLM, an agency of the USDI, manages nearly 262 million acres of land and 700 million acres of federal subsurface mineral estate nationwide. Bureau lands encompass almost 1 out of every 5 acres from the Rocky Mountains to the Pacific Ocean. There are several notable indications that the condition of public lands has degraded in some areas. In recent years, the severity and intensity of wildfires in the West has increased dramatically from levels of the 1970s and 1980s. There has also been a nearly 4-fold increase in invasive plant and noxious weed populations on public lands since 1985. Invasive plants and noxious weeds are the dominant vegetation on nearly 25 million acres of public land (USDI BLM 2000a). Invasive plants and noxious weeds are causing a steady degradation of soils, water quality and quantity, native plant communities, wildlife habitat, wilderness values, recreational opportunities, and livestock forage, and are detrimental to the agriculture and commerce of the U.S. and to public health (USDI BLM 2000b).

The Federal Land Management and Policy Act (FLMPA) requires that public lands under the jurisdiction of the BLM be managed for a variety of uses, including recreation, grazing, timber harvesting, and energy and mineral development, while at the same time ensuring that important environmental, historic, cultural, and scenic values (including threatened and endangered species and their habitats) are protected. However, many of these uses can be stressful to the land and lead to a decline in its health. In order to limit this land degradation, the BLM must use vegetation treatments, in addition to other management techniques, to restore degraded lands and to maintain lands that are healthy.

To reduce wildfire risk and improve land health, Congress directed the BLM and other federal agencies to develop *A Collaborative Approach for Reducing Wildland Fire Risks to Communities and the Environment 10-Year Comprehensive Strategy Implementation Plan* for reducing wildland fire risks to communities and the environment over the next 10 years (USDI and U.S. Department of Agriculture [USDA] Forest Service 2002). Under this plan, the BLM would use prescribed fire and other vegetation treatment methods on nearly 3 million acres of BLM-managed lands annually. In addition, under the Interagency Burned Area Emergency Stabilization and Rehabilitation Program, the BLM would restore approximately 1.5 million acres of wildfire-damaged lands annually through stabilization of soils and reseeding of fire-damaged areas. The remaining 1.5 million acres of would receive local treatments to control weeds, benefit fish and wildlife, improve riparian and wetland areas, and improve water quality in priority wetlands.

Fire Treatments

Fire is a treatment method that is used to reduce the buildup of hazardous fuels that can contribute to a fire's spread and intensity, control weeds, and maintain fire dependent species and ecosystems. Unlike other methods of vegetation management discussed in this chapter, fire can be used regardless of soil rockiness, slope steepness, or terrain irregularity, as long as adequate fuel is available to carry the fire.

A prescribed fire is the intentional application of fire to wildland fuels under specified conditions of fuels, weather, and other variables. The intent is for the fire to stay within a predetermined area to achieve site-specific resource management objectives. Prescribed fire may be used to control certain species; enhance the growth, reproduction, or vigor of certain species; manage fuel loads; and maintain vegetation community types that meet multiple-use management objectives (USDI BLM 1991).

The BLM may also utilize naturally ignited fires to accomplish resource objectives. Wildland fires may be utilized for resource benefit to maintain ecosystems that are functioning within their normal fire regime in areas where there is no threat to life and property. These fires must meet specific environmental prescriptions and be thoroughly evaluated for potential risk before being managed to benefit the resource. They are utilized only in pre-planned areas and when there are adequate fire management personnel and equipment available to achieve defined resource objectives.

The BLM conducts prescribed fire treatments in accordance with its Prescribed Fire Management Policy, which requires the preparation of a prescribed burning plan prior to every burn. Within these plans, a number of site-specific factors are evaluated, including project objectives, fuels present (quantity, type, distribution, moisture content), topography (ruggedness, elevation, slope), weather (temperature, wind, humidity), time of year, smoke dispersal, and predicted fire behavior (flame length, rate of spread). In all cases, fuel models are used to set standards for an area to be treated, and the burning treatment is delayed until the natural conditions of the site approach this standard (USDI BLM 1991). Under the proposed action, prescribed fire treatments would continue to be conducted in accordance with this policy.

Site Preparation

Prescribed fire projects typically consist of numerous activities, with the actual application of fire being only a small part of the total project (National Fire Plan Technical Team 2002). Preparation of a site for fire includes a number of activities with the potential to effect species and their habitats. The type of site preparation required depends on the local conditions and the individual project to be carried out. A number of possible activities are described here.

Road construction and maintenance may be required to provide access to some treatment sites. The extent of work related to this activity is dictated by the condition of the site and its roads. Some prescribed fire projects are located at remote locations and may require the creation of a temporary camp for personnel and their equipment. Depending on the size of the project, camps may be large and require daily shuttles of supplies and resources.

Prior to burning, a fireline is constructed to remove living and dead vegetation (i.e., fuel), or to create a break in its continuity, in order to help stop fire spread. The width of a fireline is determined by fuel type on the site and the anticipated flame length of the fire. The most common type of fireline is constructed using hand tools, by removing all plant material and downed dead material and exposing mineral soil. The equipment used is similar to the types of equipment used during manual control treatment methods. This type of fireline is often used on conjunction with other activities, such as black lining and wet lining (described below), and brush beating.

A machine-built fireline is created using mechanized equipment, such as bulldozers, tractors with plows, road graders, or four-wheelers. This type of fireline is utilized when a fuel break must be wide and/or lengthy, or when smaller fires have the potential to grow rapidly. In order to create a machine-built fireline, the site must have less than a 15% slope and be relatively free of surface rocks.

A wet line is created using water, with or without surfactants, which is sprayed on vegetation to increase moisture content or limit fire spread. Wet lines are most commonly used in short vegetation or fuel (e.g., grass, pine needles) and where flame lengths are short, and have the lowest impact of any human-constructed fireline. Because wet lines require large amounts of water, a reliable water source must be in the area to support these operations. Water can be drawn from ponds and streams using portable pumps, or pumps mounted to water tanks on fire engines or water tenders. In some cases, buckets suspended beneath helicopters may be used to strengthen a fireline or to quickly treat a hot spot. These buckets generally carry from 100 to 250 gallons of water, which is obtained from water sources nearby. A helibase or helispot must also be located close to the project, and refueling of the helicopter is typically done on-site.

Natural breaks in vegetation and fuel, such as rocky ridges or scab flats, riparian areas, wetlands, or pre-existing human-made breaks such as roads, can also be utilized to help contain prescribed fire. The vegetation in riparian areas and wetlands is too wet to support combustion and is very effective at limiting fire spread. However, these habitats can only be utilized while they are wet, and are not effective during the dry season.

An explosive built fireline is created using explosives, though this activity is used only under special circumstances and is uncommon. A long-linear explosive device is laid across the ground, and quickly removes burnable fuel and exposes mineral soil to stop the spread of a fire.

A black line is a pre-burned area that is used as a fireline, often in conjunction with other types of firelines. Vegetation is ignited on the inside of another type of fireline to create a wide fireline with minimal disturbance to the site.

Methods of Ignition

The BLM may start prescribed fires using a number of different techniques. Hand-held ignition sources include pressurized kerosene drip torches, propane torches, diesel flame-throwers, flares, and ignition grenades. Prescribed burns on large, accessible areas may be started with truck- or tractor-mounted flame-throwers. Additionally, helicopters may be used to aerially release an ignition fuel onto the area to be treated.

Hand ignition entails fire personnel walking through the burn area igniting the area in a set pattern. Hand ignition gives fire managers the highest level of control over the pattern of a prescribed burn (National Fire Plan Technical Team 2002). Mechanized ignition entails driving along a road or through the burn area, igniting vegetation. Like hand ignition, mechanized ignition allows an ignition pattern to be followed, with the added benefit of covering large areas over a short time period. Aerial ignition allows large, inaccessible areas to be treated with minimal impacts outside of the fire on the ground. Aerial ignition using large drip torches (helitorches) can ignite a large area in a relatively short amount of time, without ground impacts. The fuel used in helitorches is a gel mixture called alumagel. The chemicals used in this mixture must be transported and mixed in a level area close to the helispot, under regulations designated by the Department of Transportation and the Federal Aviation Administration (FAA).

Another aerial application device is referred to as a “ping-pong” ball dispenser, which releases ping-pong ball sized spheres filled with potassium permanganate onto the area to be treated. Just before the balls are dropped from the helicopter, they are injected with ethylene glycol, causing a chemical reaction that generates heat, which in turn causes the balls to ignite after they hit the ground. This technique is commonly used on lighter fuels, primarily for forest underburns, although its use is becoming more prevalent in shrub-steppe habitats.

Post-fire Activities

Once objectives have been achieved and ignition is no longer taking place, the so-called mop-up phase occurs, in which fire managers extinguish hot spots on the burn site. Hot spots are accumulations of dead material that continue to burn after the majority of the fire has gone out, such as stumps or downed logs. In most cases, the burning material is exposed and cooled with water and/or soil. Firefighters also use a combination of hand tools, fire engines, and hose lays to make sure the fire is contained within the unit before it is abandoned. Fire engines are used on flat terrain to bring water to the hot spots, and hose is placed along the ground in areas where vehicles cannot travel. Hoses are supplied with water from portable pumps, fire engines, or water tenders. Hand tools (e.g., shovels, backpack pumps, the Pulaski) are used to cool hotspots in areas that are inaccessible to vehicles and laying hose.

Mechanical Treatment Methods

Mechanical treatments are generally used to remove thick stands of vegetation, often to prepare the site for replanting a desired species. This method involves the use tractors or other types of vehicles with attached implements (e.g., plows, harrows, rangeland drills and mowers). These vehicles tend to remove all vegetation in the path of travel, and often uproot vegetation and disturb the soil. The type of mechanical method used on a particular site is based on characteristics of the undesired species present, seedbed preparation and revegetation needs, topography and terrain, soil characteristics, climatic conditions, and a comparison of the improvement costs to the expected productivity of the site (USDI BLM 1991). Mechanical treatment activities commonly occur in old agricultural areas, industrial sites, and roadsides (National Fire Plan Technical Team 2002). The BLM uses chaining, tilling and drilling seed, mowing, roller chopping and cutting, blading, grubbing, and feller-bunching.

Chaining entails pulling heavy (40 to 90 pounds per link) chains behind two crawler-type tractors in a “U” or “J” shaped pattern. Typically, the chain is 250 to 300 feet long, can weigh as much as 32,000 pounds, with a swath varying from 75 to 120 feet in width. Chaining works well for crushing brittle brush and uprooting woody plants. This practice can be done irregular, moderately rocky terrain, on slopes of up to 20%.

Tilling involves the use of angled disks (disk tilling) or pointed, metal-toothed implements (chisel plowing) to uproot, chop, and mulch vegetation. This technique is commonly used on sites where complete removal of vegetation or thinning is desired, often in conjunction with seeding operations. Tilling leaves mulched vegetation near the soil surface, which encourages the growth of newly planted seeds. The equipment used for tilling is typically a brushland plow, a single axle with an arrangement of angled disks that covers a swath of about 10 feet, or an offset disk plow, which consists of multiple rows of disks set at different angles to each other. Tilling equipment is pulled by either a crawler-type tractor or a large four-wheel-drive farm tractor. Tilling works best on areas with smooth terrain, with deep, rock-free soils, and is often used for removal of sagebrush and similar shrubs. Chisel plowing can be used to break up hard soils.

Seed drilling is often used in conjunction with tilling. The drills for seeding, which consist of a series of furrow openers, seed metering devices, seed hoppers, and seed covering devices, are either towed by or mounted on a tractor. The seed drill opens a furrow in the seedbed, deposits a measured amount of seed into the furrow, and then closes the furrow to cover the seed.

Mowing tools, such as rotary mowers or straight-edged cutter bar mowers, can be used to cut herbaceous and woody vegetation above the ground surface. This technique is often implemented along highway rights-of-way (ROW) to reduce fire hazards, improve visibility, prevent snow buildup, or improve the appearance of the area (USDI BLM 1991). It is most effective for treating annual and biennial plants, but rarely kills weeds after a single treatment. Although mowing does not typically remove roots, it can help eliminate undesired plant species by giving desired plants a competitive advantage (National Fire Plan Technical Team 2002).

Roller chopping tools are heavy bladed drums that cut and crush vegetation up to 5 inches in diameter using a rolling action. The drums are pulled by crawler-type tractors, farm tractors, or a special type of self-propelled vehicle designed for forested areas or range improvement projects.

Blading, which also utilizes crawler-type tractors, shears small brush at ground level. The topsoil may be scraped with the brush and piled into windrows during this operation. Blading use is limited to relatively-level areas and can only be used for certain undesirable plant species.

Grubbing is done with a crawler-type tractor that has been fitted with a brush rake or root rake attachment. The rake attachment consists of a standard dozer blade adapted with a row of curved teeth projecting forward at the blade base. The base of the blade is placed below the soil surface, allowing it to uproot brush and comb roots from the soil. Typically, grubbed areas are reseeded to prevent extensive runoff and erosion (USDI BLM 1991).

Feller-bunchers are machines that grab trees, cut them at the base, pick them up, and move them into a pile or onto the bed of a truck (Bonneville Power Administration 2000). They are used in forest thinning to remove potentially hazardous fuels.

Techniques for reseeding an area, commonly used in conjunction with mechanical control methods, include drill seeding and aerial application of seed. Drill seeding is commonly used on areas with moderate slopes, and entails the use of rangeland drills attached to tractors (National Fire Plan Technical Team 2002). Aerial seeding is the application of seed using fixed wing aircraft or helicopters.

Manual Treatment Methods

Manual treatment methods involve the use of hand-operated power tools and hand tools to cut, clear, or prune herbaceous and woody species. Plants may be cut at or above ground level, their root systems may be dug out to prevent sprouting and regrowth, or mulch may be placed around desired vegetation to limit competitive growth (USDI BLM 1991). A number of hand tools may be used during manual treatments: hand saws, axes, shovels, rakes, machetes, grubbing hoes, mattocks (a combination of axe and grubbing hoe), brush hooks, and hand clippers. Power tools, such as chainsaws and power brush saws, may also be used, particularly on thick-stemmed plants.

Manual treatments are most suitable for areas in which the weed infestation is limited and soil types allow for complete removal of the plant material. (Rees et al. 1996). Pulling also works well for annual and biennial plants, shallowly-rooted plant species that do not resprout from residual roots, and plants growing in sandy or gravelly soils (Colorado Natural Areas Program et al. 2000). Pulling is not recommended for use in dense infestations where native vegetation is not available to replace the pulled plants. Manual treatment methods can be used in many areas, usually with minimal environmental impacts. Manual techniques can be highly selective, and can be used in sensitive areas, where other treatment methods would not be appropriate, and in areas that are inaccessible to ground vehicles (USDI BLM 1991).

Biological Control Treatment Methods

Biological control methods involve the use of living organisms to selectively suppress, inhibit, or control herbaceous and woody vegetation (National Fire Plan Technical Team 2002). Biological control is often selected as an alternative to other treatment methods that have a greater environmental effect. The most common biological control agents are domestic animals, and parasitic insects that are host-specific to target weeds, although mites, nematodes, and pathogens are also used occasionally. Biological control treatments do not eradicate the target species, but do cause some mortality or weaken undesirable plants, thereby decreasing their vigor or competitive abilities in an ecosystem.

Domestic Animals

Domestic animals, such as sheep and goats, control the top-growth of certain noxious weeds, thereby weakening them. After a brief adjustment period, domestic animals can consume up to 50% of their daily diet of the weed. Sheep consume a variety of forbs, as well as grasses and shrubs, and goats can eat large quantities of woody vegetation (USDI BLM 1991). Goats and sheep can be effective control agents for leafy spurge and some types of shrubs (Colorado Natural Areas Program et al. 2000).

A number of considerations must be made before using domestic animals to control undesirable vegetation: the size of the infestation; the plant species present; the timing of consumption; the availability of a water source for stock; and whether stock can be managed to ensure beneficial effects (National Fire Plan Technical Team 2002). Cultural control treatments must be properly timed to be effective, utilizing the right combination of animals and stocking rates, and taking place during the appropriate season. Properly timed grazing of high intensity and short duration can prevent seed set of undesirable species or reduce their top-growth substantially. Domestic animal control methods are not suitable for use in erosion hazard areas, sites with compactable soils, riparian areas, or steep, erodible slopes. In addition, stock presence can encourage the spread of noxious weeds into non-infested areas; domestic animals should not be used as a treatment where such effects are likely. Because weed seeds may still be viable after passing through the digestive tract of an animal, domestic animals should not be moved to weed-free areas until all seeds have passed through their systems (Tu et al. 2001).

Other Biological Control Agents

Insects, mites, nematodes, and pathogens can reduce non-native plant populations by feeding on the plant, by destroying vital plant tissues and function, or by planting eggs in seedheads to reduce reproductive potential. These control agents are commonly used on sites where the population of target plants is large enough to support a viable population of the control agent, and when adequate numbers of the agents can be obtained. In many cases, three to five biological control agents are required to control a single plant species. In addition, it often takes several years for the biocontrol agents to establish themselves and have a visible impact on the plant population.

Insects, pathogens, and other biological control agents used by the BLM under the proposed action will have been tested to ensure that they are host specific, and they will feed only on the target plant, and not on crops, native flora, or sensitive plant species. The Plant Pest Quarantine Branch of the USDA Animal and Plant Health Inspection Service (APHIS), which issues permits and releases insects into the United States, regulates the use of these control agents. Information on the APHIS program and approval process is available at: http://www.aphis.usda.gov.

The Plant Protection Act of 2000 provides APHIS with the authority to regulate "any enemy, antagonist or competitor used to control a plant pest of noxious weed." However, the release of nonindigenous weed biocontrol agents into the environment is controlled by NEPA and the ESA.

The approval process for a biocontrol agent can be very complicated. Researchers wanting to use a candidate biological control agent should submit a proposed test plant list to the Technical Advisory Group (TAG) for Biological Control Agents of Weeds (USDA APHIS 2002). This includes consulting the USFWS to determine if threatened, endangered, or candidate species should be considered in the test plant list. The researcher must apply for a permit to import the agent into the U.S. As part of the permit process, the researcher is required to consult with the Services. In addition, if the researcher proposes to use a pathogen for weed biological control, he must obtain approval from the USEPA, which regulates microbial pathogens as biological pesticides under the Federal Insecticide, Fungicide, and Rodenticide Act of 1972 (FIFRA). Once a biological control organism has been approved for release, its release can only occur in those states that have been covered under NEPA and consultation with the Services.

Once a biological control agent such as an insect becomes established, it can reproduce and increase its numbers and continue to affect the target organism. These agents are also self-perpetuating, although it may take as many as

15 to 20 years for the agents to establish themselves and bring about the desired level of control. Treatments involving biological control agents are most suitable for large sites where the target plant is well established and very competitive with the desired species. It is unlikely that biological control agents will eradicate a pest plant, because as populations of the host plant decrease, populations of the agent will also decline.

The activities associated with non-domestic animal biological control include the collection and release of biological control agents, transport of agents by vehicle, inventory and monitoring of released agents to determine treatment success, and competitive seeding. Competitive seeding is a practice that can increase the success of biological control agents by establishing native/desirable plants that can compete with noxious weeds and help prevent soil erosion after control by agents (National Fire Plan Technical Team 2002). Competitive seeding treatments may require ground and/or aerial application of seeds and fertilizers.

Herbicide Treatments

Herbicides are chemical formulations that kill or injure plants by disrupting biochemical process. Typically, they are applied as liquids mixed with water or oil carriers, which are sprayed onto vegetation, although some are applied in solid form, as granules placed on the soil surface which are then absorbed by plant roots. An herbicide formulation includes an active ingredient, which is the chemical that kills the target plant, and one or more inert ingredients, which make the herbicide more effective. These inert ingredients may improve herbicide effectiveness by improving the solubility of the active ingredient, improving its ability to stick to plants or to penetrate protective layers on plant surface, or by limiting unintended drift of the herbicide mixture when it is sprayed. In this BA, all herbicides that contain a particular active ingredient are referred to by the name of that active ingredient, since it may be found in numerous products that are sold under different names. One exception is Overdrive®, an herbicide that includes dicamba and diflufuenzopyr as active ingredients; this herbicide is referred to in this BA by its product name, and the effects of both of its active ingredients are considered together.

Most herbicides used for the control of noxious weeds are selective for broad-leaved plants, so that they can kill weeds while maintaining grass forage species. Glyphosate is common herbicide that is non-selective, and can adversely affect non-target plants if used improperly.

Several federal laws govern herbicide use in the U.S. The *Federal Insecticide, Fungicide and Rodenticide Act (FIFRA)* establishes procedures for the registration, classification, and regulation of all pesticides. Before any pesticide may be sold legally, the USEPA must register it. The USEPA may classify a pesticide for general use if it determines that the pesticide is not likely to cause unreasonable adverse effects to applicators or the environment, or for restricted use if the pesticide must be applied by a certified applicator and in accordance with other restrictions. All the herbicides evaluated in the PEIS are registered with the USEPA, and all applicators that apply them on public lands (i.e., certified applicators or those directly supervised by a certified applicator) must comply with the herbicide label rates, uses, and handling instructions. In some cases, application rates allowed by the BLM are lower than the label application rates. The *Resource Conservation and Recovery Act (RCRA)* regulates the disposal of toxic wastes, including the disposal of unused herbicides. *The Comprehensive Environmental Response, Compensation and Liability Act (CERCLA)* regulates how to clean up spills of hazardous materials and when to notify agencies in case of spills.

The appropriate method for applying herbicides to unwanted vegetation is dependent upon a number of factors: pesticide labeling restrictions; the treatment objective (i.e., removal or reduction); the accessibility, topography, and size of the treatment area; the characteristics of the target species and the desired vegetation; the location of sensitive areas and potential environmental impacts in the immediate vicinity; the anticipated costs; equipment limitations; and the meteorological and vegetative conditions of the site (USDI BLM 1991).

Herbicide applications are scheduled and designed to minimize potential impacts to non-target plants and animals, while remaining consistent with vegetation treatment program objectives (National Fire Plan Technical Team 2002). Application rates are dependent on the presence of the target species; the condition of the non-target vegetation; soil type; depth to the water table, distance to open water sources, riparian areas and/or special status species; and the requirements printed on the herbicide label.

Over very large areas, herbicide treatments may be applied aerially by helicopter or fixed-wing aircraft. Aerial applications do not disturb the soil or protective organic layers, and are not limited by inaccessibility or rugged terrain. In general, helicopters are more maneuverable than fixed-wing aircraft, more effective in areas with irregular terrain, and more effective for treating specific target vegetation in areas with multiple vegetation types. A common problem associated with aerial application of herbicides is drift of chemicals off of the target site, which may be difficult to predict and manage.

Manual applications are suited for treatments of small areas or at sites that are inaccessible by vehicle (USDI BLM 1991). Manual spot treatments target individual plants through herbicide injections, applications on cut surfaces, or granular application to the surrounding soil (hand crank granular spreader). Application using backpack sprayers is another means of spot treatment, in which the herbicide applicator directs a spray hose at target plants. To cover a larger number of plants, mechanical equipment is used. In this method of treatment, herbicides are applied using a spray boom or wand attached to a truck, ATV, or other type of vehicle. Truck-mounted spraying is primarily limited to roadsides and flat areas that are accessible. However, ATVs can treat weeds in areas that are not easily accessible by road, such as hillsides.

Herbicides Proposed for Use by the BLM

Twenty different herbicides were approved for use in one or more states as part of the earlier EISs and the RODs for each state (Table 2-1). These decisions were based on a detailed analysis of the risks to human health and non-target species from the use of these chemicals.

Protocols used in developing ERAs for the earlier EISs were evaluated for their applicability in developing ERA protocols for new herbicides proposed for use by the BLM. Three issues were identified when reviewing the earlier ERAs. First, the ERAs may have identified risk levels for fish and wildlife that may be inconsistent with the BLM's current application rates/uses of these herbicides. Second, earlier ERAs may not have evaluated chronic and sublethal affects in sufficient detail to accurately predict risks to non-target plants, fish, and wildlife from herbicides approved for use by the BLM. Finally, the ERAs provided minimal guidance for determining appropriate mitigation and/or application methods to ensure that risks to TEP species would be below levels that could result in a taking.

A literature review was conducted as part of the PEIS to determine whether there is any new information to suggest that one or more of these 20 approved herbicides might no longer be safe for use on public lands. If so, new risk assessments would need to be conducted in order to determine whether these herbicides could continue to be used safely on public lands.

Based on the review of the earlier ERAs, the literature review, and consultations with the Services and USEPA, the BLM determined that the level of analysis of the risks to fish and wildlife in the ERAs done for the earlier EISs may have been inadequate to characterize the risks to species of concern, and that updated ERAs would be required to assess the risks of using these herbicides to species of concern.

Six chemicals currently approved for use by the BLM—2,4-DP, atrazine, asulam, fosamine, mefluidide, and simazine—have not been used, or only rarely used, by the BLM since 1997. Should these chemicals be used by the BLM in the future, the BLM would consult ERAs for these chemicals prepared by other agencies, if available, or conduct their own ERAs, to assess the risks to non-target species before using these chemicals.

During the mid- to late 1990s, the USDA Forest Service (Forest Service) conducted ERAs for eight herbicides also used by the BLM: 2,4-D, clopyralid, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, sulfometuron methyl, and triclopyr. In addition, the Forest Service prepared interactive spreadsheets that could be used to determine exposure concentrations under different application rates and exposure scenarios for these herbicides. The ERAs and spreadsheets are available on the Internet on the Forest Service Pesticide Management and Coordination website at http://www.fs.fed.us/foresthealth/pesticide/risk.htm. Information contained in these ERAs and spreadsheets was used by the BLM in the PEIS and BA to characterize risks to species of concern from these chemicals, as discussed in the following section (Ecological Risk Assessments).

The Forest Service did not conduct ERAs for four herbicides used by the BLM: bromacil, chlorsulfuron, diuron, and tebuthiuron. In addition, the BLM found that sulfometuron methyl would need to be evaluated further due to recent concerns regarding its transport in dust and potential impacts on nearby plants and animals. The BLM is also proposing to use four new herbicide active ingredients (diflufenzopyr, diquat, fluridone, imazapic), and a formulation dicamba and diflufenzopyr (Overdrive®) as part of the PEIS. These herbicides were selected based on: (1) input from BLM field offices on vegetation needing control; (2) studies that indicated these herbicides would be more effective in controlling noxious weeds and other unwanted vegetation targeted for control than herbicides currently used by the BLM; (3) USEPA approval for use on rangelands, forestlands, and/or aquatic environments; (4) responses from herbicide manufacturers to a letter from the BLM in October 2001 requesting them to submit the names of herbicides they felt would be appropriate to use on public lands to control vegetation; (5) their ability to be used on a variety of species needing control; (6) their level of risk to human health and the environment. Thus, the BLM conducted new ERAs for the four herbicide active ingredients and Overdrive® to determine the toxicity and environmental fate for these herbicides and their risks to species of concern (ENSR 2004).

Ecological Risk Assessments

Ecological risk assessments completed in support of PEIS (ENSR 2005a-j) identify the risks to plants and animals associated with using nine herbicide active ingredients and one formulation (bromacil, chlorsulfuron, diflufenzopyr, diquat, diuron, fluridone, imazapic, Overdive®, sulfometuron methyl, and tebuthiuron). In addition, Forest Service ERAs consulted by the BLM (Syracuse Environmental Research Associates, Inc. 2005) identify the risks to plants and animals associated with using eight additional herbicides (2,4-D, clopyralid, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr). The information provided in these risk assessments was used to determine the potential for effects to TEP plant and animal species and their habitats as a result of herbicide treatments on public land.

Risk assessments for these 18 herbicides characterized exposures scenarios involving a range of surrogate species, including species that have biological characteristics that are similar to those of TEP plant and animal species, and a range of exposure pathways associated with applications on a variety of upland and aquatic sites. A brief explanation of the methods used to determine the risks to non-target species as a result of herbicide use is presented below. A more detailed description of this methodology may be found in the *Vegetation Treatments Programmatic EIS Ecological Risk Assessment Methodology* (ENSR 2004) and Appendix C of the PEIS.

BLM Methodology

Surrogate species for TEP plants and animals were evaluated to determine assessment endpoints and associated measures of effect to be used in ERAs. Assessment endpoints, for the most part, reflect direct effects of an herbicide on these organisms, although indirect effects were also considered. Assessment endpoints for non-target species include mortality and adverse effects on growth, reproduction, or other ecologically important sublethal processes. Measures of effect are measurable changes in an attribute of an assessment endpoint (or its surrogate) in response to a stressor to which it is exposed (USEPA 1998). For the screening-level ERA, the measures of effect associated with the assessment endpoints generally consisted of acute and chronic toxicity data (from pesticide registration documents and from the available scientific literature) for the most appropriate surrogate species.

Because the BLM uses herbicides in a variety of programs (e.g., maintenance of rangeland and recreational sites) and application methods (e.g., via aircraft, vehicle, backpack), the following exposure scenarios were considered to assess the potential ecological impacts herbicides under a variety of uses and conditions:

Plants

- Direct spray of the receptor or water body
- Off-site drift of spray to terrestrial areas and water bodies
- Surface runoff from the application area to off-site soils or water bodies
- Wind erosion resulting in deposition of contaminated dust
- Accidental spills to water bodies

Aquatic Animals

- Direct spray of the water body
- Accidental spill to the water body
- Off-site drift of spray to the water body
- Surface runoff from the application area to the water bodies

Terrestrial Animals

- Direct spray of terrestrial wildlife
  Small mammal – 100% absorption
  Pollinating insect – 100% absorption
  Small mammal – 1st order dermal absorption (absorption occurs over 24 hours, taking into consideration the potential for some herbicide to not be absorbed)

- Indirect contact with foliage after direct spray
  Small mammal – 100% absorption
  Pollinating insect – 100% absorption
  Small mammal – 1st order dermal absorption

- Ingestion of food items contaminated by direct spray
  Small mammalian herbivore – acute and chronic exposure
  Large mammalian herbivore – acute and chronic exposure
  Small avian insectivore – acute and chronic exposure
  Large avian herbivore – acute and chronic exposure
  Large mammalian carnivore – acute and chronic exposure

- Ingestion of food items contaminated by surface runoff or off-site drift
  Piscivorous bird

Exposure scenarios involving off-site drift, surface runoff, and wind erosion were not modeled for terrestrial wildlife.

The AgDRIFT® computer model was used to estimate off-site herbicide transport due to spray drift. The GLEAMS computer model was used to estimate off-site transport of herbicide in surface runoff and root zone groundwater transport. The CALPUFF computer model was used to predict the transport and deposition of herbicides adsorbed (i.e., reversibly or temporarily attached) to wind-blown dust. Each model simulation was conservatively approached with the intent of predicting the maximum potential herbicide concentration that could result from the given exposure scenario.

In order to address potential risks to plant and animal receptors, Risk Quotients (RQs) were calculated. To facilitate the translation of RQs into readily applicable estimates of risk, the calculated RQs were compared with Levels of Concern (LOCs) used by the USEPA in screening the potential risk of pesticides. Distinct USEPA LOCs are currently defined for the following risk presumption categories:

- Acute high risk – the potential for acute risk is high
- Acute restricted use – the potential for acute risk is high, but may be mitigated
- Acute endangered species – TEP species may be adversely affected
- Chronic risk – the potential for chronic risk is high

For the analysis presented in this BA, LOCs for the acute endangered species and chronic risk categories were used. Wherever the RQ exceeded one or more of these LOCs, it was assumed that adverse effects to the TEP species in question could potentially occur under that exposure scenario.

Forest Service Methodology

The Forest Service risk assessment methodology was similar to that used by the BLM (see Syracuse Environmental Research Associates, Inc. 2001 for a complete description of the current methodology), except that some of the exposure pathways were different.

For TEP plants, the Forest Service developed four general and accidental/incidental exposure scenarios (i.e., direct spray, spray drift, runoff, and wind erosion) for groups of non-target vegetation according to the application method and the chemical and toxicological properties of the given herbicide. The Forest Service scenario of contaminated irrigation water—a direct application scenario—was not evaluated by the BLM because their vegetation treatment program does not typically involve irrigation of vegetation. In the case of wind erosion, the methodology differed from that in BLM ERAs. In BLM ERAs, long-range travel of contaminated soil was addressed, with dust deposition estimates calculated at distances ranging from 1.5 to 100 km (1 to 62 miles) from the application area. In contrast, the Forest Service ERAs looked at quantities of herbicides that could be lost from an application site, but not where eroded soil would land, or how much herbicide would be present in windblown soil within defined distances of the treatment site.

For TEP aquatic animals, Forest Service ERAs assessed risks to aquatic organisms via only two exposure pathways: 1) an accidental spill of 200 gallons of a field solution into a pond (acute exposure); and 2) long-term exposure to herbicide as a result of runoff from an adjacent right-of-way (chronic exposure).

Exposure scenarios used to determine risks to aquatic animals included exposure scenarios used to determine risks to terrestrial animals included direct spray, ingestion of contaminated media (vegetation, prey species, or water, and via grooming activities), and indirect contact with contaminated vegetation.

Risk assessments completed by the Forest Service developed hazard quotients (HQs), which are analogous to the RQs developed in BLM ERAs. To come up with estimates of risk that would be used in the BA and PEIS, HQs were compared with the USEPA's LOCs for chronic risk and acute endangered species risk categories. Wherever the HQ exceeded one or more of these LOCs, it was assumed that adverse effects to the species in question could potentially occur under that exposure scenario. Throughout this BA, the terms "adverse effect" and "adverse health effect" are used wherever ERAs predicted that an RQ or HQ exceeded an LOC for a particular exposure pathway.

Procedures to be Followed by Local Field Offices to Protect Species of Concern from Herbicide Applications

An important purpose of the ERAs is to provide guidance to BLM field offices on the proper method of application of herbicides to ensure that impacts to animals and non-target plants are minimized to the extent practical when treating vegetation. This guidance is also intended to ensure that treatment actions at the local level are not likely to jeopardize the continued existence of a listed species or result in the destruction or adverse modification of designated critical habitat. This information may also be useful in developing treatment application plans for herbicides that are already approved for use by the BLM.

The information provided in this BA, as obtained from ERAs, will allow the BLM to determine herbicide application methods and amounts that could be used without harming non-listed species. For listed species, additional safety factors have been identified (e.g., reducing the amount of chemical applied, or requiring a safety buffer between the treatment area and location of a listed species) as mitigation. This information may be used to help the BLM determine which herbicides could be applied, and how they could be applied, without jeopardizing the continued existence of a listed species or resulting in the destruction or adverse modification of designated critical habitat.

Using this information, the BLM will follow a set of procedures to protect TEP species when using herbicides currently approved for use:

- The BLM will identify appropriate application methods, including rate, time, and mode of application (source characterization) for projects involving the use of herbicides.
- The BLM will use interactive spreadsheets developed during preparation of the Forest Service and BLM ERAs to determine estimates of chemical exposure for a species of interest for herbicide applications in the action area. First, the TEP species will be sorted into the ERA surrogate classes based on food and shelter requirements and taxonomic similarity. Information on the chemical characteristics of the herbicide, mode and rate of application, and local environmental conditions (e.g., soil type, rainfall) are also entered into the spreadsheet to calculate the exposure value. These values can then be compared to a table listing risk levels to determine the potential for an acute or chronic risk to the species of interest. Risk levels for TEP species are provided in the ERA and in the following chapters.
- The BLM will incorporate mitigation measures identified in the ERA and BA, and from analysis of exposure levels based on modeling, to reduce or eliminate risks to TEP species to levels below those at which an unlawful taking could occur. It is possible that conservation measures would be less restrictive than those listed in subsequent sections of this BA if local site conditions were evaluated using the ERAs when developing project-level conservation measures.
- The BLM will use herbicides in a manner that is consistent with labeling instructions, design criteria, and any issued reasonable and prudent measures with terms and conditions to ensure that unlawful taking of an ESA-listed species does not occur. In the event incidental take is likely as a result of the action, the Biological Opinion (BO) will include an incidental take statement that exempts the BLM from the prohibitions of take under Section 9 of the ESA.

General guidance on exposure levels and on mitigation measures to reduce exposure levels to acceptable levels are provided in Sections 4 through 6 of the BA, in the ERA, and in the PEIS.

Under the PEIS Preferred Alternative, the BLM would also be able to use new chemicals that are developed in the future if: (1) they are registered by the USEPA for use on one or more land types (e.g., rangeland, forestland, aquatic, etc.) managed by the BLM; (2) the BLM has determined that the benefits of use on public lands outweigh the risks to human health and the environment; and (3) they meet evaluation criteria to ensure that the decision to use the chemical is supported by scientific evaluation and NEPA documentation. It is anticipated that the evaluation of new herbicides would include the preparation of an ERA following guidance in the PEIS.

TABLE 2-1
Herbicides Approved and Proposed for Use on Public Lands

Herbicide	Herbicide Characteristics	Areas Where Registered Use is Appropriate					
		Rangeland	**Forestland**	**Riparian and Aquatic**	**Oil, Gas, and Minerals**	**ROW**	**Recreation and Cultural Resources**
Herbicides Approved for Use on Public Lands							
2, 4-D	Selective; foliar absorbed; postemergent; annual/perennial broadleaf weeds.	•	•	•	•	•	•
2, 4-DP	Selective; foliar absorbed; postemergent; broadleaf weeds and woody species.	•	•		•	•	•
Asulam	Inhibits mitosis; controls growing grasses and certain broadleaf weeds.				•	•	
Atrazine	Selective; mostly root absorbed; inhibits photosynthesis.		•			•	
Bromacil	Non-selective; inhibits photosynthesis; controls wide range of weeds and brush.				•	•	•
Chlorsulfuron	Selective; inhibits enzyme activity; broadleaf weeds and grasses.	•			•	•	•
Clopyralid	Selective; mimics plant hormones; annual and perennial broadleaf weeds.	•	•		•	•	•
Dicamba	Growth regulator; annual and perennial broadleaf weeds, brush, and trees.	•			•	•	•
Diuron	Preemergent control; annual and perennial broadleaf weeds and grasses.				•	•	•
Fosamine ammonium	Inhibits bud and leaf formation; broadleaf weeds, brush, and trees.				•	•	•
Glyphosate	Non-selective; annual and perennial grasses and broadleaf weeds, sedges, shrubs, and trees.	•	•	•	•	•	•
Hexazinone	Foliar or soil applied; inhibits photosynthesis; annual and perennial grasses and broadleaf weeds, brush, and trees.	•	•		•	•	•
Imazapyr	Non-selective; preemergent and postemergent uses; absorbed through foliage and roots; annual and perennial broadleaf weeds, brush, and trees.	•	•	•	•	•	•
Mefluidide	Growth inhibitor; suppresses seed production of grasses, brush, and trees.				•	•	•
Metsulfuron methyl	Selective; postemergent; inhibits cell division in roots and shoots; annual and perennial broadleaf weeds, brush, and trees.	•	•		•	•	•
Picloram	Selective; foliar and root absorption; mimics plant hormones; certain annual and perennial broadleaf weeds, vines, and shrubs.	•	•		•	•	•
Simazine	Used selectively or as complete vegetation killer; requires much moisture for activation; inhibits photosynthesis.				•	•	•
Sulfometuron methyl	Broad-spectrum pre- and post-emergent control; inhibits cell division; grasses and broadleaf weeds.		•		•	•	•
Tebuthiuron	Relatively non-selective soil activated herbicide; pre- and post-emergent control of annual and perennial grasses, broadleaf weeds, and shrubs.	•			•	•	•
Triclopyr	Growth regulator; broadleaf weeds and woody plants.	•	•	•	•	•	•
Herbicides Proposed for Use on Public Lands							
Diflufenzopyr + Dicamba	Postemergent; inhibits auxin transport; broadleaf weeds.	•			•	•	•
Diquat	Non-selective and foliar applied.			•	◘	◘	◘
Fluridone	Aquatic herbicide to control submersed aquatic plants.			•			
Imazapic	Selective postemergent herbicide; inhibits broadleaf weeds and some grasses.	•	•		•	•	•

• = Areas where USEPA approved registration exists and the BLM has approval or proposes to use on public lands; ◘ = Areas where USEPA approved registration exists, but where the BLM does not propose to use on public lands.

CHAPTER 3

SPECIAL STATUS SPECIES MANAGEMENT CONSULTATION PROTOCOL

There are typically two “tiers” of action when a federal agency adopts or approves a management plan or strategy that will be used to guide the development and implementation of future projects. The first tier of action involves adopting the broad management plan or strategy, and the second tier involves implementing site-specific actions. Both tiers require consultation under Section 7 of the ESA.

Consultation with the Services is required when any action authorized, funded, or carried out by a federal agency could jeopardize the continued existence of a listed species (jeopardy) or destroy or adversely modify critical habitat (adverse modification). This chapter identifies the steps that will be taken by the BLM at the national and local level to ensure that their actions requiring authorization or approval by the BLM are consistent with guidance provided in the PEIS, PER, this BA, risk assessments (ENSR 2005a-j, Syracuse Environmental Research Associates, Inc. 2005), *Endangered Species Consultation Handbook* (USFWS and National Marine Fisheries Service [NMFS] 1998), BLM Manual 6840 (*Special Status Species Management*), BLM Handbook H-1601-1 (*Land Use Planning Handbook*), consultation with the USFWS and NOAA Fisheries as part of the preparation of the EIS and BA, Memorandum of Agreement among the BLM, NOAA Fisheries Service, and USFWS (USDI BLM 2002) in order to streamline the Section 7 consultation process, and the ESA. In particular, the focus of this protocol is to ensure that any action authorized, funded, or carried out by the BLM will not jeopardize the continued existence of any listed species or result in the destruction or adverse modification of critical habitat of such species. If followed, these steps should ensure that the conservation needs of TEP species and other special status species are met.

This BA, the PEIS, and the PER evaluate the potential for vegetation treatment programs conducted by the BLM in the western U.S., including Alaska, to affect listed and proposed species and designated and proposed critical habitat. These documents establish standards, guidelines, and design criteria to which future vegetation treatment actions must adhere. Programmatic consultation increases the efficiency of the Section 7 consultation process because much of the effects analysis is completed up-front and that the effects of future actions are broadly accounted for. For example, much of the analysis of the effects of the use of herbicides on species of concern has been completed as part of this BA and risk assessments*;* this information can be incorporated into the baseline assessment for local projects. Programmatic consultation also minimizes the potential “piecemeal” effects than can occur when evaluating individual projects out of context of the complete agency program.

Programmatic Level Consultation

The BLM began consulting with the Services beginning in November 2001 as part of development of the PEIS and BA. As part of first phase of consultation, the Services will develop a Programmatic BO that analyzes the potential landscape-level effects that may result from implementing the proposed action. For the PEIS, PER, and BA, there is substantial temporal and spatial uncertainty regarding future actions, resulting in corresponding uncertainty regarding potential effects. As a result, a second phase is required that involves development of appropriate project-specific documentation that addresses the specific effects of individual projects proposed by BLM field offices. Upon completion of the project-specific review, the associated documentation will be appended to the Programmatic BO.

An important feature of the first phase of consultation is the development of design criteria or standards that can be used to guide future projects. Design criteria are developed through a five-step process:

- Identify the conservation needs of each species.
- Identify the threats to each listed species.
- Identify the species conservation or management unit.
- Identify the species conservation goals within the context of the BLM's programs and authorities.
- Develop conservation/management strategies for implementing future activities (design criteria; conservation measures).

These five elements have been incorporated into this BA This BA helps to streamline the consultation process by completing a portion of the effects analysis early in the consultation process, and providing conservation measures that reduce potential adverse effects to listed species and which will be applied agency-wide.

Local Level Consultation

Informal Consultation

Most consultations for proposed actions will first be conducted informally between the BLM and Services. During informal consultation, the BLM will:

- Determine whether TEP species or critical habitat occurs within the proposed action area based on BLM databases or species lists requested from the Services.
- Conduct site assessments and additional studies in the action area to determine whether TEP species are present if the status of TEP species in the area is unknown.
- Determine what effect the action may have on TEP species or critical habitats.
- Identify ways to modify the action to reduce or prevent adverse effects to TEP species or critical habitats, including taking actions identified in this BA, using other treatment methods, or scheduling treatments for times of the year when TEP species are not present in the action area.
- Prepare a BA if TEP species or critical habitat may be present in the action area for any action that is likely to affect TEP species,.
- Obtain written concurrence of this determination from the Services if the BLM determines that the proposed action may affect but is not likely to adversely affect TEP species or designated or proposed critical habitat.

If modifications to the project cannot be made and the proposed action is likely to adversely affect TEP species or critical habitat; if there are undetermined effects; or if the BLM's determination of not likely to adversely affect is not based on a BA or has no written concurrence from the Services, then the BLM shall initiate formal Section 7 consultation.

Formal Consultation

Formal consultations determine whether a proposed agency action is likely to jeopardize the continued existence of a listed species (jeopardy), or destroy or adversely modify critical habitat (adverse modification). They also determine the amount and extent of anticipated incidental take in an incidental take statement. The formal consultation process results in a BO reaching either a jeopardy or no jeopardy to listed species (or adverse or no adverse modification of critical habitat) finding.

Formal consultation is initiated with submission of a BA and a written request to initiate formal consultation (initiation package). The BA and supporting documentation must include all of the following:

- A description of the proposed action
- A description of the area that may be affected by the action
- A description of any listed species or critical habitat that may be affected by the action
- A description of the manner in which the action may affect any listed species or critical habitat, and an analysis of any cumulative effects
- Relevant reports, including EISs, EAs, BAs, or other analyses prepared on the proposal
- Other relevant studies or other information available on the action, the affected listed species, or critical habitat

Within 30 days of receipt of an initiation package, the Services will provide written receipt of the consultation request, advise the BLM of any data deficiencies, and request either the missing data or a written statement that the data are not available. Section 7 regulations require that formal consultation be concluded within 90 days of receipt of all required data, and that a BO be delivered to the BLM within 45 days after conclusion of formal consultation.

Although additional surveys are not required under the ESA, the BLM may conduct surveys to better address listed species issues. If the Services conclude that the action is likely to jeopardize the continued existence of a TEP species or will result in the destruction or adverse modification of designated critical habitat, the Services will prepare a BO that identifies the availability of any reasonable and prudent alternatives. For example, the Services may determine that one or more treatment methods, including the use of herbicides, may not be appropriate for use; that not all areas proposed for treatment can be treated; or that mitigation measures must be implemented to ensure that treatment actions do not jeopardize species or critical habitat. The BLM will provide expertise to the Services in determining the availability and development of reasonable and prudent alternatives. The BO may also include an incidental take statement, with which the BLM shall comply. Incidental take statements allow the BLM to take actions, as defined by section 3 (19) of the ESA that could “harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct” toward TEP species. However, the taking must not jeopardize the continued existence of listed species or destroy or adversely modify designated critical habitat; must result from an otherwise lawful activity; and must be incidental to the purpose of the action.

BLM Responsibilities After Issuance of the Biological Opinion

After the Services issue the BO, the BLM shall notify the Services in writing of its final decision on any proposed actions that receive a jeopardy or adverse modification of critical habitat determination. If the BLM determines that it cannot comply with the requirements of Section 7(a)(2) (no jeopardy) of the ESA, it may apply for an exemption.

If the BLM accepts the BO, it will implement the proposed action or reasonable and prudent alternative. The BLM will review conservation recommendations in the BO and implement them if they are consistent with BLM land use planning and policy and are technologically and economically feasible.

CHAPTER 4

PLANTS

Background Information

This BA considers a total of 151 plant species that are listed as threatened or endangered, or that are proposed for listing. For this background discussion, these species have been arranged on the basis of the ecoregions (Bailey 1995) in which they are located. These divisions provide groupings that consider both geography and broad habitat types, and are the same divisions used for much of the analysis in the PEIS.

Most of the information contained in this section was obtained directly from Federal Register documents, species recovery plans, biological assessments and evaluations, and other sources of information. Where primary reference(s) was/were used for species background and listing information, full citations are listed in the individual sections for each species. In some instances, citations were used from the primary reference(s), and the complete citations were not available from the primary reference(s) for inclusion in the Bibliography (Chapter 8). In the instances where complete citations were not available, information is listed in the individual sections on where there complete citations can be found (e.g., USFWS Sacramento Field Office, Sacramento, California). If information is not listed on the location of complete citations from the primary reference(s), then the complete citation can be found in the Bibliography.

Temperate Desert Ecoregion Division

The Temperate Desert Ecoregion Division includes the arid lands located in the rain shadow of the Pacific mountain ranges. Portions of the Great Basin, Columbia Plateau, and the Wyoming Basin are found in this ecoregion division, which supports vegetation that is adapted to summer droughts and cold winters. Plant communities occurring in the Temperate Desert Division include sagebrush steppe, perennial grasslands, evergreen (mostly pinyon-juniper) woodlands, deciduous shrublands (found in the Great Basin and deserts of the southwest), and evergreen forests.

Malheur Wire-lettuce
The primary reference for this section is:
Hudson, B., J. Augsburger, M. Hillis, and P. Boehne. 2000. Draft Biological Assessment for the Interior Columbia River Basin Ecosystem Management Project Final Environmental Impact Statement. BLM and Forest Service. Boise, Idaho.

Malheur wire-lettuce (*Stephanomeria malheurensis*) is an annual plant that is found at only one 70-acre location near Malheur National Wildlife Refuge in Harney County, Oregon. This population is found within the high desert environment typical of the northern portion of the Great Basin, on top of a dry, broad hill. The substrate at this location is an azonal soil derived from the volcanic tuff layered with thin crusts of limestone. By contrast, the surrounding soils are derived from basalt. The top of the hill is about 500 feet above the surrounding flats, which consist of sagebrush-rabbitbrush desert. The immediate site itself is dominated by big sagebrush, common or gray rabbitbrush, and downy brome. Malheur wire-lettuce appears to be one of the few species that is able to survive on and around the otherwise barren harvester ant hills at the site. The area has been fenced to protect the population.

Because the species is an annual, the numbers of plants vary greatly from year to year, and depend largely on the amount of precipitation received prior to and during the spring growing season. Seeds germinate in the fall after a late summer / early fall rain.

The Malhuer wire-lettuce was federally listed as endangered on November 10, 1982, and critical habitat was designated to include the 160-acre Scientific Study Area on public land administered by the BLM, located 27 miles south of Burns in Harney County, Oregon. Because of its extremely restricted range and low numbers, this species is vulnerable to even small land disturbances in and around its habitat. Potential future zeolite mining in the area also endangers the continued existence of this species. Other threats to this species that have been identified include competition with downy brome, grazing by native herbivores, and possible foraging by beetle larvae.

Desert Yellowhead
The primary reference for this section is:
USFWS. 2002A. Listing the Desert Yellowhead as Threatened. Federal Register 67 (50): 11442-11449.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The desert yellowhead (*Yermo xanthocephalus*) is a recently described endemic to the south end of Cedar Rim on the summit of Beaver Rim in southern Fremont County, Wyoming. The species is restricted to shallow deflation hollows in sandstone outcrops of the Split Rock Formation (Van Houten 1964). These wind-excavated hollows accumulate drifting snow and may be moister than surrounding areas. The vegetation of these sites is typically sparse, consisting primarily of low-cushion plants and scattered clumps of Indian ricegrass.

The desert yellowhead is known from a single population occupying an area of less than 5 acres of suitable habitat. This population is located in the BLM's Lander Resource Area, which is rich in locatable mineral resources, such as gold, copper, and uranium.

The desert yellowhead is a tap-rooted, perennial herb. Flower heads are numerous (25 to 180) and are crowded at the top of the stem. The species flowers and fruits in the spring and summer.

The desert yellowhead was federally listed as threatened on March 14, 2002. On March 16, 2004, the USFWS designated approximately 360 acres in Fremont County, Wyoming, as critical habitat. This species is threatened by surface disturbances associated with oil and gas development, compaction by vehicles, trampling by livestock, and randomly occurring, catastrophic events.

Steamboat Buckwheat
The steamboat buckwheat (*Eriogonum ovalifolium* var. *williamsiae*) is a shrub that occurs most commonly on open slopes in gravelly, sandy-clay soil that is derived from hot springs deposits around the Steamboat Springs geothermal area, 10 miles south of Reno, Nevada. The associated plant community is desert shrub, and commonly includes saltbush, greasewood, rubber rabbitbush, snakeweed, and desert saltgrass. The habitat varies from 4,580 to 4,720 feet in elevation. The buckwheat occurs in distant patches, some including only a few individuals and some with several thousands individuals, scattered over an area of less than 100 acres (Williams 1982, CH2M Hill 1986a). Steamboat buckwheat does not appear to grow on moist soils or to receive supplemental moisture from thermal water, and may not tolerate high moisture conditions and associated high levels of sodium, potassium, and chloride. However, it may receive adequate moisture from rainfall to survive in at least some portions of its range.

The steamboat buckwheat tends to be the most common plant in the scattered, specific areas where it occurs. Few other species seem to occur in the gravelly, incompletely developed soils where the buckwheat flourishes. With eventual development of more soil on these sites, other plants are able to occupy the site and out-compete the buckwheat, which then declines or disappears completely in some sites (CH2M Hill 1986a).

The steamboat buckwheat grows in low, compact, woody mounds up to 18 inches across, covered with rosettes of small leaves (Nevada Division of Forestry, no date). Pink flowers appear on leafless stems from May through July, clustered into tight balls at the tips. The reproductive biology of the species is not well understood. Although each plant may produce hundreds of seeds, germination may be less than 1%. New plants grow from seeds, and may also grow from the roots of existing plants. Butterflies are potential pollinators.

The steamboat buckwheat was federally listed as endangered on July 8, 1986. Critical habitat has not been designated. The primary threat to the species is private development. The remaining (and largest) part of the population, however, is potentially protectable, but faces continued threats because of its location in an intensely developed area along a major highway (NatureServe 2001). Development would quickly destroy the plants. In addition, illegal OHV use and refuse dumping have occurred; these activities may alter moisture patterns, a habitat parameter to which this plant is especially sensitive. Because of the plant's low reproductive potential, any substantial loss of individuals may severely affect its survival.

Slickspot Peppergrass

The primary reference for this section is:

USFWS. 2002b. Listing the Plant *Lepidium papilliferum* (slickspot peppergrass) as Endangered. Federal Register 67(135): 46441-46450.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Snake River Basin Office, Boise, Idaho.

Slickspot peppergrass (*Lepidium papilliferum*) occurs in semi-arid sagebrush-steppe habitats in southwestern Idaho, at elevations of approximately 2,200 to 5,400 feet. This species is found along the Snake River Plain and Owyhee Plateau in Ada, Canyon, Gem, Elmore, Payette, and Owyhee Counties. Plants are restricted to small areas, similar to vernal pools, known as slickspots (also called mini-playas or natric sites). Slickspots range from less than 10 square feet to about 110 square feet, within communities dominated by other plants (Mancuso et al. 1998). Slickspot peppergrass is limited to slickspots covering a relatively small area. These sparsely vegetated microsites are very distinct from the surrounding shrubland vegetation, and are characterized by relatively high concentrations of clay and salt (Fisher et al. 1996). The microsites also have reduced levels of organic matter and nutrients due to a lower biomass production, as compared to surrounding habitat areas. Associated native species include Wyoming big sagebrush, basin big sagebrush, bluebunch wheatgrass, Thurber's needlegrass, Sandberg's bluegrass, and bottlebrush squirreltail. Non-native species frequently associated with slickspot peppergrass include downy brome, tumble mustard, bur buttercup, clasping pepperweed, and crested wheatgrass (Moseley 1994; Mancuso and Moseley 1998).

The restricted distribution of slickspot peppergrass is likely a product of the scarcity of suitable habitat, which is extremely localized, and the loss and degradation of suitable habitat areas throughout southwestern Idaho. Occurrences of the species can include one to several occupied slickspots within an area determined to be suitable habitat. The total amount of habitat containing interspersed slickspots that have extant occurrences of slickspot peppergrass is about 12,356 acres. Of 88 known occurrences of the species, 70 are currently extant (exist), 13 are considered extinct, and 5 are historic (Moseley 1994, Mancuso 2000, Idaho Conservation Data Center [ICDC] 2002). Only 6 of the 70 extant occurrences are considered to be high-quality habitat and contain large numbers of the plants (ICDC 2002).

Slickspot peppergrass is an annual or biennial plant in the mustard family that reaches 4 to 12 inches in height. Numerous small, white flowers terminate the branches. Slickspot peppergrass is mainly pollinated by bees (Apidae, Colletidae, and Halictidae families), flies (Syrphidae family), and some beetle species (Dermestidae and Cerambycidae families; Robertson 2001). This species produces small, spherical fruits (siliques), which are approximately 3 millimeters long. The primary seed dispersal mechanism is probably gravity, although wind and water may have a minor role (Moseley 1994). Slickspot peppergrass seeds may be viable in the soil for up to 12 years (Quinney 2002). Like many short-lived plants growing in arid environments, the above-ground number of individuals at any one site can fluctuate widely from one year to the next depending on seasonal precipitation patterns (Mancuso and Moseley 1998, Mancuso 2001). Flowering individuals represent only a portion of the population, with the seed bank contributing the remainder, and apparently the majority, in many years (Mancuso and Moseley 1998). For annual plants, maintaining a seed bank is important for year-to-year and long-term survival (Baskin and Baskin 1978). A seed bank includes all of the seeds in a population and generally covers a larger area than the extent of observable plants seen in a given year (Given 1994).

Slickspot peppergrass was proposed for listing as an endangered species on July 15, 2002. The USFWS proposed designating critical habitat for the species in the future, though not at the time of listing. This species is threatened by a variety of activities including urbanization, gravel mining, irrigated agriculture, habitat degradation due to cattle and sheep grazing, fire and fire rehabilitation activities, and continued invasion of habitat by non-native plant species (Moseley 1994, Mancuso and Moseley 1998). Much of the habitat for slickspot peppergrass occurs within a matrix of sagebrush-steppe, a community in which displacement of native plants by non-native species is a major problem (Rosentreter 1994; DeBolt pers. com 1999 cited in Office of Species Conservation 2002). Widespread grazing by livestock in the late 1800s and early 1900s severely degraded sagebrush-steppe habitat, enabling introduced annual species (especially downy brome) to become dominant over large portions of the Snake River Plain (Yensen 1980, Moseley 1994). The invasion of downy brome has shortened the fire frequency of the sagebrush-steppe from between 60 to 110 years, to less than 5 years, as it provides a continuous, highly flammable fuel through which a fire can easily spread (Whisenant 1990, Moseley 1994, Mancuso and Moseley 1998). The result has been the permanent conversion of vast areas of the former sagebrush-steppe ecosystem into non-native annual grasslands. The continued cumulative effects of overgrazing and fire suppression permit the invasion of non-native plant species into slickspot habitats (Rosentreter 1994). Slickspot peppergrass populations typically decline or are extirpated following the replacement of sagebrush-steppe habitat by non-native annuals. Another problem has been the use of non-native perennial species to restore or rehabilitate shrub-steppe habitat after a fire event. Although some slickspot peppergrass plants may temporarily persist in spite of these restoration seedings, most occurrences support small numbers of plants (fewer than five per slickspot) and long-term persistence data are unavailable (Mancuso and Moseley 1998). Habitat degradation, fragmentation, and loss of sagebrush-steppe vegetation have occurred throughout the range of the species.

Fish Slough Milk-vetch

The primary reference for this section is:

USFWS. 1998a. Determination of Endangered or Threatened Status for Five Desert Milk-vetch Taxa from California. Federal Register 63 (193): 53596-53615.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Ventura Field Office, Ventura, California.

Fish Slough milk-vetch (*Astragalus lentiginosus* var. *piscinensis*) is a prostrate perennial with lavender flowers arranged in loose, short racemes. The plant is found growing in a 6-mile stretch of alkaline flats paralleling Fish Slough, a desert wetland ecosystem in Inyo and Mono counties, California. It grows in seasonally moist alkaline flats that support a cordgrass-dropseed association, and is absent from nearby lower areas that are seasonally flooded (Ferren 1991a, 1992). Appropriate alkali habitat covers less than 540 acres of the slough, and portions of this area do not currently support the species, for unknown reasons (Ferren 1991a; Odion et al. 1992). Over 60% of Fish Slough milk-vetch plants are located in the northern portion of the slough, on land administered by the Los Angeles Department of Water and Power, and approximately 35% are in the central zone of the slough, on lands administered by both the BLM and the Los Angeles Department of Water and Power. The remaining 5% are in scattered patches downstream as far as McNally Canal. Grazing is not permitted in the habitat of Fish Slough milk-vetch on lands administered by the BLM.

Fish Slough milk-vetch was federally listed as threatened on October 6, 1998. On June 4, 2204, the USFWS proposed designating approximately 8,490 acres in Mono and Inyo counties, California. Current threats to the species include a lack of recruitment in the central zone population, trampling and grazing by cattle, modification of wetlands, and alteration of slough hydrology. A long-term threat may be the expansion of Fish Slough Lake, caused by natural geologic processes or the existence of Red Willow Dam, resulting in increased inundation of soils and loss of suitable alkali habitat for this taxon (Ferren 1991b, 1992). Historical alterations of the Fish Slough ecosystem to enhance fisheries appear to have caused similar increases in seasonally flooded habitats, which are less suitable for Fish Slough milk-vetch. Modifications include creation of dams and weirs in the main slough channel, construction of a dirt road through milk-vetch habitat, and soil compaction and trail creation by cattle. These activities have altered the slough hydrology by causing an increase in permanently flooded habitats, artificial ponding, alteration in drainage patterns, and changes in seasonal flooding of milk-vetch habitat. These changes have in turn resulted in expansion of emergent wetland vegetation and conversion of alkali flat habitats (Ferren 1991c, 1992).

Autumn Buttercup

The primary reference for this section is:

USFWS. 1991a. Autumn Buttercup (*Ranunculus acriformis* var. *aestivalis*) Recovery Plan. USFWS. Denver, Colorado.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The autumn buttercup (*Ranunculus aestivalis*) is restricted to perennially moist soils in wet meadows along the Sevier River, Garfield County, Utah. A single known population grows along the margin of a spring-fed wet meadow at an elevation of 6,440 feet, on an east facing slope. The habitat surrounding the population is grassland, with the autumn buttercup occurring on hummocks at the transition zone between a wet sedge-dominated community and a dry upland meadow. Common associated species include beaked spikerush, aster, Nebraska sedge, sea milkwort, Baltic rush, alkali buttercup, and darkthroat shootingstar.

The autumn buttercup is a perennial herb that typically grows to a height of between 1 and 2 feet. Reproduction of the species is by seed. Plants complete their life cycle of flowering to producing seed between late July and early September. Seeds are generally dispersed in close proximity to the parent plant, though they could be transported by animals and water. Flowers are likely pollinated by insects and/or wind.

The autumn buttercup was federally listed as endangered on July 21, 1989. Critical habitat has not been designated for the species. The species is apparently highly vulnerable to grazing from domestic livestock, as well as other mammals (e.g., rodents, rabbits, and possibly deer). Modification of the hydrologic regime of the species' habitat could also affect plants. In addition, the buttercup's small population and restricted habitat make it very vulnerable to any adverse impact to plants or their habitat.

Clay-loving Wild-buckwheat

The clay-loving wild-buckwheat (*Eriogonum pelinophilum*) occurs in Delta and Montrose counties of western Colorado, growing exclusively on substrates high in salt and gypsum derived from the Mancos Shale. This saline, calcareous, cretaceous deposit outcrops to form nearly barren adobe (clay) hills. Thus, the soils are typically clays, or have a high clay content, and while having the potential for a high moisture holding capacity, have little available moisture. The lack of available moisture is exacerbated by the low rainfall in the region. Because of intense competition for water, the habitat is sparsely vegetated. Species able to survive here are xerophytic (drought tolerant), with primarily woody prostrate or low-growing shrubs as dominants: mat saltbush, shadscale, valley saltbush, black sagebrush, and horsebrush. Herbaceous species include winter-fat, wildrye, and wheatgrass. The clay-loving wild-buckwheat prefers swales and bottoms, on all aspects, where the competition for water is somewhat less severe. When found, the species is codominant with other xerophytic shrubs or subshrubs. There are several streams and creeks running throughout the habitat of the clay-loving wild-buckwheat, as are roads and highways. The elevation ranges from 5,180 to 6,240 feet, with an average of 5,764 feet.

The clay-loving wild-buckwheat is a perennial woody subshrub. Leaves of this species begin to appear during the last week in April and into the first week in May. Flowers bloom from June through August, and fruit appears anywhere from late June to August. Seed dispersal occurs during late July and August (Reveal 1973, Peterson 1982, 1985, Neese 1984, O'Kane 1985). Seeds of wild-buckwheat species are usually dispersed through passive means, either by being consumed or carried by animals, windblown, or moved by gravity or water. Often, seeds are moved intact in the dying flower. Flowers are produced over a long period of time; therefore, brief events are not likely to substantially reduce seed production. Nearly every flower will produce a seed. Habitat severity and a lack of invading species capable of dominating the sites indicate that the communities occupied by the clay-loving wild buckwheat are stable, climax associations. Reproduction appears to occur as senescent individuals die. Substantial reproductive episodes may occur during relatively wet years. No data are available on causes of mortality other than observations made on human induced habitat destruction or alteration.

The clay-loving wild-buckwheat was federally listed as endangered on July 13, 1984. Critical habitat has been designated in Delta County, Colorado. This designation includes an area 3 miles east of Austin near Highway 92, located in portions of Sections 26, 27, 28, 34, and 35 Township 14 South, Range 94 West. Although the clay-loving wild-buckwheat occurs in sparsely populated area of Colorado, it is exposed to numerous threats. Much of the habitat for this species has been converted to alfalfa fields, and residential sites with accompanying barns, pastures, and corrals. Expansion of the population base in the Montrose and Delta areas has caused residential encroachment onto habitats previously occupied by the plant. Tracts of land not directly influenced by homes or pastures is subject to heavy domestic livestock grazing. These clays, which are easily eroded, are especially impacted during wet periods when large and deep impressions are made in the soil by animal hooves. In addition, the known habitat is dissected by roads (paved and unpaved) and railroads. The adobe hills, the primary habitat, are subject to a great amount of OHV use. The sparsely vegetated hills are also prone to severe erosion, as evidenced by deep rills on those hills receiving the heaviest OHV use. The land between Montrose and Delta has an exceptionally dense concentration of irrigation canals and ditches for water diversion. Intensified agricultural uses will necessitate an increased loss of habitat to irrigation projects (O'Kane 1985). Finally, given the nature of the Mancos Shale, and underlying strata, the area has a high potential for oil and gas development. Should the need for these commodities and for gypsum, another component of the adobe hills, increase, the buckwheat's habitat will be subject to use of heavy equipment for oil and gas exploration as well as surface mines for gypsum.

Uinta Basin Hookless Cactus
The primary reference for this section is:
USFWS. 1990a. Uinta Basin Hookless Cactus (*Sclerocactus glaucus*) Recovery Plan. USFWS. Denver, Colorado.

The Uinta Basin hookless cactus (*Sclerocactus glaucus*) is a regional endemic to western Colorado and adjacent Utah. The species is generally found on cobbley, gravelly, or rocky surfaces on river terrace deposits, at an elevation of 4,500 to 5,900 feet. Plants occur on varying exposures, but are most abundant on south-facing exposures, and on exposures to about 30% grade. The Uinta Basin hookless cactus occurs in desert scrub communities dominated by shadscale, galleta, black sagebrush, and Indian ricegrass. Other important species include strawberry hedgehog cactus, and Simpson's pincushion cactus. The distribution of the Uinta Basin hookless cactus includes one major population center in the Uinta Basin of northeastern Utah, and two population centers in the upper Colorado and Gunnison River valleys of western Colorado. There is no evidence that the range of this species is any more restricted today than in the recent past.

Reproduction in the Uinta Basin hookless cactus is sexual, with flowering occurring in April and May, and fruiting occurring in May and June. Bees, flies, beetles, and ants have been observed visiting flowers, though it is not known which of these insects are effective pollinators. Seeds are small and dense, with no surface structures for facilitating dispersal; rather, they are dispersed by water, gravity, water flow, or possibly by insects and/or birds. Seed dispersal is probably a limiting factor in the distribution of the species.

The Uinta Basin hookless cactus was federally listed as threatened on October 11, 1979. Critical habitat has not been designated. Realized and potential threats to the species stem primarily from mineral and energy development, water development, and plant collecting. Other potential threats include OHV use and recreational impacts, road building and maintenance, and pesticide use.

Wright Fishhook Cactus
The primary reference for this section is:
USFWS. 1985b. Wright Fishhook Cactus Recovery Plan. Prepared in Cooperation with the Wright Fishhook Cactus Recovery Committee. USFWS. Denver, Colorado.

References cited in this section are internal to the above-referenced Federal Register document. They are included in the Bibliography.

The Wright fishhook cactus (*Sclerocactus wrightiae*) occurs in the Canyonlands section of the intermountain region in Utah (Holmgren 1972), an area of relative geological stability and high plant endemism. The range of the species follows a low elevation trough around the south end of the San Rafael Swell uplift between the Swell and the Wasatch Plateau, Thousand Lake Mountain, and the Henry Mountains. Plants occur primarily on arid sites with widely spaced shrubs, perennial herbs, bunchgrasses, or scattered pinyon and juniper that provide very little surface coverage. Plant community types are salt desert shrub and pinyon-juniper, with the following associated species: pinyon pine, Utah juniper, valley saltbush, shadscale saltbush, mat saltbush, and galleta.

The Wright fishhook cactus may be found on a variety of soil types of several geologic formations, ranging from clays to sandy silts and fine sands. Populations are known from areas both with little or no gypsum and from areas with well-developed gypsum layers. Soils at most of the sites possess a surface structure with at least some cryptogramic crust. Plants are rare or absent where the cryptogramic crust has been destroyed or is undeveloped. Sites are usually littered with sandstone or basalt gravels, cobbles, and boulders. Both the surface and rock litter may aid in water infiltration and provide safe sites for germination and seedling establishment.

Reproduction of these small cacti is primarily by seed. Plants begin to flower when they are quite small and, presumably, young. Flowers form on the new growth of the current year. From one to several white to pale pink blossoms cluster at the top of each small barrel. Pollinators may include beetles and ants. Fruits mature in June, and seeds are generally dispersed near the parent plant, though they may be transported by water or animals. Seedling plants are often collected inadvertently in organic detritus clinging to adult plants. Budding, in which small cacti form at the base of an adult, also contributes to the population. As the summer progresses, and drought stress increases, the cacti shrink and become almost level with the ground surface.

The Wright fishhook cactus was federally listed as endangered on October 11, 1979. Critical habitat has not been designated. Factors that threaten this species include illegal collection, development related to the coal industry, OHV use, road upgrading, and cattle grazing. Because the Wright fishhook cactus appears to be associated with the presence of a well-developed cryptogramic crust, it is threatened by any activity in which the cryptogramic crust is removed.

Barneby Ridge-cress
The primary reference for this section is:
USFWS. 1993a. Barneby Ridge-cress (*Lepidium barnebyanum*) Recovery Plan. USFWS. Denver, Colorado.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The Barneby ridge-cress (*Lepidium barnebyanum*) occurs in a discontinuous series of marly shale barrens on three ridgelines in Duchesne County, Utah. Plants occur at elevations of 6,200 to 6,500 feet on either side of Indian Creek, with the total known population of the species located within the Uintah and Ouray Reservation of the Ute Indian Tribe. The soil characteristics of Barneby ridge-cress habitat are not common within the species' range, and effectively form "islands" of suitable habitat within a "sea" of unsuitable soil types derived from other differing geologic substrates. The abundance and distribution of the species is limited by its restrictive habitat.

The vegetation of the shale barrens on which the Barneby ridge-cress occurs is dominated by the stemless four-nerve daisy, Hooker's sandwort, table Townsend daisy, Colorado feverfew, and the Barneby ridge-cress itself. Other associated plant species include Bateman's buckwheat, tufted milk-vetch, and rough Indian paintbrush. The shale barren plant community is a small inclusion within the broader pinyon-juniper (pinyon pine and Utah juniper) woodland community that characterizes the general area (Welsh 1978a, USFWS 1989).

The Barneby ridge-cress reproduces entirely by sexual reproduction. Flowering occurs from April to May, and fruiting occurs from May to June. The specific pollination mechanism and vectors are not known.

Barneby ridge-cress was federally listed as endangered on September 28, 1990. Critical habitat has not been designated. The species is vulnerable to any event that could cause the local extirpation of one or more of its isolated stands within its only known population. Past, existing, and potential threats to the species and its habitat include oil and gas exploration, drilling, and production; OHV use; and grazing. The remaining population of the Barneby ridge-cress is underlain by petroleum deposits that are currently being developed. The 1993 recovery plan for the species indicated that continued OHV use and the future development of oil and gas wells and ancillary facilities could lead to extinction of the species in the absence of appropriate measures to protect the species and its habitat.

Deseret Milk-vetch

The primary reference for this section is:

USFWS. 1999a. Final Rule to List *Astragalus desereticus* (Deseret Milk-vetch) as Threatened. Federal Register 64(202): 56590-56596.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The Deseret milk-vetch (*Astragalus desereticus*) is a perennial, nearly stemless herb in the bean family. The only known population of the species occurs in Utah County, Utah, primarily on steep south- and west-facing slopes. The plant grows on soils derived from a specific and unusual portion of the geologic Moroni Formation, which is characterized by coarse, crudely bedded conglomerate (M.A. Franklin 1990). The plant community in which the deseret milk-vetch occurs is dominated by pinyon pine and Utah juniper. Other associated plant species include: sagebrush, scrub oak, wild buckwheat, Indian ricegrass, needle and thread grass, bitterbrush, and plateau beardtongue. The sole population of the species consists of between 5,000 and 10,000 individuals, and covers an area of less than 300 acres (M.A. Franklin 1990, Stone 1992). The species' total range is approximately 1.6 miles long, and 0.3 miles across. The land upon which the desert milk-vetch grows is owned by the State of Utah and three private land owners (M.A. Franklin 1990, 1991).

Individual plants are approximately 2 to 6 inches in height, and arise from a caudex (the persistent base of an otherwise annual herbaceous stem). The species' flowers are white in color with a purple tip on the keel, and borne on a stalk of 5 to 10 flowers. Bumblebees are thought be the primary pollinators of flowers. The fruit is a seed pod.

In 1975, the deseret milk-vetch was presumed to be extinct. In 1981, a population of the species was discovered. The deseret milk-vetch was federally listed as threatened on October 20, 1999. Critical habitat has not been designated. The species is threatened by grazing and trampling by ungulates, alteration of its habitat due to residential development and road widening, and natural events, such as fire, due to its limited distribution.

San Rafael Cactus

The primary reference for this section is:

Utah Conservation Data Center. No Date. Fact Sheet for San Rafael Cactus. State of Utah Natural Resources, Division of Wildlife Resources. Available at http://utahdc.usu.edu.

The San Rafael cactus (*Pediocactus despainii*) is a narrow endemic that is limited to Emery County in central Utah. The species is found in fine textured soils rich in calcium that are derived from the Carmel Formation and the Sinbad Member of the Moenkopi Formation. Plants occur on benches, hilltops, and gentle slopes in pinyon-juniper and mixed desert shrub-grassland communities, at elevations ranging from approximately 4,800 to 6,800 feet. In 1998, only two populations were known, and the total number of individuals was estimated at 6,000. This species is a small, subglobose to ovoid cactus, with flowers that are born near the tip of the stem during April and May.

The San Rafael cactus was federally listed as endangered on September 16, 1987. Critical habitat has not been designated. The habitat of this species is vulnerable to surface disturbance from OHV use, trampling by humans and livestock, and by mineral resource explorations and development. The limited habitat and small population size make it especially vulnerable to extinction by natural or human-induced habitat disturbances. The species is also highly desirable to cactus collectors, and illegal collection is a threat.

Clay Reed-mustard
The primary reference for this section is:
USFWS. 1994a. Utah Reed-mustards: Clay Reed-mustard (*Schoenocrambe argillacea*), Barneby Reed-mustard (*Schoenocrambe barnebyi*), Shrubby Reed-mustard (*Schoenocrambe suffrutescens*) Recovery Plan. USFWS. Denver, Colorado.

References cited in this section are internal to the above-referenced Federal Register document. They are included in the Bibliography.

The clay reed-mustard (*Schoenocrambe argillacea*) occurs on fine-grained soils in desert shrub in the Colorado River drainage of eastern Utah. It is found in the south-central Uintah Basin near the Green River in Uintah County, Utah. The clay reed-mustard grows on clay soils that are rich in gypsum and overlain with sandstone talus, and that are derived from a mixture of shales and sandstones. This species occurs on steep, usually north-facing slopes in mixed desert shrub communities at elevations ranging from approximately 4,720 to 5,790 feet. Common associates include: Utah serviceberry: western wheatgrass; black sagebrush; Mojave bricklebush; wavyleaf Indian paintbrush; yellow rabbitbrush; Rollins' cryptantha; saline wildrye; granite prickly phlox; fleshy beardtongue, grassy rock-goldenrod, turpentine wavewing, Indian ricegrass, Navajo tea, and various species of rock-cress, milk-vetch, horsebrush, buckwheat, and saltbush (Shultz and Mutz 1979, Franklin 1992). All known populations of the species occur within a limited range of about 19 miles across, from the west side of the Green River to the east side of Willow Creek in southwestern Uintah County, Utah. These populations occur on land administered by the BLM.

The clay reed-mustard reproduces entirely by sexual means. Flowering occurs from April to May, and fruiting occurs from May to June. Possible pollinators include native bee species.

The clay reed-mustard was listed as threatened on January 14, 1992. Critical habitat has not been designated. Threats to the species include oil and gas exploration, drilling, and production, oil-shale mining and processing, building stone removal, and OHV use. All known populations of the clay reed-mustard are on federal lands leased for oil and gas energy reserves. The species is also vulnerable to surface disturbance associated with energy developments within its habitats (USFWS 1990b). Trampling by livestock is also a potential threat.

Barneby Reed-mustard
The primary reference for this section is:
USFWS. 1994a. Utah Reed-mustards: Clay Reed-mustard (*Schoenocrambe argillacea*), Barneby Reed-mustard (*Schoenocrambe barnebyi*), Shrubby Reed-mustard (*Schoenocrambe suffrutescens*) Recovery Plan. USFWS. Denver, Colorado.

References cited in this section are internal to the above-referenced Federal Register document. They are included in the Bibliography.

The Barneby reed-mustard (*Schoenocrambe barnebyi*), like the clay reed-mustard discussed in the previous species account, occurs on fine-grained soils in desert shrub in the Colorado River drainage of eastern Utah. The species occurs in two populations: one in the San Rafael Swell of Emery County, Utah, and the other in Capital Reef National Park in Wayne County, Utah. The Barneby reed-mustard grows on red clay soils rich in selenium and gypsum, overlain with sandstone talus, that are derived from the Moenkopi and Chinle formations. Plants occur on steep slopes, and usually occupy northern exposures. Typical habitat for this species is sparsely vegetated sites in mixed desert shrub and pinyon-juniper communities, at elevations ranging from approximately 4,790 to 6,510 feet. Associated plant species include snowball sand verbena, Utah serviceberry, tarragon, Brandegee's milk-vetch, shadscale saltbush, rabbitbrush, Torrey's jointfir, Mormon tea, crispleaf buckwheat, woollygrass species, gallenta, plains pricklypear, dropseed, desert princesplume, and hoary Townsend daisy.

The Barneby reed-mustard reproduces entirely by sexual means. Flowering occurs from April to May, and fruiting occurs from May to June. Possible pollinators include native bee species.

The Barneby reed-mustard was federally listed as endangered on January 14, 1992. Critical habitat has not been designated. The Barneby reed-mustard is threatened by habitat destruction associated with potential uranium mining activity. In addition, the species' highly restricted distribution and very small population make it particularly vulnerable to any activities that would disturb its habitat (Spence 1991, Heil 1992).

Shrubby Reed-mustard
The primary reference for this section is:
USFWS. 1994a. Utah Reed-mustards: Clay Reed-mustard (*Schoenocrambe argillacea*), Barneby Reed-mustard (*Schoenocrambe barnebyi*), Shrubby Reed-mustard (*Schoenocrambe suffrutescens*) Recovery Plan. USFWS. Denver, Colorado.

References cited in this section are internal to the above-referenced Federal Register document. They are included in the Bibliography.

The shrubby reed-mustard (*Schoenocrambe suffrutescens*), like the species discussed in the previous two species accounts, occurs on fine-grained soils in desert shrub in the Colorado River drainage of eastern Utah. The shrubby reed-mustard is found within close proximity of the Clay reed-mustard, in the south-central Uintah Basin near the Green River in Uintah County, Utah. The species grows on clay soils with chips of white shale littered on the ground surface, derived from the Green River geologic formation. Species populations are commonly on level to moderately-sloping ground surfaces. Plants grow in mixed desert shrub and pinyon-juniper communities, at elevations ranging from approximately 5,100 to 6,700 feet. Prominent associated shrub and herbaceous species include: pygmy sagebrush, saltbush, mountain mahogany, cryptantha species, saline wildrye, Mormon tea, basin fleabane, ephedra buckwheat, spiny greasebush, hyaline herb, winged four o'clock, Colorado feverfew, shortspine horsebrush, table Townsend daisy, and Spanish bayonet. Many of these species are local endemics that are found only in the Uintah Basin.

The shrubby reed-mustard reproduces entirely by sexual means. Flowering occurs from April to May, and fruiting occurs from May to June. Possible pollinators include native bee species.

The shrubby reed-mustard was federally listed as endangered on October 6, 1987. Critical habitat has not been designated. Threats to the shrubby reed-mustard include oil and gas exploration, drilling, and production, oil-shale mining and processing, building stone removal, and OHV use. All known populations of this species are on federal lands leased for oil and gas energy reserves. The shrubby reed-mustard is also vulnerable to surface disturbance associated with energy developments within its habitats (USFWS 1990b). Trampling by livestock is also a potential threat.

Last Chance Townsendia
The primary reference for this section is:
USFWS. 1993b. Last Chance Townsendia (*Townsendia aprica*) Recovery Plan. USFWS. Denver, Colorado.

The Last Chance townsendia (*Townsendia aprica*) is a low-growing perennial, herbaceous plant that is known from a series of small populations in Emery, Sevier, and Wayne counties in central Utah, at elevations ranging from approximately 5,500 to 8,400 feet. Most populations occur in a band about 5 miles wide and 30 miles long, beginning near Interstate 70 at the western edge of the San Rafael Swell to near Fremont Junction, then south along the Emery-Sevier county line to the vicinity of Hartnet Draw. Populations of Last Chance townsendia generally occur with galleta and salt desert shrubs, in small barren openings of pinyon-juniper communities. Commonly associated plant species include galleta, blue grama, black sagebrush, shadscale, snakeweed, Indian ricegrass, and yellow rabbitbrush.

The surface geology in the area where the Last Chance townsendia occurs is highly mixed and contains a wide variety of soils with unusual soil chemistries. Most known populations of the species grow in soils derived from shale, that have a very fine silt texture and very high alkalinities, and that occur at the surface in small, isolated pockets. These pockets effectively form “islands” of suitable habitat in a “sea” of unsuitable geologic substrates with their resultant soil types.

The Last Chance townsendia reproduces by sexual means. Flowering occurs from April to May, and fruiting occurs in May and June. Self-pollination is virtually non-existent; instead, pollination is accomplished by several species of solitary bees. A few species of flies also visit the flowers. It appears that seed set is frequently limited by pollination. Lack of pollination may be caused by various factors, including low pollinator numbers, inclement weather affecting pollinator flight activity, and possibly other unidentified factors.

The Last Chance townsendia was federally listed as threatened on August 21, 1985. Critical habitat has not been designated for this species. Because the Last Chance townsendia is so restricted in its distribution, any event that could result in the loss of individuals or habitat within one or more populations is a potential threat to the species survival. Threats to the species come primarily from mineral and energy development, road building, and livestock trampling.

Maguire Daisy

The primary reference for this section is:
USFWS. 1996a. Reclassification of *Erigeron maguirei* (Maguire Daisy) from Endangered to Threatened. Federal Register 61 (119): 31054-31085.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The Maguire daisy (*Erigeron maguirei*) is endemic to sandstone canyons and mesas, and occurs in the San Rafael Swell in Emery County, Utah, and Capitol Reef in Wayne County, Utah. These two occurrences were once considered to be two taxonomically distinct varieties (*E. maguirei* var. *maguirei* and *E. maguirei* var. *harrisonii*, respectively). However, through DNA analysis, it has been documented that the two varieties are not genetically distinct, and that recognition at the varietal level is not genetically warranted (Van Buren 1993). Surveys during 1990 documented that about 3,000 individuals of the Maguire daisy occur at 12 sites in the San Rafael Swell and Capitol Reef (Kass 1990, Heil 1989). These 12 sites are reproductively isolated, forming separate populations (Heil 1994, Van Buren 1994).

The Maguire daisy is a perennial, herbaceous plant with decumbent to sprawling or erect stems. One to three flower heads are borne at the end of each stem. Small and isolated populations of this species have a high potential of becoming genetically homozygous, rendering them vulnerable to the loss of genetic viability (Van Buren 1994).

The Maguire daisy was originally listed as endangered by the USFWS, as *E. maguirei* var. *maguirei*. However, once recognized at the species level, the Maguire daisy was reclassified as threatened on June 19, 1996. Critical habitat has not been designated for this species. Even after reclassification, the Maguire daisy remains vulnerable to threats such as the loss of habitat and genetic viability. The small and isolated populations are susceptible to disturbances such as OHVs and trampling by humans and livestock. Mineral and energy exploration and development are also potential threats to the species. Individually, natural factors such as disease, flash floods, grazing by native species, erosion, and vegetative competition may not pose a definitive threat to this species. However, because of the daisy’s low population numbers, the cumulative effect of these threats could jeopardize its continued existence.

Maguire Primrose
The primary reference for this section is:
USFWS. 1990c. Maguire Primrose (*Primula maguirei*) Recovery Plan. USFWS. Denver, Colorado.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The Maguire primrose (*Primula maguirei*) is restricted to cool, moss-covered dolomite cliffs and boulders at the lower elevations (4,800 to 6,000 feet) of Logan Canyon in northern Utah. Plants appear to be dependent on the favorable temperature and moisture conditions of this microhabitat. Plants grow in cracks or crevices, or in a well-developed mat of moss. Associated plant species include pink alumroot, rock spiraea, tadpole buttercup, and narrowleaf wildparsley. The cliff face vegetation grows within a larger mosaic of mountain shrub, montane coniferous forest, and riparian vegetative communities, characteristic of the Wasatch Mountains (Cronquist et al. 1972).

The Maguire primrose is an herbaceous perennial plant that grows from 2 to 4 inches tall. Flowering typically occurs from mid-April to mid-May, and fruit development and seed dispersal occur from May through June. Both bees and flies have been observed visiting Maguire primrose flowers (Beedlow et al. 1980; Padgett 1986).

The Maguire primrose was federally listed as threatened on August 21, 1985. Critical habitat has not been designated. Threats to the species include road construction, water development, recreation, and collecting (Welsh 1979a, b; Beedlow et al. 1980; USFWS 1985b; Padgett 1986).

Clay Phacelia
Clay phacelia (*Phacelia argillacea*) is a narrow endemic to Spanish Fork Canyon, Utah County, Utah. The species is found in fine textured soil and fragmented shale derived from the Green River Formation. It grows on barren, precipitous hillsides in sparse pinyon-juniper and mountain brush communities, at elevations ranging from about 6,040 to 6,170 feet (Utah Conservation Data Center, no date). The dominant species occurring in habitats that support clay phacelia are Utah juniper and Utah serviceberry. The phacelia grows in openings between widely spaced woody plants, which are mostly 2 to 10 feet in height. Other common associates plant species include bluebunch wheatgrass, Indian ricegrass, shortstem buckwheat, smoothstem blazingstar, and gypsyflower (USFWS 1982a).

Clay phacelia is an herbaceous winter annual that grows up to about 14 inches tall. It germinates in the fall (September - October) if there is sufficient moisture, or as early in the spring (typically late April to early May) as the required moisture is available. Flowers are produced from June to mid-August, fruiting occurs from mid-June to September, and seed/fruit dispersal occurs from August through September. Flowers are pollinated by the wind and possibly bees or other insects. Seeds are dispersed by birds, gravity, and wind. Seed production varies depending on the climatic regimen of any given year. If there is sufficient moisture the number of plants will be greater and there will be a concomitantly greater number of seeds. Winter annuals tend to have seeds with long viability, so it is inferred that the clay phacelia also has long-lived seeds.

Clay phacelia was federally listed as endangered on September 28, 1978. Critical habitat has not been designated for the species. Construction activities have modified some of this species' habitat, and grazing by native ungulates and the presence of exotic plant species in its habitat are both potential threats (Utah Conservation Data Center, no date).

Heliotrope Milk-vetch
The primary reference for this section is:
USFWS. 1987a. Final Rule to Determine *Astragalus montii* (Heliotrope Milk-vetch) to be a Threatened Species, with Designation of Critical Habitat. Federal Register 52(215): 41652-42657.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

Heliotrope milk-vetch (*Astragalus montii*) is a narrow endemic of the Wasatch Plateau in central Utah. The species is restricted to Sanpete and Sevier counties, on outcrop barrens formed from a substrate of partially decomposed limestone. These limestone barrens are of a very limited extent, occurring at or near timberline (elevations between 10,000 and 11,000 feet) on top of the Wasatch Plateau. Heliotrope milk-vetch is found in subalpine communities of cushion plants and other low growing species, scattered within more extensive conifer, forb, and grass communities.

Heliotrope milk-vetch is a perennial herb of the pea family that grows to about 1/3 to 2 inches tall. Plants produce pink-purple, white-tipped flowers that bloom from June to August, and fruits are bladdery inflated pods.

Heliotrope milk-vetch was federally listed as threatened on November 6, 1987. At the time of listing, the USFWS designated approximately 65 acres of federal land in the Manti-LaSal National Forest, in Sanpete County, as critical habitat. Populations of heliotrope milk-vetch are in a general area of active oil and gas exploration. The associated oil and gas exploration and development are a threat to the species. Domestic livestock grazing also occurs within the species' habitat.

Dudley Bluffs Bladderpod
The primary reference for this section is:
USFWS. 1993. Dudley Bluffs Bladderpod and Dudley Bluffs Twinpod Recovery Plan. U.S. Fish and Wildlife Service. Denver, Colorado.

The Dudley Bluffs bladderpod (*Lesquerella congesta*) is endemic to the Piceance Basin in Rio Blanco County, Colorado. Within the Basin, the species occurs along drainages, on barren white oil shale outcrops that have been exposed through erosion from downcutting of streams. The species microenvironment is level surfaces at the points of ridges, and narrow, exposed outcrops of level white shale. Plants range from 6,140 to 6,644 feet in elevation.

The Dudley Bluffs bladderpod, which was discovered in 1982, is known from five major populations (in 1993). Most sites are on public land administered by the BLM, with the remainder on privately-owned land or Colorado Division of Wildlife land. Plants grow on tongues of White Green River shale within the overlying Uinta Formation, which is considered overburden to the thick underlying oil shale deposits. Plants are therefore vulnerable to impacts resulting from future development and extraction of these oil shale minerals and associated activities.

The Dudley Bluffs bladderpod was federally listed as threatened on February 6, 1990. Critical habitat has not been designated. Potential threats to the species include future underground mining of oil shale and the associated development. In addition, because the species is locally abundant on small areas of specialized habitat, the Dudley Bluffs bladderpod is particularly vulnerable to surface disturbances, despite its high densities.

Dudley Bluffs Twinpod
The primary reference for this section is:
USFWS. 1993c. Dudley Bluffs Bladderpod and Dudley Bluffs Twinpod Recovery Plan. Denver, Colorado.

The Dudley Bluffs twinpod (*Physaria obcordata*), like the Dudley Bluffs bladderpod discussed in the previous species account, is endemic to the Piceance Basin in Rio Blanco County, Colorado. Plants grow along drainages on barren white oil shale outcrops that have been exposed through erosion from downcutting of streams. The microenvironment for the Dudley Bluffs twinpod is steep sideslopes. Plants range from 5,960 to 7,440 feet in elevation.

The Dudley Bluffs twinpod, like the Dudley Bluffs bladderpod, was discovered in 1982, and is known from five major populations (in 1993). Most sites are on public land administered by the BLM, with the remainder on privately-owned land or Colorado Division of Wildlife land. Plants grow on tongues of White Green River shale within the overlying Uinta Formation, which is considered overburden to the thick underlying oil shale deposits. Plants are therefore vulnerable to impacts resulting from future development and extraction of these oil shale minerals and associated activities.

The Dudley Bluffs twinpod was federally listed as threatened on February 6, 1990. Critical habitat has not been designated. Potential threats to the species include future underground mining of oil shale and the associated development. In addition, because the species is locally abundant on small areas of specialized habitat, the Dudley Bluffs twinpod is particularly vulnerable to surface disturbances, despite its high densities.

Subtropical Desert Ecoregion Division

The Subtropical Steppe Ecoregion Division occurs in the southeastern portion of California, in southern Nevada, Arizona, and New Mexico, and in western Texas, and includes the Chihuahuan, Sonoran, and Mojave deserts. The dry, desert habitats that predominate in this ecoregion support communities in which xerophytic plants (e.g., small, hard-leaved or spiny shrubs; cacti; and hard grasses) are dominant. The inhospitable environs of shifting sand dunes and nearly sterile salt flats occur in this ecoregion. The important broad community types of the Subtropical Desert Ecoregion Division are desert grasslands and shrublands, and the higher elevation oak and pinyon-juniper woodlands.

Coachella Valley Milk-vetch
The primary reference for this section is:
USDI BLM. 2001a. Biological Evaluation on Effects of California Desert Conservation Area Plan as Amended and Proposed to be Amended by the NEMO and NECO Preferred Alternatives and With Other Interim Measures on Ten Threatened and Endangered Plants. California Desert District, BLM. Riverside, California.

The Coachella Valley milk-vetch (*Astragalus lentiginosus* var. *coachellae*) is restricted to the Coachella Valley in Riverside County, with the exception of six outlying occurrences in the Chuckwalla Valley north of Desert Center. Occurrences of this species are known from locations between the One Horse Spring area near Cabazon, to the sand dunes off Washington Avenue, north and west of Indio, in a longitudinal west to east range of approximately 33 miles. Extensive dune systems which once occurred at the base of the Santa Rosa Mountains from what is now the cities of Palm Springs, Cathedral City, Rancho Mirage, Palm Desert, Indian Wells, and La Quinta provided suitable habitat for the Coachella Valley milk-vetch. Today, only scattered remnants of these populations remain in sand dunes south of Interstate Highway 10.

The preferred habitat for the Coachella Valley milk-vetch has been described as dunes and sandy flats. It is also often associated with disturbance along the margins of sandy washes, and in sandy soils along roadsides, in areas formerly occupied by undisturbed sand dunes. Within dune habitat, this species is found in the coarser sands at the margins of dunes, rather than in the most active blowsand areas. Other populations have been located on sand substrates in creosote bush scrub, where the topography is rolling, stabilized dunes, or in pockets of sandy soil on the valley floor. The species may occur in localized pockets where sand has been deposited by wind or by an active wash, but would not be expected on rocky alluvial slopes.

Natural History. The Coachella Valley milk-vetch is described as a perennial or biennial, sometimes flowering as a winter annual. This plant flowers from February to May. In good years it may occur in large numbers, but most reports are of small populations of less than 20 plants. Specific data on population size and dynamics are not available for this species. However, great annual variation in population size has been observed, depending on rainfall.

Habitat Requirements. The Coachella Valley milk-vetch was federally listed as endangered in 1998. On December 14, 2204, the USFWS proposed designating approximately 3,583 acres in three units in Riverside and San Bernardino counties, California. The primary threat to the Coachella Valley milk-vetch is habitat destruction by urban development on private lands in the Coachella Valley. Other impacts to the species include the results of increased human activity, including OHV use, trampling, and the introduction of non-native plants. Development of wind energy parks has impacted this species, although the plants can persist within wind parks as long as disturbance to the sandy habitat is minimized.

Fragmentation of the extensive dune systems in the Coachella Valley has resulted in fragmentation of the existing populations and alteration of the natural processes that maintain the blowsand ecosystem. Development on the dunes has disrupted the flow and replenishment of sand to the remaining fragments. Though Coachella Valley milk-vetch does not appear to require, or even prefer, active blowsand dune habitats, the species does appear to be dependent on sand dune ecosystems.

Lane Mountain Milk-vetch

The primary reference for this section is:
USDI BLM. 2001b. Biological Evaluation on Effects of California Desert Conservation Area Plan as Amended and Proposed to be Amended by the NEMO and NECO Preferred Alternatives and With Other Interim Measures on Ten Threatened and Endangered Plants. California Desert District, BLM. Riverside, California.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The Lane Mountain milk-vetch (*Astragalus jaegerianus*) is a very rare and highly localized species with a range that occurs entirely within the western Mojave Desert. It is known to occur at elevations of approximately 3,150 to 3,850 feet. This species appears to be confined to granitic substrates in Mojave creosote bush scrub with a few widely scattered Joshua trees. It occurs on rocky, very low ridges, only a foot or two higher than the main bajada slope (i.e., a broad, gently inclined slope), and rocky low hills, 10 to 20 feet high, where bedrock is exposed at or probably near the surface (Lee and Ro Consulting Engineers 1986). Soils are shallow, rocky and coarse sandy decomposed granite (Lee and Ro Consulting Engineers 1986; Bagley 1989; Brandt et al. 1997). The scrub community at Lane Mountain milk-vetch sites is typically a diverse mix of shrub species including California buckwheat, Nevada Mormon tea, Cooper goldenbush, turpentine-broom, paper-bag bush, Mojave aster, hop-sage, Anderson box-thorn, creosote bush, and burro bush. Twenty-four perennial species were recorded in the vicinity of Lane Mountain milk-vetch at one population site on Fort Irwin (Lee and Ro Consulting Engineers 1986). A diversity of annual species may also occur in years with adequate moisture. Creosote bush and burro bush are dominant on the surrounding sandy bajada slopes, but are not dominant on the thin soils where Lane Mountain milk-vetch occurs (Bagley 1989; Brandt et al. 1997).

Only about 840 plants have ever been reported, including observations that may have reported the same plant more than once (USDI BLM 1997; Brandt et al. 1997; California Department of Fish and Game 1997). The entire known range of this species lies between Barstow and Goldstone, San Bernardino County, in an area no more than 13 miles in diameter. There are two population areas where this species is known to occur. The largest is to the north and northwest of the Paradise Range, northeast of Lane Mountain, where plants occur at scattered sites that cover a total of fewer than 875 acres. Most of the known sites occur within half a mile of a road. The second population area is located approximately 6 miles to the southwest, west of Lane Mountain on Coolgardie Mesa. Only two small sites, less than 10 acres, are known to occur here.

Lane Mountain milk-vetch is a spring-flowering perennial, with straggling, freely branched stems that arise from a buried root-crown (Barneby 1964a, b). The weak, sparsely leafy stems typically grow under and entangled within the canopy of low shrubs. Few plants have been observed in the open, not associated with a host or nurse shrub. It is believed that this host shrub provides some protection from herbivores, and may also benefit from the association because the milk-vetch is a nitrogen fixer.

Little has been reported on the growing season of Lane Mountain milk-vetch. However, it is known to grow in the spring and bloom in April and May. Presumably, like other desert perennials, it begins growth sometime in the late fall or winter, going dormant sometime in the late spring or summer when the soil moisture has been depleted in its rooting zone. The inflorescence bears from 5 to 15 dull yellowish-white or lavender-rose flowers. Nothing is known of the reproductive biology of Lane Mountain milk-vetch. Factors in pollination, seed production and dispersal, seed viability and longevity, seed germination, seedling establishment, and predation are all unknown.

The Lane Mountain milk-vetch was federally listed as endangered on October 6, 1998. Critical habitat has been designated for this species. However, because all three critical habitat units proposed were excluded from the final designation, zero acres in San Bernardino County, California have been identified as critical habitat. Because of its small population and small range, the Lane Mountain milk-vetch is particularly vulnerable to extinction as a result of random events (USFWS 1992a). It is potentially threatened by ongoing military activities at the Fort Irwin National Training Center and by proposed expansion of Fort Irwin onto adjacent public lands. The largest population occurs on Fort Irwin, in an area thus far not used for training. Except for the small population on Coolgardie Mesa, the remainder of the plants occur within one of the proposed alternative sites for Fort Irwin expansion (USDI BLM 1996a). The primary threat to the species is from OHV travel, particularly from heavy trucks and tracked vehicles. Sheep grazing, a minor threat noted by the USFWS (1992a), has been alleviated by closure of the grazing allotments within the range of the plant due to conflicts with the listed desert tortoise. Mineral claims on public land could also potentially pose a threat to this species.

Peirson's Milk-vetch

The primary reference for this section is:
USDI BLM. 2001b. Biological Evaluation on Effects of California Desert Conservation Area Plan as Amended and Proposed to be Amended by the NEMO and NECO Preferred Alternatives and with Other Interim Measures on Ten Threatened and Endangered Plants. California Desert District, BLM. Riverside, California.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

Peirson's milk-vetch (*Astragalus magdalenae* var. *Peirsonii*) is endemic to the Algodones Dunes, also known as the Imperial Sand Dunes, in the Colorado Desert of Imperial County, California. In addition, it is known to occur in the dunes of the Gran Desierto in northern Sonora, Mexico (USFWS 1998a). This species has been reported from the Borrego Valley in San Diego County (Barneby 1964b), but this location has never been confirmed. It is also believed that the species may occur in dunes in southwestern Arizona. Peirson's milk-vetch is distributed as one extensive population of scattered colonies throughout the length of the Algodones Dunes, an active dune system stretching more than 40 miles southeasterly from the Salton Sea to just across the U.S.-Mexican border into Baja California Norte, Mexico. Management of the Algodones Dunes is primarily by the BLM, with the exception of a few privately-owned parcels, and is contained within the boundaries of the Imperial Sand Dunes Recreation Area.

Although Peirson's milk-vetch colonies are scattered throughout the Algodones Dunes, suitable habitat for this species does not occur everywhere within the dunes system. The plant occurs only on wind-blown hollows and slopes primarily in the western two-thirds of the dunes, (WESTEC Services Inc. 1977; USDI BLM 2000c). The plant usually occurs on the leeward side of dunes where sand movement is less extreme. Peirson's milk-vetch is commonly found in association with other dune species, particularly dune buckwheat, sandpaper plant, and Wiggin's croton.

Peirson's milk-vetch is a short-lived perennial plant in the pea family. Seeds germinate after late summer or fall rains. The large size of Peirson's milk-vetch seeds allows for germination and growth from depths of several inches (Bowers 1996). After germination, seedling mortality is high as a result of burial and excavation due to shifting sand. Surviving seedlings grow rapidly and may flower as early as 2 months after germination (Barneby 1964b). The taproot is extraordinarily long, often extending more than 6 feet into the ground from a plant 1 foot in height. Typically, the plant flowers in winter and either dies or becomes dormant by late spring. It is not known how long plants may remain dormant if they do not receive adequate rainfall to resume growth the following season. Small bees have been seen visiting Peirson's milk-vetch and are most likely the pollinators of this species.

Peirson's Milk-vetch was federally listed as threatened on October 6, 1998. On August 4, 2004, the USFWS designated in 21,836 acres of Imperial County, California, as critical habitat. Suitable habitat for this species in the Algodones Dunes has been substantially reduced due to impacts from OHV use and associated camping. Monitoring studies conducted in 1977 and 1998 show that Peirson's milk-vetch has been eliminated from OHV staging and camping areas, but it still occurs in areas of low to moderate OHV use (WESTEC Services Inc. 1977; ECOS, Inc. 1990; USDI BLM 2000c).

The subspecies is still threatened by OHV use in the Algodones Dunes. The small stature of Peirson's milk-vetch provides little obstacle to riders and the brittle nature of its stem causes it to break rather than bend when hit by a vehicle (ECOS, Inc. 1990). The lack of lateral roots also reduces its ability to survive vehicle damage (Romspert and Burk 1979). Seedling establishment of this species occurs during the winter and spring, which are the most popular periods for OHV use in the dunes. The young seedlings are particularly vulnerable to crushing and dislodging by vehicles and may be destroyed by being run over by a vehicle. Indirect effects from OHV use such as sand compaction, disruption of hydrologic factors and changes in community composition may also be responsible for the decline of this species in areas used by OHVs (ECOS, Inc. 1990).

The North Algodones Wilderness was established in 1994, but it protects only 20% of Peirson's milk-vetch habitat. The most suitable habitat containing the highest plant numbers for this species occurs in the OHV open area, formerly Wilderness Study Area 362, in the central portion of the dunes. (WESTEC 1977, USDI BLM 2000c). Fortunately, OHV use in much of this area has been low due to its remoteness from staging and camping areas, and most of the habitat remains relatively intact (USDI BLM 2000c).

Triple-ribbed Milk-vetch

The primary reference for this section is:

USDI BLM. 2001b. Biological Evaluation on Effects of California Desert Conservation Area Plan as Amended and Proposed to be Amended by the NEMO and NECO Preferred Alternatives and With Other Interim Measures on Ten Threatened and Endangered Plants. California Desert District, BLM. Riverside, California.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

Triple-ribbed milk-vetch (*Astragalus tricarinatus*) is endemic to California and is restricted to the dry slopes and canyons around the head of the Coachella Valley (Barneby 1964a, Munz 1974). It is primarily known from the vicinity of Whitewater Canyon and from Dry Morongo Canyon along Highway 62, as well as from scattered occurrences farther east in the Little San Bernardino Mountains, including an anomalous, relatively high elevation, site at Keys Ranch in Joshua Tree National Park. The species has also been collected in the Martinez Canyon area in the Santa Rosa Mountains on the southwest side of the Coachella Valley. Thus, this plant may also occur in the rugged canyons of the San Jacinto and Santa Rosa mountains between Whitewater Canyon and Martinez Canyon, although it was not located during extensive surveys of the Santa Rosa Mountains.

Triple-ribbed milk-vetch is restricted to sandy or gravelly soils in arid canyons at the edge of the desert, but its habitat requirements are otherwise very poorly described. Plants occur at elevations between 1,300 and 4,000 feet, and are most commonly found along washes, on canyon bottoms and on the alluvial fans below, or as small populations or solitary individuals on decomposed granite slopes in canyons. All populations found to date appear marginal or transitory, and it appears that no large well-established permanent population has ever been found. The species appears to require open soil and is somewhat tolerant of, or may even require, soil disturbance, either natural or man made. It may in fact benefit by the open loose soils left by flooding or construction activities. However, given the small size of most populations and the instability of the habitats occupied, it is difficult to see how this species can maintain itself. It is possible that "permanent" populations may exist on the slopes above the washes, but have not been located yet. If the species is, in fact, largely restricted to canyon bottoms and wash margins, then it is extremely rare.

Triple-ribbed milk-vetch is a somewhat bushy herb, generally described as a perennial, but apparently more commonly behaving as an annual. At best, it is a short-lived perennial persisting for about 3 to 5 years. Mature plants are usually 12 to 20 inches tall and the stems are erect or ascending. Based on specimen records, the species flowers from February 12 through April 6, though the true range is likely to extend a few days beyond these dates. The inflorescence bears 10 to 15 widely spaced flowers. Fruits appear as early as mid-March and are present until at least early May. Pollinators, germination requirements, seed longevity, and most other aspects of the biology of this species are unknown. The color and form of the flowers suggest that this species may be bee pollinated, as many legumes are.

Triple-ribbed milk-vetch was listed as endangered by the USFWS on October 6, 1998 (USFWS 1998a). Critical habitat has not been designated. Known populations are few, small, and highly unstable. Since habitat modification within its range has not been extensive, it does not appear likely that human activity has been an important factor in its present scarcity. Current threats to this species do not appear serious, given the ruggedness of the area, but are not well documented. If the species is restricted to wash margins, then OHVs, which typically use such washes as access routes in rugged landscapes could be a potential threat. There is some disturbance due to pipeline construction or maintenance, and a potential threat of future mining of gravel in Whitewater Canyon.

Amargosa Niterwort
The primary reference for this section is:
USDI BLM. 2001. Biological Evaluation on Effects of California Desert Conservation Area Plan as Amended and Proposed to be Amended by the NEMO and NECO Preferred Alternatives and with Other Interim Measures on Ten Threatened and Endangered Plants. California Desert District, BLM. Riverside, California.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Amargosa niterwort (*Nitrophila mohavensis*) is confined to a few small depressions, or sinks, of the Carson Slough in Nevada and California (from Ash Meadows Wildlife Refuge in Nevada downstream to Franklin Playa, California) and from at least one locale on the eastern shore of the Amargosa River at Grimshaw Basin, California. This habitat is composed of highly saline and alkaline soils that are hydrated to varying degrees and are formed by seepage from freshwater springs that lie many miles to the north and east in Ash Meadows, Nevada (Beatley 1977). The Amargosa niterwort grows on open, highly alkaline mudflats and low sand deposits in sinks, around alkali sink vegetation. All populations are known from wet alkaline flats lacking appreciable standing water and which support very little vegetation, with extensive salt crust development. The species occurs in the open and is generally not found with, or under, any type of cover. It is found at elevations between approximately 1,970 and 2,460 feet. Associated plants include spiny saltbush, Parry's saltbush, iva, Tecopa bird's-beak, short-pedicelled cleomella, pickleweed, and saltgrass. Natural and unaltered hydrology within Lower Carson Slough appears critical for the survival of the Amargosa niterwort (California Department of Fish and Game 1990).

The Amargosa niterwort is a small, erect perennial from an extensive, heavy, underground rootstock. The largest population of the species is thought to consist of several thousand individuals (Reveal 1978a), many of which are interconnected via underground rootstocks. Plants can over-winter as underground rootstocks, with new plants starting their growth in March. Flowering is from late April to October. Each flower produces one solitary, shiny black seed. Viability, longevity, dormancy and germination requirements of seeds are unknown (Reveal 1978a).

On June 19, 1985, the Amargosa niterwort was federally listed as an endangered species, with designated critical habitat. The restricted range of this species makes it susceptible to natural catastrophic events such as flooding and drought, as well as the genetic and demographic consequences of small populations. A majority of all suitable habitat in California for this species is on public lands. Potential threats to the species include local groundwater depletion; streambed alteration; highway maintenance; mining, including exploratory drilling and claim marker placement; OHV travel; and trampling by wild horses. An additional threat is the potential introduction and spread of the exotic plant saltcedar. Saltcedar has not been observed near Franklin Playa to date, though it does occur downstream on the Amargosa River in the vicinity of Grimshaw Basin.

Ash Meadows Milk-vetch
The primary reference for this section is:
Nevada Natural Heritage Program. 2001a. Rare Fact Sheet for *Astragalus phoenix* Barneby (1970), Ash Meadows Milk-vetch. Available at http://www.state.nv.us/nvnhp/atlas/.

Other references used are cited in the text and included in the Bibliography.

The Ash Meadows milk-vetch (*Astragalus phoenix*) is endemic to the Ash Meadows area of Nye County, Nevada, cienega (a desert wetland) ecosystem maintained by several dozen springs and seeps. The species occurs on dry, hard, seasonally moist, white, barren flats, washes, and knolls of calcareous alkaline soils. Associated species include saltgrass, shadscale saltbush, Ash Meadows blazingstar, alkali goldenbush, and Ash Meadows sunray. Its habitat can be generally described as warm desert scrub. This species has only been known to grow in areas of mineral encrusted soil; no growth of this species has been observed in areas that have been disturbed (Reveal 1978b, Monzingo and Williams 1980). However, the species is also found most commonly in open places without any vegetation cover (Reveal 1978b). The maximum range of the species is approximately 7 miles, on lands administered by the USFWS and the BLM, as well as on privately-owned lands. There are 10 occurrences of this species throughout its range.

The Ash meadows milk-vetch is a low, mat-forming perennial herb that forms mats of up to 1.6 feet in diameter. Germination probably occurs in the spring or fall but depends on late fall or early winter rains (Reveal 1978b). Leafing occurs from March to early April. Flowering occurs from late April through May and requires sufficient rains in the winter or early spring. Fruiting occurs from May to June with seed/fruit dispersal occurring from May to July (Reveal 1978b, Monzingo and Williams 1980). Sufficient rain is probably necessary for seedling establishment (Reveal 1978b). Plants are small and long-lived, and seed production for this species is relatively low.

The Ash Meadows milk-vetch was federally listed as threatened on May 20, 1985. Critical habitat has been designated in the Ash Meadows area of Nye County, Nevada, in portions of sections 14, 21, 22, and 26, Township 17 South, Range 50 East, sections 1, 12, 13, and 24, Township 18 South, Range 50 East, and sections 7, 18, and 19, Township 18 South, Range 51 East. Major threats to the species include development, rabbit grazing, horses, and dust from disturbed soil.

Spring-loving Centaury
The primary reference for this section is:
Nevada Natural Heritage Program. 2001b. Rare Fact Sheet for *Centaurium namophilum* Reveal, Broome and Beatley (1973), Spring-Loving Centaury. Available at http://www.state.nv.us/nvnhp/atlas/.

Other references used are cited in the text and included in the Bibliography.

The spring-loving centaury (*Centaurium namophilum*) is another endemic to the Ash Meadows area of Nye County, Nevada. Historically, the species was found in adjacent California as well. The species occurs along the Amargosa River drainage on open, moist to wet, alkali-crusted soils of seeps, springs, outflow drainages, meadows and hummocks. It is found at elevations of 2,100 to 2,350 feet. The species is aquatic or wetland-dependent, and commonly occurs with the following species: saltgrass, goldenweed, Baltic rush, Yerba mansa, western niterwort, saltbush, Tecopa bird's-beak, ash, mesquite, saltcedar, baccharis, and cattail. There are 14 occurrences of this species, over a range of 9 miles, on lands administered by the USFWS and the BLM, and on privately-owned land.

The spring-loving centaury is an annual that flowers from July to September (Reveal et al. 1973). Fruiting occurs in October. Little else about reproduction and life history of this species is known.

The spring-loving centaury was federally listed as threatened on May 20, 1985. Critical habitat has been designated in the Ash Meadows area of Nye County, Nevada, in portions of sections 21, 23, 28, 34, and 35, Township 17 South, Range 50 East; sections 1, 2, 3, 7, 23, and 24, Township 18 South, Range 50 East; and sections 7, 18, 19, 20, 29, and 30, Township 18 South, Range 51 East. The species is threatened by regional groundwater pumping, competition from invasive weeds, impacts from past agricultural conversion, and water diversion.

Ash Meadows Ivesia
The primary reference for this section is:
Nevada Natural Heritage Program. 2001c. Rare Plant Fact Sheet for *Ivesia kingii* S. Watson var. *eremica* (Coville [1892]) Ertter, Ash Meadows Ivesia. Available at http://www.state.nv.us/nvnhp/atlas/.

Other references used are cited in the text and included in the Bibliography.

The Ash Meadows ivesia (*Ivesia kingii* var. *eremica*) is endemic to the Ash Meadows area of Nevada. The species occurs in open areas, on moist to saturated, heavy to chalky alkaline soils. Plants grow in meadows on flats, drainages, and bluffs near springs and seeps. They are commonly associated with highly alkaline, clay lowlands or depressions where soil moistures remains high from perched groundwater maintained by springs and seeps (USFWS 1985c). The taxon is typically found in saltgrass meadow, shadscale, and ash-mesquite, associated with the following species: shadscale saltbush, saltgrass, baltic rush, mesquite, Mojave thistle, spring-loving centaury, velvet ash, Yerba mansa, and iva. The Ash Meadows ivesia is a matted perennial herb/shrub that bears white flowers from August to October. The Ash Meadows ivesia is aquatic or wetland-dependent, and occurs at elevations ranging from 2,200 to 2,300 feet. There are nine occurrences of the species that cover a combined total area of approximately 9 acres, on land administered by the USFWS and the BLM, and on privately-owned land.

The Ash Meadows ivesia was federally listed as threatened on May 20, 1985. Critical habitat has been designated in the Ash Meadows area of Nye County, Nevada, in portions of sections 21 and 35, Township 17 South, Range 50 East; and sections 1, 2, 3, 12, 23, and 24, Township 18 South, Range 50 East. This species is threatened by development, trampling and grazing, and the associated large-scale drawdown of water resources.

Ash Meadows Gumplant
The primary reference for this section is:
USDI BLM. 2001b. Biological Evaluation on Effects of California Desert Conservation Area Plan as Amended and Proposed to be Amended by the NEMO and NECO Preferred Alternatives and with Other Interim Measures on Ten Threatened and Endangered Plants. California Desert District, BLM. Riverside, California.

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The Ash Meadows gumplant (*Grindelia fraxino-pratensis*) is an erect, biennial or more often perennial, herb of the sunflower (Asteraceae) family. It is known only from moist, meadow habitats along Carson Slough in Nevada and California; from Ash Meadows Wildlife Refuge in Nevada downstream to Franklin Playa, California; and has also been reported along the Amargosa River from near Tecopa, California. The populations of the Ash Meadows gumplant follow drainage patterns from spring sources in the Ash Meadows region into Carson Slough, the major drainage system of Ash Meadows. The current population status of the Ash Meadows gumplant is unknown, and population trends are difficult to determine because long-term data are unavailable.

The Ash Meadows gumplant primarily occurs in saltgrass meadows along streams and surrounding pools in the vicinity of ash-screwbean-mesquite woodlands and desert shadscale scrub vegetation. It occasionally occurs sparsely on open alkali clay soils in drier shadscale habitats or in the unique clay barrens where groundwater is at or near the surface, and where other Ash Meadow endemics are supported. The species is quite robust in marshy areas along some dirt roads where runoff accumulates and saturates soils throughout a longer portion of the year. The Carson Slough populations occur in full sunlight and in the lowest topographic areas associated with water (Cochrane 1981).

The Ash Meadows gumplant appears to colonize recently disturbed areas, almost appearing weed-like, along roadsides adjacent to meadows. The quick colonization may be due to the removal of the usual associated plant competitors (Beatley and Reveal 1971, Cochrane 1981).

The dominant plant species occurring with the gumplant is saltgrass. Common associates within the saltgrass meadow type community include spring-loving centaury, seep willow, Yerba mansa, western niterwort, loosestrife, and iva. In wooded areas and on drier sites, common associates include velvet ash, screwbean mesquite, shadscale, alkali sacaton, alkali goldenbush, rabbitbush, seepweed, and other saltbush species.

The Ash Meadows gumplant flowers from June through October (Beatley 1977). Seed dispersal could occur by means of wind/water transportation and possibly by mammals or birds. The pollinators for this species are unknown at this time (Cochrane 1981).

The Ash Meadows gumplant was federally listed as threatened with designated critical habitat on May 20, 1985 (USFWS 1985c). It is likely that before human-caused habitat modifications such as grazing, farming, and water diversions occurred, the distribution of this species was more or less continuous (Cochrane 1981). Existing threats to the Ash Meadows gumplant include the reduction of spring outflow caused by adjacent land development and/or water diversion; the destruction and/or modification of the limited habitat available to this species from camping, staging area, road maintenance and/or mining activities; and the degradation of habitat resulting from wild horse grazing/trampling and OHV use impacts.

The potential also exists for the exotic plant saltcedar to establish and spread on gumplant habitat. Saltcedar replaces native plants, alters the composition and structure of native plant communities, and generally “dries up” wetland and meadow habitats. If this exotic plant were to become well established in the vicinity of gumplant populations, the surface water necessary for the species’ survival could be affected.

Ash Meadows Blazing Star

The primary reference for this section is:
Nevada Natural Heritage Program. 2001d. Rare Fact Sheet for *Mentzelia leucophylla* Brandegee (1899), Ash Meadows Blazingstar. Available at http://www.state.nv.us/nvnhp/atlas/.

Other references used are cited in the text and included in the Bibliography.

The Ash Meadows blazingstar (*Mentzelia leucophylla*) is endemic to the Ash Meadows area of Nye County, Nevada. It occurs in open areas, on dry, hard, salt-crusted alkaline clay or sandy-clay soils. Plants grow on low bluffs, swales, flats, and drainages, in shadscale vegetation that surrounds spring and seep areas. This habitat can be generally categorized as warm desert scrub. Associated species include shadscale saltbush, alkali goldenbush, Ash Meadows sunray, and Ash Meadows milk-vetch. The Ash Meadows blazingstar is found at elevations of between 2,240 and 2,300 feet. There are eight occurrences of this species over a range of approximately 6 miles, on land administered by the USFWS and the BLM, as well as on privately-owned land.

The Ash Meadows blazingstar is a biennial herb with bright yellow flowers that bloom from late May into September. Flowers open only for brief periods in the late afternoon. Observations made in early spring indicate that individuals of this species do not overwinter, and that there was no new growth from previous years (typical of a biennial; Reveal 1978c). Sufficient rain is probably necessary to allow flowering. Since populations of mature plants vary greatly from year to year, it is likely that the total number of seeds produced varies also. The dispersal of this species’ seeds is restricted to the sides of gullies and on raised knolls of the flats and lower foothills in the area of the existing populations. Like the Ash Meadows milk-vetch, the Ash Meadows blazingstar is apparently sensitive to disturbance or habitat alteration, as it is not found on any disturbed sites either as seedlings or as established plants.

The Ash Meadows blazingstar was federally listed as threatened on May 20, 1985. Critical habitat has been designated in the Ash Meadows area of Nye County, Nevada, in portions of sections 15, 21, 22, 23, 28, 35, and 36, Township 17 South, Range 50 East, and sections 1, 2, 11, and 12, Township 18 South, Range 50 East. This species is threatened by agricultural development.

Ash Meadows Sunray

The primary reference for this section is:

Nevada Natural Heritage Program. 2001e. Rare Fact Sheet for *Enceliopsis nudicaulis* (A. Gray) A Nelson var. *corrugata* Cronquist (1972), Ash Meadows Sunray. Available at http://www.state.nv.us/nvnhp/atlas/.

The Ash Meadows sunray (*Enceliopsis nudicaulis* var. *corrugata*) is also endemic to the Ash Meadows area, occurring in both Nevada and adjacent California. The species occurs on dry to somewhat moist, hard, strongly alkaline silty to clay soils, in open areas, often on or near low calcareous outcrops. Plants are found in spring and seep areas, at elevations from 2,200 to 2,360 feet, in creosote-bursage and shadscale zones. Common associated plant species include shadscale saltbush, alkali goldenbush, saltgrass, broom snakeweed, ratany, basin yellow cryptantha, desert bearpoppy, Ash Meadows blazingstar, and Ash Meadows milk-vetch. There are 11 occurrences of this species, which together total an area of 27 acres.

The Ash Meadows sunray is a perennial shrub that flowers in April and May. Flowers are borne singly on leafless flower stalks. Little is known about the reproductive biology and life history of this species.

The Ash Meadows sunray was federally listed as threatened on May 20, 1985. Critical habitat has been designated in the Ash Meadows area of Nye County, Nevada, in portions of sections 15, 21, 22, 34, and 35, Township 17 South, Range 50 East; sections 1, 2, 12, and 13, Township 18 South, Range 50 East; and sections 7 and 18, Township 18 South, Range 51 East. This taxon is threatened by groundwater pumping and other agricultural development activities, road construction, and OHV traffic.

Nichol's Turk's Head Cactus

The primary reference for this section is:

Arizona Game and Fish Department. 1999. *Echinocactus horizonthalonius* var. *nicholii*. Unpublished abstract compiled and edited by the Heritage Data Management System, Arizona Game and Fish Department. Phoenix, Arizona.

Nichol's Turk's head cactus (*Echinocactus horizonthalonius* var. *nicholii*) is a succulent perennial that occurs in desert scrub in the Waterman Mountains of Pima and Pinal counties, Arizona. Plants occur in lime siltstone talus or bedrock, at elevations of 2,050 to 3,600 feet. The habitat is typically open, characterized by few trees and scattered low shrubs. Common associates include yellow paloverde, triangle burr ragweed, white ratany, goldenhills, cactus apple, saguaro, ocotillo, buckhorn cholla, and woody crinklemat. Extant populations occur on land administered by the BLM, the Bureau of Indian Affairs, and the state, as well as on privately-owned land.

Nichol's Turk's head cactus is a very slow growing plant, requiring 10 to 32 years to reach 2 inches in height. Germination occurs in mid-summer, and vegetative growth takes place primarily in March through May. The majority of flowering occurs in late April to mid July, often in response to the first warm-weather rain, but plants can flower as late as November. Flowers remain open from approximately 10 a.m. to 5 p.m., for 1 or 2 days. Common pollinators include bees and butterflies. An average of 200 seeds are produced per plant per year, and seeds are dispersed by birds, mammals, and rainwater.

Nichol's Turk's head cactus was federally listed as endangered on October 26, 1979. Critical habitat has not been designated. Direct human interference remains the most important ongoing threat to this subspecies. Blading a landing strip, mining, and road construction have all destroyed a sizeable number of plants. In addition, there is persistent illegal collection of plants.

Kearney's Blue-star

The primary reference for this section is:

Arizona Game and Fish Department. 1997a. *Amsonia kearneyana*. Unpublished abstract compiled and edited by the Heritage Data Management System, Arizona Game and Fish Department. Phoenix, Arizona.

Kearney's blue-star (*Amsonia kearneyana*) is an herbaceous perennial that is limited to the South and Sysamore canyons of the Baboquivari Mountains in Pima County, Arizona. In addition, there is an introduced population in Brown Canyon on the east side of the mountains. This species generally occurs in canyon bottoms on sandy alluvium, in partial shade under deciduous riparian trees, at elevations between 3,680 and 6,400 feet. The habitat is not strictly riparian, however, as plants may also be found on hillsides. Plant communities that support this species include the Mexican blue oak association, Sonoran desertscrub, and semidesert grassland.

Kearney's blue-star flowers from March through April, and fruits ripen from June through July. Hawk moths may pollinate plants at night. Observed predation of seeds by boring insects has made this species largely sterile. Extant populations of this species occur on land administered by the Bureau of Indian Affairs and the BLM, and on privately-owned land. The reintroduced population is at a site in the Buenos Aries National Wildlife Refuge.

Kearney's blue-star was federally listed as endangered on January 19, 1989. Critical habitat has not been designated. This species is threatened by its extreme rarity, physical damage from livestock, and other disturbances that can cause mortality to plants.

Pima Pineapple Cactus

The primary reference for this section is:

Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation: Wildland Urban Interface Fuel Treatment. Forest Service, Southwestern Region. Albuquerque, New Mexico.

The Pima pineapple cactus (*Coryphantha scheeri* var. *robustispina*) is sparsely distributed in Sonoran desert scrub and semi-desert grasslands of Arizona. Its range extends east from the Baboquivari Mountains to the northeastern foothills of the Santa Rita Mountains, and from near Tucson south into Mexico. The Pima pineapple cactus occurs most commonly in open areas on flat ridegetops or areas with less than 10% slopes, and at elevations from 2,400 to 4,200 feet. Preferred sites have silty to gravelly deep alluvial soils. The cactus does not typically occur in mountainous areas, but is found instead on valley floors and bajadas. Habitats for the Pima pineapple cactus can be broken into two major divisions: ridges in what is now or once was grassland, and alluvial fans in Sonoran Desert scrub. On a smaller scale, the plant occupies habitats that are relatively flat and sparsely vegetated. In hilly landscapes, the Pima pineapple cactus is found on flat hilltops, but is missing from slopes or drainages separating the hilltops. It is not found in riparian areas.

On average, the Pima pineapple cactus is a semi-circular plant, with single or numerous stems, and spine clusters. Flowers, which are yellow to nearly white, appear in early July with the onset of summer rains. With adequate moisture, flowering can continue until August. The fruits are green and succulent, and they may be taken quickly by animals for broad dispersal of the seeds, or they may wither and dry among the spine clusters. Under conditions of sufficient moisture, these withered fruits disintegrate, scattering seeds into the immediate vicinity of the dispersing cactus.

The Pima pineapple cactus was federally listed as endangered on September 23, 1993. Critical habitat has not been designated for this species. Threats to the taxon include collection, OHV use, development related to mining and housing, the introduction and spread of non-native grasses for livestock forage, and use by increasing numbers of javelinas.

Huachuca Water-umbel

The primary reference for this section is:

Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation: Wildland Urban Interface Fuel Treatment. Forest Service, Southwestern Region, Albuquerque, New Mexico.

Huachuca water-umbel (*Lilaeopsis schaffneriana recurva*) is an herbaceous, semi-aquatic plant that occurs in healthy riverine systems, cienegas (desert wetlands), and springs in Arizona and Mexico. In watersheds that generally do not experience scouring floods, it occurs in microsites where competition between plant species is low. At these sites, the plant occurs on wet soils, interspersed with other plants at low densities, along the periphery of the wetted channel, or in small openings in the understory. In stream and river habitats, it can occur in backwaters, side channels, and nearby springs. After a flood, the water-umbel is able to rapidly expand its population by occupying the disturbed habitat, persisting until it is no longer able to compete with other plant species. The Huachuca water-umbel occurs at 19 sites in four major watersheds: San Pedro River, Santa Cruz River, Rio Yaqui, and Rio Sonora. All sites are between 3,500 and 6,500 feet in elevation.

The Huachuca water-umbel is a perennial plant with slender, erect leaves that grow from creeping rhizomes. Flowers are borne on an umbel in groups of 3 to 10. The plant reproduces sexually through flowering, and asexually from rhizomes, with the latter the primary reproductive strategy. The taxon may also vegetatively disperse when clumps of plants are dislodged from one location and then re-root in a different site along the aquatic system. The density of plants and the size of populations fluctuate in response to both flood cycles and site characteristics. The number of individuals in any given population may be difficult to detect because the creeping rhizomes tend to intermesh, and because reproduction is predominantly asexual.

The Huachuca water-umbel was federally listed as endangered on January 6, 1997. Critical habitat was designated on July 12, 1999 on the Upper San Pedro River, in Garden Canyon of Fort Huachuca and other areas of the Huachuca Mountains, in the San Rafael Valley, and on Sonoita Creek. This taxon is threatened primarily by wetland degradation and loss, which reduces the amount of available habitat. Human activities such as groundwater withdrawals, surface water diversions, impoundments, channelization, improper livestock grazing, chaining, agriculture, mining, sand and gravel operations, road building, the introduction of non-native species, urbanization, timber harvest, and recreation all contribute to the loss and degradation of riparian and cienega habitat. In addition, limited numbers of populations and the small size of populations make the Huachuca water-umbel vulnerable to extinction through chance events, such as drought, disease, or lightning-induced wildfires.

Canelo Hills Ladies'-tresses

The primary reference for this section is:

Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation, Wildland Urban Interface Fuel Treatment. Forest Service, Southwestern Region. Albuquerque, New Mexico.

Other references used are cited in the text and included in the Bibliography.

Canelo Hills ladies'-tresses (*Spiranthes delitescens*) is an orchid that is known from five sites in cienega and streamside habitats within the San Pedro River watershed in Santa Cruz and Cochise counties, Arizona. These sites occur in areas where scouring floods are unlikely. Soils supporting the populations are finely grained, highly organic, and seasonally or perennially saturated. Springs are the primary water source, but a creek near one locality contributes near-surface groundwater. The five sites for this orchid occupy less than 200 acres of habitat near the U.S./Mexico border. Four sites occur on privately-owned land. The dominant vegetation associated with Canelo Hills ladies'-tresses includes grasses, sedges, rushes, spike rushes, cattails, and horsetails (USFWS 1997a). The surrounding vegetation is semidesert grassland or oak savannah.

Canelo Hills ladies'-tresses is a slender plant with linear, grass-like basal leaves, that produces a flower stalk of up to 40 spirally-arranged white flowers. Mature plants seldom flower in consecutive years, and in some years have no visible aboveground structures. Although it is presumed that fire once played a role in the life history of this orchid, a full understanding of both fire and other disturbances is lacking. Since little cienega habitat remains, and with so few known individuals, fire events that once may have been beneficial to the species could now depopulate an entire site.

Canelo Hills ladies'-tresses was federally listed as endangered on January 6, 1997, but critical habitat has not been designated. The primary threats to this species are activities that result in wetland habitat degradation, such as groundwater overdrafts, surface water diversions, impoundments, channelization, improper livestock grazing, agriculture, mining, invasive non-native species, and recreation. This orchid may also be threatened by collection. In addition, the limited distribution and low numbers of individuals leave it vulnerable to extinction from chance events.

Cochise Pincushion Cactus

The primary reference for this section is:
Arizona Game and Fish Department. 2001a. *Coryphantha robbinsorum*. Unpublished abstract compiled and edited by the Heritage Data Management System, Arizona Game and Fish Department. Phoenix, Arizona.

The Cochise pincushion cactus (*Coryphantha robbinsorum*) is a succulent perennial that is endemic to desert scrub communities in southern Arizona and northern Sonora, Mexico. Arizona populations are limited to southeastern and southwestern Cochise County on both state and privately-owned land. Plants are found on the rolling gray limestone slopes of hills in the transition zone between Chihuahuan Desert scrub and semidesert grassland, at elevations of 4,200 to 4,650 feet. Plants are rooted in bedrock cracks or thin soil, where there is an abrupt vegetation change. They prefer areas with good drainage, and full sun to light shade. Associated species include alkali muhly, fairyduster, Palmer's century plant, pinkflower hedgehog cactus, dissodia, spinystar, cactus apple, and ocotillo.

Plants tend to be solitary or scattered in discrete sub-populations, rather than randomly spread out. Flowering occurs in late March and into April, with flowers opening at around mid-day and pollinated by bees. Fruiting occurs from late June through August. There may be short-distance dispersal year-round, with seeds coming off the mother plant and germinating below it. In addition, the red, fleshy fruits attract birds, which then disseminate the seeds over long distances.

The Cochise pincushion cactus was federally listed as threatened On January 9, 1986. Critical habitat has not been designated. The species is at risk because it is a local endemic with specific substrate requirements. Plants are subject to illegal collecting, and occasionally suffer direct damage by livestock uprooting plants.

Arizona Cliff-rose

The primary reference for this section is:
Arizona Game and Fish Department. 2001b. *Purshia subintegra*. Unpublished abstract compiled and edited by the Heritage Data Management System, Arizona Game and Fish Department. Phoenix, Arizona.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The Arizona cliff-rose (*Purshia subintegra*) is a low, woody shrub that is endemic to limestone soils in central Arizona. The current range of the species includes Maricopa County (near Horseshoe Lake), Yavapai County (near Cottonwood), Mohave County (near Burro Creek), and Graham County (near Bylas). The species occurs where the winters are mild, summers are hot, and the rainfall (9 to 34 inches) is evenly distributed between summer and winter rainfall periods. The landscape is dissected by ephemeral drainages and is sparsely vegetated. Plants typically grow on rolling, rocky, limestone hills and slopes, within Sonoran Desert scrub, at elevations of 2,120 to 4,000 feet. The species requires white tertiary limestone lakebed deposits high in lithium, nitratesm and magnesium. The Arizona cliff-rose tends to be the dominant or codominant shrub on sites where it occurs.

There are four disjunct populations of Arizona cliff-rose (listed above), which exist along an area of central Arizona that is 200 miles wide. The Cottonwood population includes the greatest number of individual plants, including seedlings. Extant populations are found on land under a number of different ownerships: private, BLM, Bureau of Indian Affairs, Forest Service, State of Arizona, and possibly the Bureau of Reclamation.

The Arizona cliff-rose was federally listed as endangered on May 29, 1984. Critical habitat has not been designated for this species. This species is very vulnerable because of its limited number of populations, habitat specificity, and a number of threats. Browsing by livestock and burrows, poor reproduction, mineral exploration and development, construction and maintenance of roads and utility corridors, recreation, OHV use, urbanization, pesticides, and urbanization are all threats to the species (USFWS 1995a). The relative importance of each of these threats varies from population to population.

Arizona Hedgehog Cactus

The primary reference for this section is:

Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation: Wildland Urban Interface Fuel Treatment. Forest Service, Southwestern Region. Albuquerque, New Mexico.

The Arizona hedgehog cactus (*Echinocereus triglochidiatus* var. *arizonicus*) occurs in interior chaparral, madrean evergreen woodland, and desert grassland plant communities in Arizona, primarily at elevations ranging from 3,400 to 5,300 feet. Habitat consists of exposed bedrock or boulders in rugged, steep-walled canyons and boulder pile ridges and slopes. Typically, the cactus is scattered on open, rocky exposures, rooting in shallow soils and narrow crevices among the boulders. The plant may also grow beneath an understory of shrubs, but moderate to high shrub densities and associated deeper soils tend to preclude its establishment.

Because there are numerous red-flowered hedgehog cacti, which are variously grouped and separated as different species and varieties by different authors, there is some confusion surrounding both the morphology and range of the Arizona hedgehog cactus. As hybridization readily occurs among plants, and isolated populations rapidly evolve slightly different morphological characteristics, defining this variety becomes even more complicated. Investigations conducted between 1992 and 1994 stated that the Arizona hedgehog cactus occupies a range of 30 square miles, and that a very small distribution of the taxon was readily accessible to the general public. However, the BLM has identified over 300,00 acres of potential habitat for the Arizona hedgehog cactus on public lands in east-central and southeastern Arizona. The main distribution is thought to occur in the vicinity of Globe/Miami, Arizona, though there are likely thousands of plants occurring in satellite populations disjunct from the main distribution.

The Arizona hedgehog cactus is a robust, succulent perennial, with dark green stems that occur singly or in clusters of a few to 10 stems, though some plants may have over 100 stems. Flowers erupt along the sides of the stems, and are a brilliant scarlet to deep red in color. Flowering occurs from late April to mid-May. Likely pollinators include insects—primarily bees—and perhaps hummingbirds. Fruits are present from May through June, with several fruits occurring per plant and 100 seeds produced per fruit. The amount of variation in annual seed production, and in seed viability and longevity are unknown. Seed dispersal is likely by birds and mammals. Seeds do not appear to have special germination requirements apart from protection from extended direct sunlight and extreme temperatures (i.e., above 110 °F), and germination can occur in mid-summer. Natural insect predators include borers and leaf-foot bugs that attack the stems. Rodents may also gnaw on stems.

The Arizona hedgehog cactus was federally listed as endangered on October 25, 1979. Critical habitat has not been designated. Plant collection, mining, and livestock grazing have all been identified as threats to this plant, although it is likely that at present these threats have far less impact than originally believed. A substantial portion of the range has been designated as wilderness and receives additional protections. From a biological standpoint, it is possible that this cactus is not as in danger of extinction as was previously thought.

Dwarf Bearclaw-poppy

The dwarf bearclaw-poppy (*Arctomecon humilis*) is a narrow endemic to Washington County, Utah, where it is found on gypsiferous clay soils derived from the Moenkopi formation. Plants occur on low hills, bluffs, and outcrops of this formation, or at the bases of ridges and buttes (Welsh and Thorne 1979, USFWS 1982b). These isolated populations are surrounded by creosote bush-dominated vegetation, but in general the species is associated with the mixed warm desert shrub community (Welsh 1978b, USFWS 1979, Welsh and Thorne 1979, USFWS 1982b). Dominant plant species include creosote bush and longspine horsebrush (Welsh 1978b, USFWS 1979). Other associated species include Fremont's dalea, burrobrush, Torrey's jointfir, saltbush, crispleaf buckwheat, desert pepperweed, Parry's sandpaper plant, beautiful phacelia, and Palmer's phacelia (USFWS 1982b).

The dwarf bearclaw-poppy grows only in clay to sandy or rocky clay soils containing a high amount of gypsum (Welsh 1978b, USFWS 1979, Welsh and Thorne 1979). This soil type is highly alkaline and has shrink-swell properties, which allow the soil to become a sticky mud during spring and fall, then extremely hard during summer (USFWS 1979, 1982b). The elevation at which the poppy grows varies between 2,000 and 3,500 feet (Welsh 1978b, Welsh and Thorne 1979, USFWS 1982b), and the plant requires a southern exposure, with open sun (Welsh 1978b).

The dwarf bearclaw-poppy is an evergreen, herbaceous, perennial species. Plants flower in mid-April through May, and fruit in May and June. Reproduction is sexual. The soil seedbank is apparently critical for the persistence of populations of this species, since mortality rates are high and germination events are widely spaced. Because transplanting and cultivating the poppy is usually unsuccessful, recovery potential for this species is poor unless its habitat is preserved and protected (USFWS 1979, 1982b).

The dwarf bearclaw-poppy was federally listed as endangered on November 6, 1979. Critical habitat has not been designated. Causes of mortality to the species include OHV use, mineral exploration, and land utilization for urban and industrial development (Welsh 1978b, Anderson 1982a). The dwarf bearclaw-poppy has probably always been restricted to its small range, possibly due to specific soil and elevation requirements. It is estimated that 10 to 20% of the species' historic habitat has been destroyed by the development of the cities of St. George and Bloomington, and by the construction of the Interstate 15 freeway (Anderson 1982a, USFWS 1982b). Presently, the dwarf bearclaw-poppy is threatened by housing, recreational and industrial development throughout its existing range (USFWS 1979, Welsh and Thorne 1979). Expanding land use around St. George and excessive motorcycle and other OHVs use is damaging much of the remaining habitat of the poppy (USFWS 1979, Welsh 1979b, Anderson 1982a, USFWS 1982b). Although not as great a problem as OHVs, the collection of the poppy for ornamental gardening has occurred. Mineral exploration, strip mining of gypsum deposits, the Warner Valley Power Project, and privatization of public land are all possible future threats.

Holmgren Milk-vetch

The primary reference for this section is:

USFWS. 2001a. Determination of Endangered Status for *Astragalus holmgrenorium* (Holmgren Milk-vetch) and *Astragalus ampullarioides* (Shivwitz milk-vetch). Federal Register 66(189): 49560-49567.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Holmgren milk-vetch (*Astragalus holmgrenorium*) is a narrowly-distributed endemic of the Mojave Desert, restricted to the immediate vicinity of the city of St. George, Utah. The Holmgren milk-vetch grows on shallow, sparsely vegetated soils derived primarily from limestone. The species is a principal member of a warm-desert shrub community dominated by desert goldenhead, white burrobush, and Anderson wolfberry. Also associated with the Holmgren milk-vetch are several perennial and annual forbs and grasses, most importantly the non-native foxtail brome, storksbill, and African mustard (Armstrong and Harper 1991, Van Buren 1992, Stubben 1997, Harper and Van Buren 1998, Van Buren and Harper 2000a). Only three populations of the Holmgren milk-vetch are known, one with about 9,000 to 10,000 plants, one with about 1,000 plants, and one with about 30 plants (Stubben 1997; Van Buren 1998; Bolander 2000).

The Holmgren milk-vetch is a stemless herbaceous perennial plant that produces leaves and flowers in the spring, both of which die back to its roots after the flowering season. Fruits are pods that eventually dries out and opens, releasing seeds. Plants are pollinated by native solitary ground-dwelling bees (Tepidendo 2000, Bolander 2000).

Fragmented, isolated populations of the species restrict pollinator exchange between occupied population sites. This situation may cause genetic isolation, which potentially lead to inbreeding and local extirpation of isolated populations.

The Holmgren milk-vetch was federally listed as endangered on September 28, 2001. Critical habitat was deemed prudent by the USFWS, but has not yet been designated. Substantial portions of the species' habitat are subject to disturbance from urban development, OHVs, grazing, displacement by exotic weeds, and mineral development. In addition, the introduction of frequent fire into the Mojave Desert ecosystem with the spread of non-natives such as downy brome and foxtail brome has been identified as a threat to these species.

Shivwitz Milk-vetch
The primary reference for this section is:
USFWS. 2001a. Determination of Endangered Status for *Astragalus holmgrenorium* (Holmgren Milk-vetch) and *Astragalus ampullarioides* (Shivwitz milk-vetch). Federal Register 66(189): 49560-49567.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Shivwitz milk-vetch (*Astragalus ampullarioides*), like the Holmgren milk-vetch discussed in the previous species account, is narrowly-distributed endemic of the Mojave Desert, restricted to the immediate vicinity of the city of St. George, Utah. The Shivwitz milk-vetch grows only on purple clay soils derived from the Petrified Forest member of the Chinle geological formation. Native plant species associated with the Shivwitz milk-vetch include beautiful bluedicks, birdsfoot trefoil, snakeweed, mariposa lily, and several other Mojave Desert plants. However, the most important plant species associated with the Shivwitz milk-vetch are the non-natives foxtail brome, downy brome, and African mustard (Armstrong and Harper 1991; Van Buren 1992, 1998; Harper and Van Buren 1998; Van Buren and Harper 2000b). The Shivwitz milk-vetch is known from five separate sites in Washington County, Utah, which are distributed over a narrow band of the exposed Chinle formation over a distance of about 45 miles. These 5 populations contain a total of approximately 1,000 plants (Van Buren 1998, 2000).

The Shivwitz milk-vetch is a perennial herbaceous plant, with flowering stems that may attain a height of 40 inches if not grazed. Each plant produces about 45 small flowers on a single stalk in the spring. Seeds are produced in small pods, and the plant dies back to its root crown after the flowering season. Plants are pollinated by native solitary ground-dwelling bees (Tepidendo 2000, Bolander 2000). Fragmented, isolated populations of the plants restrict pollinator exchange between occupied population sites. This situation may cause genetic isolation, which potentially lead to inbreeding and local extirpation of isolated populations.

The Shivwitz milk-vetch was federally listed as endangered on September 28, 2001. Critical habitat was deemed prudent by the USFWS, but has not yet been designated. Substantial portions of the species' habitat are subject to disturbance from urban development, OHVs, grazing, displacement by exotic weeds, and mineral development. In addition, the introduction of frequent fire into the Mojave Desert ecosystem with the spread of non-natives such as downy brome and foxtail brome has been identified as a threat to the Shivwitz milk-vetch.

Gypsum Wild Buckwheat
The gypsum wild buckwheat (*Eriogonum gypsophilum*) occurs in Chihuahuan Desert scrub in Eddy County, New Mexico, at three known locations: north of Carlsbad at Seven River Hills; south of Black River Village; and in the drainages of Ben Slaughter Draw and Hay Hollow (New Mexico Rare Plant Technical Council 1999). The species occurs on almost pure gypsum that is sparsely vegetated with other gypsophilous plants such as hairy crinklemat, gypsum blazingstar, and southwestern ringstem. Plants occur at elevations of 3,280 to 3,600 feet, on the eroded hillsides and tops of the gypsum hills, as well as on the gypsum colluvial fans at the base of the hills.

The gypsum wild buckwheat occurs on slopes of 0 to 45 degrees, and does not have an apparent exposure preference (Spellenberg 1977, Wagner and Sabo 1977, USFWS 1984a). There appears to be some correlation between surface disturbance and plant density, with larger numbers of the plant present where the tough surface crust of the gypsum is broken. Hence, plants are often most abundant adjacent to erosion channels on the hillsides, and roadways along the base.

The gypsum wild buckwheat is a small herbaceous perennial that arises from a persistent woody root crown (Wooten and Standley 1913, Reveal 1977). Budding occurs in early May, flowering occurs from mid-May to early July, and fruiting occurs in late July to early August (Wooten and Standley 1913; Reveal 1977; Martin and Hutchins 1980; Fletcher et al. 1984). Seed dispersal probably also occurs in August.

The gypsum wild buckwheat was federally listed as threatened on January 19, 1981. Critical habitat has been designated in Eddy County, NM, in portions of Township 20 South Range 25 East, Sec. 19 and Township 20 South, Range 24 East, Sec. 24. The gypsum wild buckwheat is an extremely rare plant that is restricted to one locality of approximately 500 acres in size. With such a limited distribution this species is sensitive to both limited scale projects as well as those of regional impact. The present threats to this species include OHV use, oil and gas exploration, and excessive grazing of cattle. While large numbers of cattle likely pose a threat to this species through trampling and browsing, light cattle use may be beneficial to the plant by breaking the gypsum crust (USFWS 1984a). Aside from the potential inundation of a small segment of the population, the presence of a large body of water adjacent to the critical habitat zone could cause a variety of secondary effects.

Lee Pincushion Cactus

The Lee pincushion cactus (*Coryphantha sneedii* var. *leei*) occurs in semi-desert grassland in the high Chihuahuan Desert of Carlsbad Canyon National Park in the Guadalupe Mountains, Eddy County, New Mexico. The species is restricted to limestone substrates on terraces and rimrock, with the majority of the plants growing in cracks in the rocks on north facing slopes between 5,000 and 5,900 feet in elevation (Martin and Hutchins 1980: Fletcher et al. 1984; New Mexico Department of Natural Resources 1985). Plants occur in an agave-juniper association, which is dominated by large almost arborescent shrubs. Associated species include muhly, prairie clover, Pinchot’s juniper, common sotol, yucca, Texas sacahuista, oak, cactus apple, and Apache plume (Heil and Brack 1985). Plants are usually sparsely distributed among the scrubby vegetation, and rarely occur under cover (Martin and Hutchins 1980; Fletcher et al. 1984; Heil and Brack 1985; New Mexico Department of Natural Resources 1985).

The Lee pincushion cactus is long-lived succulent perennial species. Reproduction is sexual; although plants can be propagated vegetatively for cutting, they have no natural mechanism for doing so. Lee pincushion cactus plants bud in early to mid-April, flowers are produced in early May, and fruit is developed in late summer. The seeds are thought to be dispersed in October, with germination taking place in late May to early June (Heil and Brack 1985). Pollinating agents are believed to be bees, and seed dispersal agents are thought to be rodents and ants.

The Lee pincushion cactus was federally listed as threatened on October 25, 1979. Critical habitat has not been designated. The subspecies is threatened by illegal collecting by cactus enthusiasts (Heil and Brack 1985, New Mexico Department of Natural Resources 1985, USFWS 1985d). Plants are relatively tough, not being affected by many of the fungi and insect predators that other cacti are susceptible to. The recovery potential of the Lee pincushion cactus appears to be quite high.

Sneed Pincushion Cactus

The Sneed pincushion cactus (*Coryphantha sneedii* var. *sneedii*) is restricted to limestone substrates on terraces, ridgetops, hillsides, and ledges in the high Chihuahuan Desert of the Franklin, Guadalupe, and Organ mountains of Texas and New Mexico. Plants occur primarily in cracks in the limestone substrate or in shallow pockets of loamy soil on hillsides and ridgetops between 3,900 and 7,700 feet in elevation (USFWS 1985d). The subspecies typically occurs in semi-desert grasslands or woodlands, in an agave-juniper association. In the Guadalupe Mountains it extends upward in elevation to the lower pinyon-juniper woodland. Like the Lee pincushion cactus (discussed in the previous species account), it usually occurs in sparsely vegetated areas with shrubby species, but is rarely under cover. Associated plant species include lechuguilla, sideoats grama, whitecolumn foxtail cactus, common sotol, longleaf jointfir, Apache plume, Pinchot’s juniper, Texas sacahuista, cactus apple, oak, and pinyon pine (USFWS 1985d, New Mexico Department of Natural Resources 1985).

The Sneed pincushion cactus is a long-lived succulent perennial species. Reproduction is sexual; although plants can be propagated vegetatively for cutting, they have no natural mechanism for doing so. Sneed cactus plants likely germinate from late May to early June, but do not begin blooming until after 3 to 4 years of age. The plants bud in March and April, flower in mid- to late April, and fruit from August to November. Pollinating agents are believed to be bees, and seed dispersal agents are thought to be rodents and ants.

The Sneed pincushion cactus was federally listed as endangered on November 7, 1979. Critical habitat has not been designated. The taxon is threatened by illegal collecting by cactus enthusiasts (Heil and Brack 1985, New Mexico Department of Natural Resources 1985, USFWS 1985d). Plants are relatively tough, not being affected by many of the fungi and insect predators that other cacti are susceptible to. The recovery potential of the Sneed pincushion cactus appears to be quite high.

Pecos Sunflower
The primary reference for this section is:
USFWS. 1999b. Determination of Threatened Status for the Plant *Helianthus paradoxus* (Pecos Sunflower). Federal Register 64(202):56581-56590.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS New Mexico Ecological Services Field Office. Albuquerque, New Mexico.

The Pecos sunflower (*Helianthus paradoxus*) is dependent on desert wetlands for its survival. The species grows in permanently saturated soils, and though it is found most commonly in desert wetlands associated with springs, it may also occur along stream and lake margins. Plants commonly associated with Pecos sunflower include Transpecos sealavender, limewater brookweed, clasping yellowtops, Olney bulrush, common reed, saltgrass, alkali sacaton, alkali muhly, Mexican rush, Pursh seepweed, and saltcedar (Poole 1992, Sivinski 1995). All of these species are good indicators of saline soils.

The Pecos sunflower is an annual member of the sunflower family that flowers from September to November. It is very similar in appearance to the common sunflower, with large, bright yellow flowers. It is known from 22 sites in Cibola, Valencia, Guadalupe, and Chaves counties, New Mexico, and from three sites in Pecos and Reeves counties, Texas. Various federal, state, tribal, municipal, and private interests own and administer the Pecos sunflower sites. Federal agencies include the BLM and National Park Service.

The Pecos sunflower was federally listed as endangered on October 20, 1999. Critical habitat has not been designated for this species. The loss or alteration of wetland habitats is the main threat to the Pecos sunflower. The lowering of water tables through aquifer withdrawals and diversion of water from wetlands for irrigation, livestock, or other uses; wetland filling; and invasion of saltcedar and other non-native species continue to destroy or degrade desert wetlands. Mowing of some municipal properties and highway ROW regularly destroys some plants. Livestock will eat Pecos sunflowers, particularly if other green forage is scarce. There has been some unregulated commercial sale of Pecos sunflowers in the past, and some plant collection for breeding programs to improve commercial sunflowers. Pecos sunflower will naturally hybridize with common sunflower, and it is possible that backcrosses from hybrids could affect the genetic integrity of small Pecos sunflower populations.

Subtropical Steppe Ecoregion Division

The Subtropical Steppe Ecoregion Division is located in a large portion of northern Arizona, New Mexico, and Texas, and a small portion of southern Utah and Colorado. Composed of plateaus and high plains, and occurring at a higher elevation than the warm deserts to the south, this ecoregion is typified by a semiarid steppe climate. Grassland vegetation predominates, with locally developed shrubs and woodlands. Common plant community types found in the Subtropical Steppe Ecoregion Division include pinyon-juniper woodlands, perennial grasslands, chaparral and other shrublands, with ponderosa pine and other evergreen forests occurring in the mountainous regions.

Arizona Agave

The primary reference for this section is:
Arizona Fish and Game Department. 1997b. *Agave arizonica*. Unpublished abstract compiled and edited by the Heritage Data Management System, Arizona Game and Fish Department. Phoenix, Arizona.

The Arizona agave (*Agave arizonica*) is a succulent perennial that inhabits open chaparral, desert grassland, and transition zones between grasslands and pinyon-juniper. Its range is limited entirely to central Arizona, where it is found in the New River Mountains in Yavapai and Maricopa counties and southeast of Payson and in the Sierra Ancha Mountains in Gila County. Plants typically grow on mesas and slopes, from 3,000 to 6,000 feet in elevation, on mixed gravelly loam soils and granitic outcrops. Commonly associated plant species include goldenflower century plant, Toumey agave, pricklypear, shrub live oak, juniper, mountain mahogany, and mesquite. Extant populations of the Arizona agave occur on land in the Tonto National Forest administered by the Forest Service, and on privately-owned land.

Plants mature in 22 to 35 years, flower once, and then die. Flowering occurs throughout the month of June, and major pollinators of the species include hummingbirds and insects. The species exhibits very poor reproduction, and cloning through the production of suckers has been observed, although sparingly. Seed production in the wild is also low.

The Arizona agave was federally listed as endangered on May 18, 1984, without critical habitat. Threats to the species include its limited distribution and low numbers, herbivory of flowerstalks by cattle and deer, and damage by snout-weevil beetles. There is also some risk from collection of plants, although they are difficult to find.

Brady Pincushion Cactus

The primary reference for this section is:
Arizona Game and Fish Department. 2001c. *Pediocactus bradyi*. Unpublished abstract compiled and edited by the Heritage Data Management System, Arizona Game and Fish Department. Phoenix, Arizona.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The Brady pincushion cactus (*Pediocactus bradyi*) is endemic to Marble Canyon, Coconino County, Arizona. Plants occur in Great Basin desert scrub habitats, on gently sloping benches and terraces with very specific soil characteristics. Although the species does not exhibit a preference for a specific soil type, it is always found on Kaibab limestone chips overlaying soil derived from shale, mudstone, and siltstone. Plants are typically found in open, exposed habitats with sparse vegetation characterized by scattered low shrubs (saltbush, snakeweed, jointfir), grasses (grama, dropseed), and annuals (globemallow, buckwheat).

Scattered populations occur along both sides of the rim of Marble Canyon and tributary canyons for a distance of about 25 miles, from below Lee’s Ferry to the vicinity of Bedrock Canyon on the west side, to Tanner Wash on the east side. The densest populations occur along the rims of Soap Creek and Rider Canyon, and nearby portions of the rim of Marble Canyon. Total potential habitat has been estimated to be 17,000 acres, though only 10 to 20% appears to be occupied. Known populations of plants occur on land administered by the BLM, the Bureau of Indian Affairs, and the National Park Service (Glen Canyon National Recreation Area).

The Brady pincushion cactus is a globose succulent perennial that flowers from late March to April. On sunny days, flowers open mid-morning and close in the evening, and may open for four of five successive days (Spence 1992). There is some evidence that this species is an obligate out-crosser, and that flowers are pollinated by insects, primarily native bees. Fruits mature in late May to early June. A mature fruit may contain 15 seeds, and the total number produced by a single plant over its life is relatively small. The roots of the Brady pincushion cactus are associated with beneficial microorganisms called mycorrhizal fungi. Under cool temperatures and wet conditions, the species is highly susceptible to root rot.

The Brady pincushion cactus was federally listed as endangered on October 26, 1979. Critical habitat has not been designated. The species is highly desired for its ornamental value in the cactus and succulent trade, and cultivation difficulties make wild populations a target for collectors. In addition, highway and road maintenance has affected at least one population, and trampling associated with livestock grazing has also had local impacts on this species. Additional threats include OHV usage and impacts from dispersed recreation. Many Brady pincushion cacti are eaten by rodents, especially under drought conditions (Hughes 1991).

Peeble's Navajo Cactus

The Peebles Navajo cactus (*Pediocactus peeblesianus* var. *peeblesianus*) grows on specialized soils of the Chinle Formation in a very small area of Northern Arizona. Populations of this taxon occur in the Plains and Great Basin grassland, near the ecotone with the Great Basin scrub (Brown et al. 1980). The plants in these communities are generally low in stature, and vegetative cover is sparse, characterized by low shrubs, grasses, and seasonal annuals (USFWS 1984b). Occasional junipers, associated with Peebles Navajo cacti are about 10 feet tall, and the canopy is open. This species occurs between about 5,150 and 5,300 feet, which is the elevation of the geologic formation around Holbrook, Arizona. The plants grow in exposed, sunny situations in gravelly alluvium on 0- to 30-degree slopes, and sloping to flat hill tops. The soils are shallow to deep, well drained to excessively well drained and formed in mixed alluvium. The Peebles Navajo cactus occurs in the mixed rangeland land use/land cover associations, specifically in the desert grasslands forest/rangeland associations. Dominant plants in these associations are snakeweed, shadscale, four-winged saltbush, rabbitbrush, sagebrush (Bigelow and big), Mormon tea, Cutler's jointfir, and galleta. Cactus associates are beehive cactus, whipple devil claw, and several prickly pear species.

The Peebles Navajo cactus is a succulent perennial that germinates in early April. Flowering occurs from mid-April to early May, and fruiting occurs approximately 1 month later, in May. Seed/fruit dispersal occurs within days of the fruit opening. Seeds do not germinate immediately after they are shed (June) because conditions are too hot and dry. Some will germinate the following spring, but optimum germination occurs after 2 to 3 years. Germination depends upon proper moisture at the right time, and all phenological dates are dependent on environmental conditions (Phillips et al. 1979). Seed dispersal is by wind, rainwater, and ants, and tends to produce relatively scattered colonies with fairly high density (Heil et al. 1981). Disturbance of the habitat by overgrazing or OHVs causes erosion and compaction of soil, and influences the success of seed dispersal to suitable habitats (Phillips et al. 1979). Limiting factors for Peebles Navajo cactus include its specialized soil needs, cold winters, moist, cool springs, and drying out periods (Phillips et al. 1979; Heil et al. 1981; Benson 1982; USFWS 1984b).

The Peebles Navajo cactus was federally listed as endangered on October 26, 1979. Critical habitat has not been designated. The most immediate threat to this species is quarrying operations, which are stripping much of the habitat for gravel used in road construction and for commercial purposes. The gravel and sand deposits of this soil unit are used extensively and contribute to most of the sand and gravel used in the Holbrook area (Soil Conservation Service 1982). The Peebles Navajo cactus is also in demand by collectors of rare cacti (Fletcher 1979a; Newland 1979a, b). Cattle trample plants on BLM and State of Arizona lands as well as on private grazing lands, especially during wet seasons when the ground is muddy and the plants are emergent (Phillips et al. 1979, USFWS 1984b). In addition, OHVs cause damage to the plants and their habitat through crushing of plants, erosion, and soil compaction. The potential use of this habitat for homesites is a real threat, since 70% of the potential habitat is in private ownership. Holbrook, Arizona, is expanding rapidly into the surrounding countryside, and the nearby hills are considered prime land for future development (Soil Conservation Service 1982).

Welsh's Milkweed

The primary reference for this section is:
USFWS. 1992b. Welsh's Milkweed (*Asclepias welshii*) Recovery Plan. USFWS. Denver, Colorado.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

Welsh's milkweed (*Asclepias welshii*) occurs on active aeolian sand dunes in Kane County, Utah, and Coconino County, Arizona. Populations occur on "islands" of suitable habitat that are surrounded by vegetated, stabilized sands, sandstone slickrock, or various exposed shales and other fine grained rock types or their developed soils. The plant community in which Welsh's milkweed occurs is dominated by sand mule's-ear, with prominent groves of ponderosa pine and clumps of gambel. Other plant species commonly associated with Welsh's milkweed include blowout grass, sand dropseed, giant dropseed, Indian ricegrass, giant dunegrass, sand hill muhly, sand-spurge, silvery sophora, dune scurfpea, Kanab yucca, rubber rabbit-brush, and winged wild-buckwheat. The vegetation surrounding the sand dune habitat is dominated by pinyon-juniper (Utah juniper) woodlands with big sagebrush parks. Plants are found at elevations ranging from approximately 5,000 to 6,500 feet, with the largest population of this species occurring on the Coral Pink Sand Dunes, located about 7.5 miles west of Kanab, Utah.

Welsh's milkweed is a tall, herbaceous plant in the milkweed family. Reproduction in this species is both sexual and asexual. Flowering occurs from May to June, and fruit and seed development and dispersal occur from July to September. Self-pollination is impossible in this species, and the highly-evolved floral structures appear to be pollinated by certain bees, wasps, butterflies, and moths. Welsh's milkweed has a deep-seated clustered root and stem system, a dense tomentum, and very large seeds, all of which are adaptations that allow this species to survive on the unusual sand dune habitat to which it is restricted. However, because Welsh's milkweed has a very low rate of fruit development (Wyatt 1976), vegetative reproduction by sprouting from rhizomes is also important.

Welsh's milkweed was federally listed as threatened on October 28, 1987. Critical habitat has been designated for this species, and includes the entire Coral Pink Sand Dunes west of Kanab, Utah, as well as the area of the Sand Hills (Section 8 in Township 42 South, Range 6 West) about 10 miles north of Kanab, Utah. Because of its very limited specific habitat requirements and its small population size, Welsh's milkweed is vulnerable to any event that could cause the local extirpation of one or more of its isolated populations. Realized and potential threats to this species stem primarily from recreational OHV use. Mineral and energy development, road building, and livestock grazing are minor threats.

Jones Cycladenia

The primary reference for this section is:
Utah Conservation Data Center. No Date. Fact Sheet for Jones' Cycladenia. State of Utah Natural Resources, Division of Wildlife Resources. Available at http://utahdc.usu.edu.

The Jones cycladenia (*Cycladenia humilis* var *jonesii*) is restricted to the canyonlands of the Colorado Plateau in Emery County, Garfield County, Grand County, and Kane County, Utah, and in adjacent Coconino and Mohave counties, Arizona. Plants grow in salty clay and gypsum soils that are derived from the Summerville, Cutler, and Chinle formations. These soils are shallow, fine textured, and intermixed with rock fragments. The Jones cycladenia can be found in wild buckwheat-Mormon tea, mixed desert shrub, and scattered pinyon-juniper communities, at elevations ranging from approximately 4,000 to 6,800 feet.

The Jones cycladenia is a rhizomatous herb with round, somewhat succulent leaves, and small flowers that bloom from mid-April to early June. This plant has very low sexual reproductive success, resulting from low rates of pollinator visitation and frequent abortion of fruit. However, the individual plants spread by underground rhizomes and can form clones up to 35 feet across.

Jones' cycladenia was federally listed as threatened on May 5, 1986. Critical habitat has not been designated. Off-highway vehicle activity and the presence of mining claims and oil and gas leases on or immediately adjacent to known sites are the biggest threats to this species. In addition, the relatively small number of populations make the species especially vulnerable to natural and human-caused disturbances. The arid climate and harsh soils of the ecosystem in which the Jones cycladenia is found make it fragile and slow to recover from surface disturbance.

Siler Pincushion Cactus

The primary reference for this section is:
Utah Conservation Data Center. No Date. Fact Sheet for Siler Pincushion Cactus. State of Utah Natural Resources, Division of Wildlife Resources. Available at http://utahdc.usu.edu.

The Siler pincushion cactus (*Pediocactus sileri*) ranges from near Fredonia, Coconino County, Arizona, westward to near St. George, Washington County, Utah. Its distributional center is in Mohave County, Arizona. The species is ecologically restricted to gypsiferous and calcareous sandy or clay soils derived from the various members of the Moenkopi Formation or the nearly identical Kaibab Formation. Plants grow on rolling hills, often with a badlands appearance, in warm desert shrub, sagebrush-grass, and, at its upper limits, pinyon-juniper communities. This species occurs at elevations ranging from approximately 2,640 to 5,400 feet. In most cases, individual plants are widely separated. Flowers bloom during March and April.

The Siler pincushion cactus was federally listed as endangered in 1979. Since that time, many more plants were discovered, and the species was reclassified as threatened on December 27, 1993. This species and its habitat are vulnerable to disturbance from OHV use, trampling by livestock, and possibly mining. In addition, illegal collection has adversely affected some populations.

Navajo Sedge

The Navajo sedge (*Carex specuicola*) occurs in the canyons of Kane County and San Juan County, Utah, and in Apache, Coconino, and Navajo counties, Arizona. The species is restricted to hanging garden habitats within the Great Basin conifer woodland of the Colorado Plateau (Brown and Lowe 1980). Plants grow in moist sandy to silty soils of seep-spring hanging gardens (USFWS 1987b; Phillips et al. 1981). These hanging gardens are found in pinyon-juniper woodlands on south-facing Navajo Sandstone Formation cliffs, at slopes ranging from 80 to 90%. Plants occur at elevations between approximately 5,700 and 6,000 feet. Hanging gardens are produced by water percolating through the porous sandstone, contacting an impervious stratum along which it flows laterally, forming a drip or spring-line along the cliff face. It is on or under this drip line or spring that hanging gardens are developed (Smith 1977).

Plant communities in hanging gardens differ in composition and in kinds of species, not only along an apparent north-south climatic gradient and along an elevational gradient, but also from one garden to another on the same cliff face (Welsh and Toft 1981; Brotherson et al. 1978). Plant species commonly associated with the Navajo sedge include monkey flower, helleborine, sand bluestem, and common reed (USFWS 1987b).

The Navajo sedge is a perennial herb that flowers in June and July. Plants are pollinated by wind. Seed dispersal occurs in late July.

The Navajo sedge was federally listed as threatened on May 8, 1985. Critical habitat has been designated in three 40-by-5-meter (131-by-16-foot) rectangular areas of moist, sandy to silty soils at shady seep-springs within the Navajo Sandstone Formation, Navajo Indian Reservation, Coconino County, Arizona. Most species of sedge are palatable to livestock, and it is suspected that domestic livestock (horses, sheep, goats, and cows) as well as wildlife graze the plants. The two major threats to the species are grazing and a lowering of the water table from water development for livestock. Water is vital to the survival of the species; thus, any change in the water table level will have an effect on the populations (USFWS 1987b). Other potential threats to the species include use of OHVs and illegal collection.

Kodachrome Bladderpod

The primary references for this section are:

Utah Conservation Data Center. No Date. Fact Sheet for Kodachrome Bladderpod. State of Utah Natural Resources, Division of Wildlife Resources and USFWS. 1993d. Final Rule to Determine a Utah Plant, of *Lesquerella tumulosa* (Kodachrome Bladderpod), as an Endangered Species. Federal Register 58(192): 52027-52031. Available at: http://utahdc.usu.edu

References cited in this section are internal to the above-referenced Federal Register document. They are included in the Bibliography.

The Kodachrome bladderpod (*Lesquerella tumulosa*) is a narrow endemic to Kane County, Utah. It grows on sparsely vegetated white shale knolls, in thin, poorly developed soils that are developed from the Winsor member of the Carmel geologic formation (Welsh and Reveal 1977, Welsh 1978c, M.C. Franklin 1990). Plants grow in scattered pinyon-juniper communities south of Kodachrome Basin, at elevations ranging from approximately 5,600 to 6,050 feet. Plant species commonly associated with the Kodachrome bladderpod include pinyon pine, Utah juniper, bitterbrush, yellow cryptantha, Indian ricegrass, pallid milkweed, hyaline herb, and morning-lily. There are two known occurrences of this plant, with a combined area of approximately 45 acres. In 1989, the population was estimated at nearly 20,000 plants.

A member of the mustard family, this species is a perennial herb that forms densely matted and depressed mounds. It has a many-branched woody base, and produces yellow flowers that bloom in May and early June.

The Kodachrome bladderpod was federally listed as endangered on October 6, 1993. Critical habitat has not been designated. The small population size of the Kodachrome bladderpod and its restricted habitat make the species vulnerable to human-caused and natural environmental disturbances. It occurs at locations that are subject to OHV use and domestic livestock grazing. In addition, the shale soils on which this plant grows are being actively quarried, and its habitat is threatened by mineral exploration and mining claim assessment work.

Winkler Cactus

The primary reference for this section is:

USFWS. 1998b. Final Rule To Determine the Plant *Pediocactus winkleri* (Winkler Cactus) to be a Threatened Species. Federal Register 63 (161): 44587-44595.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Salt Lake City Field Office. Salt Lake City, Utah.

The Winkler cactus (*Pediocactus winkleri*) is endemic to lower elevations of the Colorado Plateau in south-central Utah. Plants typically grow on the tops and sides of rocky hills or benches in saltbush-dominated desert shrub communities (Heil 1984). The species grows in alkaline silty loam or clay loam soils derived primarily from the Dakota formation, the Brushy Basin member of the Morrison formation, and the Emery sandstone member of the Mancos formation (Heil 1984, Neese 1987, USFWS 1997b).

The Winkler cactus is a small globose cactus with stems 1 to 2.5 inches tall and up to 2 inches in diameter, and clusters of small radial spines. Its flowers are urn-shaped, and its fruits are barrel-shaped, opening along vertical slits and expelling seeds. Four populations of the Winkler cactus are known. These populations total about 20,000 plants that grow on widely separated parcels of habitat between 2.4 acres and 48 acres in size. Three of the four populations form a narrow arc extending from near Notom in central Wayne County to the vicinity of Last Chance Creek in southwestern Emery County, Utah. The fourth is a disjunct population occurring near Ferron, Utah, in western Emery County. About two thirds of the plants occur on lands administered by the BLM east and north of the Capitol Reef National Park boundary. The remainder of the plants are found within the park.

The Winkler cactus was federally listed as threatened on August 20, 1998. Critical habitat has not been designated. This species is threatened by collection and by habitat disturbances caused by mining, recreation, and livestock.

Mesa Verde Cactus

The Mesa Verde cactus (*Sclerocactus mesae-verdae*) is a long-lived perennial species that occurs on sparsely vegetated, low rolling clay hills in San Juan County, New Mexico, and Montezuma County, Colorado (New Mexico Rare Plant Technical Council 1999). Plants require a substrate of highly eroded clay derived from shales and mudstone of marine origin, and typically occur in habitats characterized by little or no ground cover (USFWS 1984c). They occur at elevations between 5,250 and 6,600 feet, and are usually found on the tops and the benches of the slopes of rolling clay hills. The species is found in the Colorado Plateau, in the floristic province defined as the Navajoan Desert (Smith 1970). The associated plant community is predominated by the following species: mat saltbush, Nuttall’s saltbush, fragrant white sand verbena, Navajo evening-primrose, plains pricklypear, gallenta, scarlet globemallow, patch phacelia, longbeak streptanthella, yellow spiderflower, coral gilia, and sand dropseed.

The Mesa Verde cactus occurs in diffuse population complexes composed of widely scattered loci of individuals and clustered plants. It is not uncommon to walk a quarter of a mile between individual plants. The usual situation encountered is a grouping of 3 to 50 plants scattered over several acres forming a population center and connected to the next population center in the complex by a web of individual plants spread several hundred yards apart.

The Mesa Verde cactus is a long-lived perennial with a low reproductive potential. Seeds produced by this species are large and difficult to germinate, often requiring several years of the proper growth conditions for germination to occur. Once the seeds are set they may lie dormant in the soil for many years until the right set of conditions trigger germination (i.e., a dry summer following a wet spring). Reproduction in this species is entirely sexual. Budding occurs from early to late April, flowering occurs in late April to mid-May, fruiting occurs from late May through June, and seed dispersal occurs from mid- to late June (Benson 1982). The cactus is pollinated by a particular species of bee.

The Mesa Verde cactus was federally listed as threatened on October 30, 1979. Critical habitat has not been designated. Since its discovery in 1940, this species has been a favorite for cactus enthusiasts (USFWS 1984c). Even today, the populations are ravaged by hobby collectors and by commercial collectors who can make large profits by selling plants from natural populations. In addition, oil and gas development and pipeline and powerline construction occur throughout the range of this species. Apart from the human impacts, this species it is also beset by a variety of insect predators whose larval stages inflict heavy damage upon the cactus, often resulting in death. There is also a present and future threat of habitat destruction by OHV use on the population sites. The habitat affords a marginal existence for most of the species it supports, and is highly sensitive to disturbance or modification. Once the surface crust is broken it may take years for plant species to recolonize. A possible future threat to this species is agricultural development and the associated pesticide use, which can impact bee pollinators.

Mancos Milk-vetch

The Mancos milk-vetch (*Astragalus humillimus*) occupies bowl-like sandy depressions on nearly flat sheets of exposed sandstone bedrock from Mancos Canyon, Colorado, southward to just south of the San Juan River in San Juan County, New Mexico. It is also found in cracks and fissures in the sandstone and at the base of gentle slickrock inclines (Knight and House 1986). The plants grow on level or near-level sites with full exposure to the sun. Runoff from the surrounding bare rock surfaces tends to concentrate moisture in the crevices and depressions which the plants occupy. The vegetation at the population sites is very sparse, and includes small trees and shrubs, scattered forbs, and grasses. Overall cover is probably less than 5%, and vegetation is largely concentrated in the sandy depressions on the bedrock. Dominant associated species are Indian ricegrass, snakeweed, yucca, and big sagebrush (Barneby 1964a). Also present are scattered small trees, including single leaf ash, Utah juniper, and pinyon pine. Plants are found at elevations ranging from 5,000 to 6,000 feet.

The Mancos milk-vetch is a long-lived, slow-growing perennial herb. The leaves of this species appear in the early spring, along with budding, which may also occur in the fall. Flowering occurs anytime from late April to early May, and fruit (a leguminous pod) begins to appear in late May and may last until mid June. Seed dispersal begins in late June and continues on into July (Barneby 1964a, USFWS 1985e, New Mexico Native Plants Protection Advisory Committee 1984, Knight and House 1986). The painted lady butterfly has been identified as a pollinator for this plant, in addition to honey bees and other insects. Seed dissemination agents are not definitely known, but are likely to include sheet erosion and perhaps rodents (USFWS 1985e, Knight and House 1986). The disjunct distribution of the species impedes the flow of genetic material and the broad general dissemination of seeds.

The Mancos milk-vetch was federally listed as endangered on June 27, 1985. Critical habitat has not been designated. The species is narrowly endemic to a small area, and consists of a very low number of plants, which increases the possibility that one catastrophic disturbance could destroy a substantial portion of the species (USFWS 1985e). Furthermore, plants do not tolerate disturbance well. The major serious threats to the Mancos milk-vetch are disturbance and habitat destruction. The range of the species includes an oil field, and is in the vicinity of drilling pads, oil wells, pipelines, and roads, where the possibility of future exploration and drilling is high. Disturbed areas within the species' habitat that resulted from the construction of transmission powerlines have not been recolonized. Plants underneath the powerlines have been driven over by either maintenance vehicles or off-highway recreational vehicles.

Knowlton Cactus

The Knowlton cactus (*Pediocactus knowltonii*) occurs in pinyon-juniper communities on the Colorado plateau of northwestern New Mexico. The species is endemic to San Juan County, New Mexico, and possibly adjacent Colorado, in Archuleta County. Plants grow on rolling, gravelly hills between 6,400 and 7,200 feet in elevation. The habitat is an open-spaced woodland with pinyon pine and Rocky Mountain juniper pinyon as dominants, and big sagebrush as the subdominant species (Brown 1982).

The Knowlton cactus is a stem succulent with no permanent visible leaves. Plants undergo both sexual and vegetative reproduction. Budding occurs in early to mid-April, and flowering occurs from mid April to early May. Flowers open by mid-morning and close in the late afternoon. Typically, they last 2 to 3 days. Plants fruit from late May to early June, and seeds are dispersed in mid to late June (USFWS 1985f). Pollinators are believed to be ants, and seed dispersal agents include water, birds, and rodents (Knight 1981).

The Knowlton cactus was federally listed as endangered on October 29, 1979. Critical habitat has not been designated. The Knowlton cactus is one of the rarest cactus species in the United States. Since its discovery, the plant has been over-collected by botanists and cactus dealers. In recent years, collecting pressures have not been as great and some recovery has been observed. However, at the present population level, it is easily conceivable that the act of one collector could eliminate the species. The Los Pinos River Valley has excellent potential for recreational development. Although Knowlton cactus habitat itself would not be sold for such development, the influx of people to the area could have adverse effects on the cactus (Heil and Porter 1985).

Zuni Fleabane

The Zuni fleabane (*Erigeron rhizomatus*) occurs in pinyon-juniper woodlands in Catron and McKinley counties, New Mexico, and Apache County, Arizona. The species is found on nearly barren detrital clay hillsides on shale-derived soils, at elevations between 7,300 and 8,000 feet. Plants prefer slopes of up to 40 degrees and north-facing aspects, but may also occur on eastern and western exposures. Common associates include pinyon pine, oneseed juniper, Gambel oak, fourwing saltbush, and mountain mahogany (Fletcher 1978, Martin and Hutchins 1980, Sabo 1982).

The Zuni fleabane is often a rather diffusely distributed species. A dense population might contain a few hundred plants spread over several acres. The primary limiting factor for this species is the presence or absence of its preferred microhabitat, which is highly specific and very susceptible to disturbance. Habitat requirements include the proper substrate, at the right elevation, on a gentle slope with the right exposure, as described above. However, even if all of these conditions are met, the plant might not occur on the site.

The Zuni fleabane is a long-lived herbaceous perennial that forms large rhizomatus clumps (Cronquist 1947). The plant reproduces sexually, as well as asexually by pronounced spreading rhizomes that lead to the formation of localized clonal groups. The germination date is likely early spring, and leaves appear from late March to early April. Budding occurs from late April through May, and flowers are produced from late May through June, with fruit noticeable between mid-June and August. Seed dispersal occurs from late June through August (Cronquist

1947, USFWS 1986). This species is pollinated by a variety of insects, and seeds are wind- and possibly animal and/or bird-dispersed.

The Zuni fleabane was federally listed as threatened on April 26, 1985, but critical habitat has not been designated. The chief cause of mortality for this species is surface disturbance. This species seems to be intimately associated with formations that contain known reserves of uranium, and plants are therefore threatened by disturbances caused by exploration, mining, transportation and processing of uranium ore (i.e., habitat destruction and heavy equipment resulting in surface disturbance; USFWS 1986). The soil on which the Zuni fleabane grows is a highly erodable clay that can be disturbed by such activities as trampling associated with grazing, and OHV traffic.

Sacramento Prickly Poppy
The primary reference for this section is:
Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation: Wildland Urban Interface Fuel Treatment. Forest Service, Southwestern Region. Albuquerque, New Mexico.

The Sacramento prickly poppy (*Argemone pleiacantha* ssp. *pinnatisecta*) is a robust perennial species that occurs in canyons of the west side of the Sacramento Mountains in New Mexico. The species favors disturbed areas that are either semi-riparian or have a reliable seasonal provision of water. Thus, it appears particularly adapted to the periodic flooding of normally dry to intermittently perennial canyons. The plant is also often found at springs, and appears to be able to withstand permanently wet sites as long as the soils are well drained. Mature plants are often found in drier sites such as terraces above the normal level of flood flows. The Sacramento prickly poppy is known to occur in seven canyon systems: Fresnal, Dry, Alamo, Mule, San Andres, Dog, and Escondido. In total, approximately 80% of the species' range is on National Forest system lands, 18% is on privately-owned land, and the remainder is on lands administered by the BLM.

The Sacramento prickly poppy is adapted to withstand some scouring by summer floods, which may encourage seed germination. However, loss of riparian vegetation in Alamo Canyon as a result of water diversion has increased the scouring intensity of flood events, rendering much of the active channel either less suitable or unsuitable for the species. Loss of the system's ability to capture fine material also makes the channels drier, reducing survivability of seedlings that do germinate. Seedlings are readily desiccated, and survival is limited to sites with higher moisture availability or to periods of above average precipitation. With the capture of most perennial flows on the west face of the Sacramento Mountains for use in the valley below, the amount of suitable habitat has been much reduced. Pipeline ROW and roadsides provide the reduced vegetative competition and increased moisture the plant requires, and frequently serve as artificial habitat for a substantial number of plants. Once established, plants can survive for years in places that are ordinarily too dry for seedling germination and survival.

The Sacramento prickly poppy was federally listed as endangered on August 24, 1989. Critical habitat has not been designated for this species. It is likely that most of the remaining plants currently occupy the extreme margins of what can be considered suitable habitat. It is not known how much occupied habitat was depopulated when water was developed for human use. The loss of at least seasonal flows out of the canyon and across the bajadas of the west slope could have resulted in the loss of at least as many plants as exist today in the degraded conditions of the canyon proper.

Kuenzler Hedgehog Cactus
The Kuenzler hedgehog cactus (*Echinocereus fendleri* var. *kuenzleri*) occurs in the central highlands of New Mexico. Populations are found in Chaves, Eddy, Lincoln, and Otero counties, on the southern side of the Capitan Mountains, on the eastern and northwestern lower sides of the Sacramento Mountains, and on the northern end of the Guadalupe Mountains. The Kuenzler hedgehog cactus is normally found on gentle slopes or near the shoulders of hilltops or hillsides, at elevations from 5,800 to 6,400 feet (Fletcher 1979b). This species is a minor component of the lower fringes of pinyon-juniper woodland, a broad-ranging and stable community (Fletcher 1979b, USFWS 1985g). Within the range of the Kuenzler hedgehog cactus, the dominant species include yerba de pasmo, blue

grama, plains lovegrass, Harvard's buckwheat, eggleaf silktassle, ribbed false pennyroyal, alligator juniper, oneseed juniper, trong bladderpod, little nipple cactus, pinyon pine, and mealycup sage.

The Kuenzler hedgehog cactus reproduces exclusively by sexual reproduction, and is unable to reproduce vegetatively by fragmentation like other species of cactus. There are no defined germination dates for this species. It appears that it can germinate during any part of the spring, summer, or fall if sufficient rainfall is present. Budding occurs in April, and flowering normally occurs in early May, although the species can flower earlier in warm, wet years. Fruits form in August, and the dispersal of seeds, which typically occurs in September and October, is dependent on the abundance of summer rainfall. If the summer season is good, and the fall food supply for rodents is high, then seed dispersal may be prolonged. Conversely, if the summer is dry, and food supplies are low, then rodents will attack the fruit as soon as it matures. Pollinators are primarily bees, and to a lesser degree beetles and butterflies. Seed dissemination agents include rodents, wind, and water. Seeds are over 90% viable, and survive about 5 years.

The Kuenzler hedgehog cactus was federally listed as endangered on October 26, 1979. Critical habitat has not been designated. It appears that there are few natural threats to the species, and that individuals protected from man made factors die from old age. Although most of the area in which the species occurs is relatively open with little ground cover, it is believed that at one time stands of grass covered the region, which may have acted as a crucial element in catching seeds and hiding seedlings from herbivores. The removal of grass and forb cover from the pinyon-juniper woodland appears to be the major factor contributing to the overall decline of this species. However, the construction of highways throughout the region also resulted in loss of habitat. At present, the major cause of mortality is destruction by grazing, as cattle, sheep, and other grazers remove essential grass cover. The species is also sensitive to trampling that is associated with grazing activities. Other threats to the species include illegal collection and development.

Todsen's Pennyroyal
The primary reference for this section is:
USFWS. 2001b. Todsen's Pennyroyal (*Hedeoma todsenii*) Revised Recovery Plan. USFWS. Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

Todsen's pennyroyal (*Hedeoma todsenii*) is a rhizomatous perennial that is known from the San Andreas and Sacramento mountain ranges of southern New Mexico, where it occurs in loose gypseous-limestone soils. The species occurs in the Great Basin conifer woodland community where pinyon pine and oneseed juniper are the dominant species (Brown and Lowe 1980). Other common associates include mountain mahogany, yellowleaf silktassle, wavyleaf oak, white ragweed, snakeweed, and muhly grass. The species grows in the shade of pinyon pines and junipers, and in woodland openings with thin grasses. Most plants are on steep (20 to 70 degree) north-facing slopes, with a surface of scree or gravelly cobble. The substrates have a thin layer of conifer litter over a mixture of limestone and finer materials. In general, these gypsum-derived soils appear to retain more moisture than other soils in similar situations (New Mexico Forestry and Research Conservation Division 1992).

In the San Andres Mountains, there are three sites supporting Todsen's pennyroyal, all of which occur on the White Sands Missile Range in Sierra County, New Mexico. In the Sacramento Mountains in Otero County, there are a total of 15 sites. There are often thousands of stems on a single site; however, the number of genetically distinct individuals is unknown because of the highly rhizomatous nature of the plants. An entire population could potentially be one genetic individual interconnected through this rhizome system (New Mexico Forestry and Research Conservation Division 1991).

Todsen's pennyroyal exhibits low sexual reproduction, with less than 20% of clumps flowering per season (New Mexico Forestry and Research Conservation Division 1992). Seed set is also low. The species flowers from June to September, with most flowers produced from late August to early September, concurrent with the period of highest rainfall. The flowers appear to be specialized for hummingbird pollination; however, hummingbirds only rarely visit plants (New Mexico Forestry and Research Conservation Division 1992, Huenneke 1993, Ulaszek 1993). Because most reproduction is asexual through an underground rhizome, a population of this species can potentially occupy all suitable habitat at a specific locality. Although these large populations are probably able to survive droughts, floods, and other natural disasters, if a population were eradicated, the species would be unlikely to recolonize that locality because of low seed production and poor seed dispersal.

Todsen's pennyroyal was federally listed as endangered on January 19, 1981. Two parcels of critical habitat, each 0.6 square miles in size, were designated on the White Sands Missile Range. The relatively remote or inaccessible locations of Todsen's pennyroyal afford the species some protection. Yet, because of the fragile nature of the habitat and the small size of some populations, accidental disturbances or changes in land use could destroy them. Potential threats to the species include livestock grazing, future military activities, mammal and insect herbivory, and low genetic diversity. There is no information on how fire affects Todsen's pennyroyal. The species would be expected to resprout after fire, and a potential decrease in competition for light, water, and nutrients could result in greater vigor. However, increased erosion and reduced soil moisture could adversely affect populations.

Temperate Steppe Ecoregion Division

The Temperate Steppe Ecoregion Division includes areas with a semiarid continental climate (i.e., evaporation typically exceeds precipitation) in the Rocky Mountains and Great Plains regions. Important communities in this ecoregion division include the shortgrass and mixed grass prairies of the Great Plains, the Northwest bunchgrass prairies (also called Palouse grasslands), and evergreen and deciduous forests, woodlands, and shrublands.

Western Prairie Fringed Orchid

The primary reference for this section is:
USFWS. 1996b. *Platanthera praeclara* (Western Prairie Fringed Orchid) Recovery Plan. USFWS. Fort Snelling, Minnesota.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The western prairie fringed orchid (*Platanthera praeclara*) is a perennial orchid of the North American tallgrass prairie that is found most often on unplowed, calcareous prairies and sedge meadows. This species may also occur at disturbed sites in successional communities, such as borrow pits, old fields, and roadside ditches (Minnesota Department of Natural Resources 1979 to present, Nebraska Games and Parks Commission 1987 to present, Freeman and Brooks 1989). Populations occur in six states: North Dakota, Minnesota, Iowa, Nebraska, Kansas, and Missouri.

The western prairie fringed orchid occurs in several kinds of fire- and grazing-adapted communities dominated by grass species. The tallgrass prairies in which the orchid occurs are typically dominated by big bluestem, little bluestem, and Indiangrass, with tufted hairgrass and switchgrass common associates in wetter sites. These prairies generally support a great variety of annual and perennial forbs and grasses, with few shrubs unless fire or grazing is suppressed. The orchid generally occurs within the wetter areas of such prairies or in associated sedge meadows. Sedge meadows occur in seasonally hydric to wet-mesic conditions, and are dominated by sedges and spikerushes. A variety of annual and perennial grasses and forbs also occurs in this community type, with shrubs becoming increasingly prevalent northward.

Root systems of the genus *Platanthera*, including the western prairie fringed orchid, are tubers that regenerate during the growing season by forming a new tuber and a bud, which gives rise to vegetative shoots the following season. This asexual reproduction is the main mode of perpetuation of established populations. Vegetative shoots develop from a bud and emerge from the soil in the late spring after a period of soil warming, which usually occurs from mid April in the southern portion of the species' range to late May in the northern portion (Pleasants 1995). Two months of vegetative growth may pass before an inflorescence will fully develop into a flowering plant.

Studies suggest that it is also common for the orchid to remain vegetative throughout the entire growing season (Sather and Smith 1994, Sieg and King 1995). Sexual reproduction is believed to be the principal means of recruitment of new individuals into populations (Bowles 1983, Bowles and Duxbury 1986). Plants bloom from mid-June in the southern portion of the range to late July in the northern portion. Individual flowers last up to 10 days, and inflorescences may produce flowers for up to 3 weeks.

Pollination is required for seed production, with moths thought to be the primary pollinators. Seeds mature on the plant in capsules and are released in early fall (Bowles and Duxbury 1986). A single capsule may produce thousands of seeds. Therefore, under ideal circumstances for germination and survivorship, the reproductive potential of a small population could be very large. Seeds are wind-dispersed, and may also be adapted for dissemination through the soil profile by water (Bowles 1983). Growth of orchid seedlings in natural conditions requires association with soil-inhabiting mycorrhizal fungus (Cronquist 1981, Bowles and Duxbury 1986, Currah et al. 1990). Seedling establishment may also be linked to the availability of suitable microhabitats, edaphic factors controlling soil mycorrhizae, and interspecific competition.

Habitat management, such as burning, grazing, or mowing, could have a positive or negative effect on recruitment and survivorship, depending on its frequency, intensity, and timing. It has been suggested that flowering may be suppressed by plant litter accumulation and stimulated by fire (Bowles 1983, Bowles and Duxbury 1986). The effect of fire on flowering is probably influenced by intensity and timing of the burn and weather conditions both at the time of the burn and the time of flowering.

The western prairie fringed orchid was federally listed as threatened on September 18, 1989. Critical habitat has not been designated. The prairie fringed orchids have declined substantially throughout their ranges as a result of conversion of most of their habitats to cropland, overgrazing, intensive hay mowing, drainage, and fire protection; these and related threats continue. Other factors threatening the species include herbicide use, poor reproduction, collection, alteration of the water regime, and competition with non-native and other invasive species.

Blowout Penstemon
The primary reference for this section is:
USFWS. 1992c. Blowout Penstemon (*Penstemon haydenii*) Recovery Plan. Denver, Colorado.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The blowout penstemon is a perennial herb that occurs naturally only within the Sandhills region of north-central Nebraska (Weedon et al. 1982a). Plants are found on dune blowouts, or depressions in the topography caused by wind erosion, where vegetation is distinctly different than vegetation associated with adjacent noneroding areas (Stubbendieck et al. 1989). Commonly associated plant species include blowout grass, lemon scurfpea, sandhill muhly, prairie sandreed, and birdegg milk-vetch. Blowout penstemon is a primary invader of blowouts, disappearing from the site once secondary invasion of the blowout begins (Tolstead 1942; Weedon et al. 1982b; Flessner 1988). Therefore, the species is dependent on continuing wind erosion, or some other source of new blowouts. The stems of blowout penstemon root adventitiously, stabilizing the plant in shifting sands.

The blowout penstemon reproduces primarily by rhizomes, and naturally occurring seedlings are relatively rare (Stubbendieck et al. 1983, 1984; Stubbendieck and Weedon 1984). It appears that the species is dependent on vegetative reproduction for survival. Plants flower from mid-May through mid- to late June, develop fruits from late May through early July, and begin dispersing seeds in late July or early August. The species is commonly pollinated by insects, primarily bees (Flessner and Stubbendieck 1992).

The blowout penstemon was federally listed as endangered on September 1, 1987. Critical habitat has not been designated. Because the Nebraska Sandhill region is used primarily for cattle grazing, range management in the area focuses on stabilizing the sand dunes with later successional species. These activities result in a reduction in available blowout penstemon habitat and numbers of plants. Use of OHVs in penstemon habitat is an additional minor threat to the species.

Colorado Butterfly Plant

The primary reference for this section is:

USFWS. 2000a. Threatened Status for the Colorado Butterfly Plant (*Gaura neomexicana* ssp. *coloradensis*) from Southeastern Wyoming, Northcentral Colorado, and Extreme Western Nebraska. Federal Register 65(202): 62302-62310.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Wyoming Field Office. Cheyenne, Wyoming.

Colorado butterfly plant (*Gaura neomexicana* ssp. *coloradensis*) is endemic to moist soils in mesic or wet meadows of floodplain areas in north central Colorado, extreme western Nebraska, and southeastern Wyoming. This subspecies occurs primarily in habitats created and maintained by streams active within their floodplains, with vegetation that is relatively open and not overly dense or overgrown. Colonies are often found in low depressions or along bends in wide, active, meandering stream channels a short distance upslope of the actual channel. The plant requires early- to mid-successional riparian habitat. It commonly occurs in communities dominated by redtop and Kentucky bluegrass on wetter sites, and wild licorice, Flodman's thistle, curlytop gumweed, and smooth scouring rush on drier sites.

Colorado butterfly plant is a perennial herb that lives vegetatively for several years before bearing fruit once and then dying. Only a few flowers are open at any one time, and these are located below the rounded buds and above the hard, nutlike fruits. Nonflowering plants consist of a stemless, basal rosette of leaves. Colorado butterfly plant is an early successional plant that is adapted to use periodically disturbed stream channel sites. Historically, flooding was probably the main cause of disturbances in the plant's habitat, although wildfire and grazing by native herbivores also may have been important. Although flowering and fruiting stems may undergo increased mortality because of these events, vegetative rosettes appear to be little affected (Mountain West Environmental Services 1985). In addition, the establishment and survival of seedlings appears to be enhanced at sites where tall and dense vegetation has been removed by some form of disturbance. In the absence of occasional disturbance, the plant's habitat can become choked out by dense growth of willows, grasses, and non-native plants.

All currently known populations are within a small area (17,000 acres) in southeastern Wyoming, western Nebraska, and north-central Colorado. Two of the populations occur on F.E. Warren Air Force Base in Cheyenne, Wyoming, and five small populations on state land (Chambers Preserve, Colorado; Oliver Reservoir State Recreation Area, Nebraska; and state school trust land, Wyoming). One population occurs on the Meadow Springs Ranch, northern Colorado (owned by City of Fort Collins). The remaining populations occur on privately-owned lands.

The Colorado butterfly plant was federally listed as threatened on October 18, 2000. On January 5, 2005, USFWS designated 8,486 acres along approximately 113.1 stream miles in Laramie and Platte counties, Wyoming; Kimball County in Nebraska; and Weld County in Colorado, as critical habitat. Threats include the indiscriminate spraying of broadleaf herbicides and the disturbance of riparian areas that contain native grasses, water diversions, channelization, and urban development.

North Park Phacelia

The North Park phacelia (*Phacelia formosula*) is a narrow endemic of an area in northern Colorado known as North Park. The species occurs on barren exposures where the Coalmont Formation forms outcrops of sandy soil or ledges containing pockets of sandy soil. Vegetative cover is very low, and the barren outcrops are contained in a matrix of sagebrush communities (Peterson and Wiley-Eberle 1984, Colorado Department of Wildlife 1985). The area is considered rangeland. North Park phacelia appears to prefer steep-sided ravines, although relatively flat areas support the species in low numbers if the soil is nearly pure sand and is nearly devoid of vegetative cover. Slopes and aspects are variable and elevations range from 8,000 to 8,200 feet. The North Park phacelia is dominant or co-dominant on the sites on which it is found. Commonly associated plants include species of blazingstar, rabbitbrush, ricegrass, sandwort, buckwheat, beardtongue, rose, sagebrush, and phlox.

The North Park phacelia is a biennial or short-lived perennial herb that does not reproduce by vegetative means. Germination occurs in spring, and leafing occurs in late spring to early summer. Flowering and fruiting occurs from July to August, and seeds are dispersed from July to September (Peterson and Wiley-Eberle 1984, Colorado Department of Wildlife 1985). Pollinators are insects, and seed dissemination agents are wind, water, and possibly ants. Seed production is directly dependent on the number of plants maturing in any particular year. Since the species is a biennial or short-lived perennial, the climate 2 years prior to any seed crop is the primary factor influencing seed production.

The North Park phacelia was federally listed as endangered on September 1, 1982, but critical habitat has not been designated. The sandy areas in which it occurs are vulnerable to habitat destruction because of their extremely friable nature and very sparse vegetation cover. Cattle tend to disrupt the sand, causing plants to be uprooted, and plants are trampled by grazing animals. Off-highway vehicle usage at one of the two largest occurrences of this species has resulted in severe disturbance of the site (Wiley-Eberle 1979; Peterson and Wiley-Eberle 1984). Road work around the known sites, cattle trampling, and OHVs have added to erosion, yet another factor causing loss of habitat and individuals. The rarity of the species itself is a threat, and some populations are so small that the gene pool is restricted. Finally, the area of occurrence has potential for low-grade coal and oil and gas production. These activities, as well as seismic and geothermal exploration, may become important, should exploration and extraction become profitable in the future.

Spalding's Catchfly
The primary reference for this section is:
USFWS. 2001c. Final Rule to List *Silene spaldingii* (Spalding's catchfly) as Threatened. Federal Register 66 (196): 51598-51606.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Snake River Basin Office, Boise, Idaho.

Spalding's catchfly (*Silene spaldingii*) is primarily restricted to mesic grasslands that make up the Palouse region in southeastern Washington, northwestern Montana, adjacent portions of Idaho and Oregon, and British Columbia. Palouse prairie is considered a subset of the Pacific Northwest bunchgrass habitat type (Tisdale 1986). Spalding's catchfly is also found in canyon grassland habitat, which is another division of the Pacific Northwest bunchgrass habitat type. Canyon grasslands are dominated by the same bunchgrass species as the Palouse prairie, but the two habitat types differ in their overall plant species composition (Hill 2000, Yuncevich 2000). In addition, canyon grasslands occur in steep, highly dissected canyon systems, whereas Palouse grasslands generally occur on gently rolling plateaus. The steep slopes in canyon grasslands result in pronounced habitat diversity (Yuncevich 2000). This steepness has also prevented the conversion of canyon grasslands to other uses, such as agriculture.

Spalding's catchfly is typically associated with grasslands dominated by native perennial bunchgrasses such as Idaho fescue or rough fescue. Other associated species include bluebunch wheatgrass, prairie Junegrass, snowberry, Nootka rose, yarrow, prairie smoke avens, sticky purple geranium, and arrowleaf balsamroot (Lichthardt 1997, Montana Natural Heritage Program 1998). Scattered individuals of ponderosa pine may also be found in or adjacent to Spalding's catchfly habitat. Sites on which Spalding's catchfly occurs range from approximately 1,500 feet to 5,100 feet in elevation (Oregon Natural Heritage Program 1998, Washington Natural Heritage Program 1998).

At the time of listing in 2001, this species was known from a total of 52 populations in the United States and British Columbia, 51 of which were in the United States (7 in Idaho, 7 in Oregon, 9 in Montana, and 28 in Washington). The range of individuals in each population ranges from one to several thousand. Much of the remaining habitat occupied by Spalding's catchfly is fragmented, with clusters of populations geographically isolated from one another.

Spalding's catchfly is a long-lived perennial herb that ranges from 8 to 24 inches in height (Lichthardt 1997). The species does not possess rhizomes or other means of vegetative reproduction, and reproduces by seed only (Lesica 1992). Plants are typically pollinated by bumblebees, which appear to be critical to population viability (Lesica 1993).

Spalding's catchfly was federally listed as threatened on October 10, 2001. At the time of listing, designation of critical habitat was deemed prudent, but was deferred until resources become available. Large-scale ecological changes in the Palouse region over the past century, including agricultural conversion, changes in fire frequency, and alterations of hydrology, have resulted in the decline of Spalding's catchfly. More than 98% of the original Palouse prairie habitat has been lost or modified be agricultural conversion, grazing, invasions of non-native plant species, altered fire regimes, and urbanization (Noss et al. 1995). In addition, the less accessible canyon grasslands have been disturbed by livestock grazing and the invasion of non-native plant species. Threats to this species include habitat destruction and fragmentation resulting from agriculture and urban development, grazing and trampling by domestic livestock and native herbivores, herbicide treatment, and competition from non-native plant species.

Howell's Spectacular Thelypody

The primary reference for this section is:
USFWS. 1999c. Threatened Status for the Plant *Thelypodium howellii* ssp. *spectabilis* (Howell's Spectacular Thelypody). Federal Register 64(101): 28393-28403.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

Howell's spectacular thelypody (*Thelypodium howellii* var. *spectabilis*) occurs in moist, alkaline meadow habitats at approximately 3,000 feet to 3,500 feet elevation in northeastern Oregon. The plant is currently known from 11 sites (five populations) ranging in size from 0.03 to 41 acres in the Baker-Powder River Valley in Baker and Union counties. The total occupied habitat for this species is approximately 100 acres, and its range lies entirely within a 13-mile radius of Haines, Oregon. Howell's spectacular thelypody usually grows in valley bottoms around woody shrubs that dominate the habitat on the knolls, and along the edge of wet meadow habitat between the knolls. Associated species include greasewood, alkali saltgrass, giant wild rye, alkali cordgrass, and alkali bluegrass (Kagan 1986). Soils are pluvial-deposited alkaline clays mixed with recent alluvial silts, and are moderately well-drained.

Howell's spectacular thelypody is an herbaceous biennial that reaches a height of approximately 2 feet, with branches arising from near the base of the stem. Flowers are purple and borne on short stalks, and fruits are long, slender pods (Greenleaf 1980, Kagan 1986). The taxon may be dependent on periodic flooding, since it appears to rapidly colonize areas adjacent to streams that have flooded (Kagan 1986). In addition, this taxon does not compete well with encroaching weedy vegetation such as teasel (Davis and Youtie 1995).

Howell's spectacular thelypody was listed as threatened on May 26, 1999. Critical habitat has not been designated. Factors that threaten this taxon include habitat destruction and fragmentation caused by agricultural and urban development, grazing by domestic livestock, competition from non-native vegetation, and alteration of wetland hydrology.

McFarlane's Four-o'clock

The primary reference for this section is:
USFWS. 1996c. Reclassification of *Mirabilis Macfarlanei* (MacFarlane's Four-o'clock) from Endangered to Threatened Status. Federal Register 61(52): 10693-10697.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Snake River Basin Office. Boise, Idaho.

MacFarlane's four-o'clock (*Mirabilis macfarlanei*) is found on talus slopes in canyonland corridors where the climate is regionally warm and dry, and where precipitation occurs mostly during the period from winter to spring. It can be found in three disjunct areas in Oregon and Idaho that are associated with the Snake, Salmon, and Imnaha rivers. The species occurs as scattered plants on open, steep (50%) slopes of sandy soils, which generally have a west to southeast aspect. Talus rock underlies the soil in which the plants are rooted. Although a variety of soils support this plant throughout its range, the more common sandy soils are quite susceptible to displacement by wind and water erosion.

The plant community in which MacFarlane's four-o'clock occurs is a transition zone between bluebunch wheatgrass-Sandberg bluegrass and smooth sumac-bluebunch wheatgrass, consisting of bluebunch wheatgrass, downy brome, sand dropseed, scorpion weed, desert parsley, hackberry, smooth sumac, yarrow, and rabbit bush (Daubenmire 1970, Franklin and Dyrness 1973).

One geographic unit of MacFarlane's four-o'clock includes approximately 25 acres along 6 miles of Hells Canyon on the banks and canyonland slopes above the Snake River in Idaho County, Idaho and Wallowa County, Oregon. The second geographic unit includes approximately 68 acres along 18 miles of banks and canyonland slopes above the Salmon River in Idaho County, Idaho. The third geographic unit includes about 70 acres of habitat along 3 miles of canyonland slopes over the Imnaha River in Wallowa County, Oregon.

MacFarlane's four-o'clock is a perennial plant with a stout, deep-seated taproot. Flowering occurs from early May to early June, and peaks in mid May.

MacFarlane's four-o'clock was federally listed as endangered on October 26, 1979. After additional populations were discovered, the plant was reclassified as threatened on March 15, 1996. Critical habitat has not been designated. Threats to the species include lack of plant recruitment in some areas, insect predation, invasions of non-native plants (often as a result of grazing practices), and the small size of some populations.

Osterhout Milk-vetch
The primary reference for this section is:
USFWS. 1992d. Osterhout Milk-vetch and Penland Beardtongue Recovery Plan. Denver, Colorado.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The Osterhout milk-vetch (*Astragalus osterhoutii*) is endemic to Middle Park, a high-elevation sagebrush park located near Kremmling, Colorado, in Grand County. Middle Park is located at an elevation of approximately 7,500 feet and surrounded by various ranges of the Rocky Mountains. The Osterhout milk-vetch occurs in scattered colonies over a 15-mile range, from 3 miles east of Troublesome Creek to a few miles west of Muddy Creek. A majority of plants occur on land administered by the BLM, although important colonies also occur on privately-owned land.

Plants are restricted to badlands of shale and siltstone sediments. These badlands are characterized by open, grassy vegetation with scattered shrubs of big sagebrush, rabbitbrushes, bitterbrush, horsebrush, winterfat, snowberry, and/or mountain mahogany. Common perennials include lupine and wild buckwheat. Where shrubs—particularly big sagebrush—have increased in density, resulting in a more closed shrubland vegetation type, the Osterhout milk-vetch is reduced in density. This species shows evidence of light grazing, and can be found on old road cuts and fills, indicating some tolerance for disturbance (Bio/West 1989).

The Osterhout milk-vetch has white flowers and long, pendulous fruits. Flowers are pollinated by bees.

The Osterhout milk-vetch was federally listed as endangered on July 13, 1989. Critical habitat has not been designated. The Osterhout milk-vetch is a naturally rare species, limited to the small existing area of available habitat in the desert badlands. In addition, it is disjunct 150 miles from its nearest relatives; expansion and migration to potentially suitable habitats elsewhere is blocked by the high mountains surrounding Middle Park. Threats to the Osterhout milk-vetch include water projects along Muddy Creek, grazing, and oil and gas exploration and development. In addition, the density of Osterhout's milk-vetch has been observed to be lower in big sagebrush stands than in the adjacent open benchlands where it normally grows. It may be that the past grazing history has caused an increase in big sagebrush density with a resultant increase in competition for soil moisture. The Osterhout milk-vetch may then be outcompeted and populations reduced in numbers or lost entirely where big sagebrush dominates.

Penland Beardtongue
The primary reference for this section is:
USFWS. 1992d. Osterhout Milk-vetch and Penland Beardtongue Recovery Plan. Denver, Colorado.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The Penland beardtongue (*Penstemon penlandii*), like the Osterhout milk-vetch discussed in the previous species account, is endemic to Middle Park, Colorado, in Grand County. The Penland beardtongue is rarer than the Osterhout milk-vetch, and is only known to occur along Troublesome Creek. A majority of plants occur on land administered by the BLM, although important colonies of the species also occur on privately-owned land. The Penland beardtongue is limited to siltstone sediments in badlands. These badlands where these species grow are characterized by open, grassy vegetation with scattered shrubs of big sagebrush, rabbitbrushes, bitterbrush, horsebrush, winterfat, snowberry, and/or mountain mahogany. Common perennials include lupine and wild buckwheat.

Little is known about the reproductive biology of the Penland beardtongue, except that it must be visited by animals (including several native bee species) to reproduce sexually.

The Penland beardtongue was federally listed as endangered on July 13, 1989. Critical habitat has not been designated. The Pendland beardtongue, like the Osterhout milk-vetch, is a naturally rare species, limited to the small existing area of available habitat in the desert badlands. It is also disjunct 150 miles from its nearest relatives, and expansion and migration to potentially suitable habitats elsewhere is blocked by the high mountains surrounding Middle Park. Threats to the species include water projects along Muddy Creek, grazing, and oil and gas exploration and development.

Penland Alpine Fen Mustard
The primary reference for this section is:
USFWS. 1993e. The Plant *Eutrema penlandii* (Penland Alpine Fen Mustard) Determined to be a Threatened Species. Federal Register 58(143): 40539-40547.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The Penland alpine fen mustard (*Eutrema penlandii*) occurs in alpine tundras of Colorado, where small populations of the plant are distributed in a 25-mile stretch of the Continental Divide. The species is habitat-specific, growing only in oligotrophic (nutrient deficient), rheotrophic (groundwater fed) alpine marshes (Weber and Shushan 1955). It grows in a macroclimate of long, cold, wet winters and cool, windy summers, and a microclimate of relatively protected, wet, springy bogs (Johnston et al. 1981). Major components of its microenvironment include moss-covered peat fens, perennial subirrigation, and high elevations (above 12,150 feet).

The peat mats on which the alpine fen mustard grows form on small, flat to gently sloping benches in steep-walled, rounded glacial valleys. Water required for the development and sustenance of these peat mats comes from snowfields that persist through the summer. Conditions for maintaining these persistent snowfields exist along the east-west trending portion of the Continental Divide, where the plant is found on slopes (Schwendinger et al.

1991). The alpine fen mustard is found on deep organic soils in moist areas that are usually adjacent to clear running water from snowmelt. Plant emergence at a site appears to be dependent on the availability and timing of sufficient water to continuously moisten the mosses in which the plants are rooted, but not so much water as to flood them.

The Penland alpine fen mustard is a small, herbaceous, perennial plant that grows up to about 3 inches in height. Clusters of small, white flowers grow atop the plants' stems. A plant of the Colorado alpine tundra, the alpine fen mustard grows in a harsh environment, with a growing season that may only last 70 days per year (Colorado Native Plant Society 1989). In addition, freezing and thawing soil, drying winds, and windblown snow and ice crystals diminish plant productivity (Zwinger and Williard 1972).

The Penland alpine fen mustard was federally listed as threatened on July 28, 1993. Critical habitat has not been designated for the species. The wetland habitat in which the species occurs is fragile, and sensitive to watershed alterations that divert flows of surface water. Direct impacts to plants and habitats occur from mining, and from OHV use and other forms of recreation. In addition the few small populations of the species on small areas of specialized habitat make it particularly vulnerable to human disturbances as well as random environmental occurrences.

Mediterranean Ecoregion Division

The Mediterranean Ecoregion Division includes most of California and a portion of southern Oregon. The Mediterranean climate in this region is characterized by dry, hot summers and wet, mild winters. Chaparral, a fire- and drought-adapted vegetation type that is comprised of hard-leaved evergreen trees and shrubs, is endemic to the Mediterranean Ecoregion Division. A number of TEP plant species occur in chaparral communities. Chaparral communities, which have been altered by fire suppression, often pose a threat to encroaching human populations because of the large amount of highly flammable fuels found in these communities. Vernal pools, seasonal ponds that fill during winter rains and dry up during the summer drought also provide habitat for TEP species in this ecoregion. Other communities include grasslands that once supported perennial native grasses but now support primarily non-native annuals, sagebrush, coastal scrub, and the forests and woodlands of the Sierra Nevada Mountains and surrounding foothills.

Mediterranean climates have numerous rare, locally endemic species. For this reason, a total of 63 TEP plant species with the potential to be affected by BLM treatment activities occur in the Mediterranean Ecoregion Division, a greater number than in any other division.

Gentner's Fritillary
The primary reference for this section is:
USFWS. 1999d. Final Endangered Status for the Plant *Fritillaria gentneri* (Gentner's fritillary). Federal Register 64(237): 69195-69203.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Oregon Fish and Wildlife Office. Portland, Oregon.

Gentner's fritillary (*Fritillaria gentneri*), an herb of the lily family, is restricted to southwestern Oregon, where it is known only from scattered localities in the Rogue and Illinois River drainages in Josephine and Jackson counties. The species occurs in dry, open woodlands of fir or oak at elevations below approximately 4,450 feet. The species is highly localized within a 30-mile radius of Jacksonville Cemetery, and 73% of the known plants are in a central cluster located within a 7-mile radius of the cemetery. The remaining plants occur as single individuals or occasional clusters of individuals sparsely distributed across the landscape. Plants occur on lands managed by the Medford District of the BLM, the Oregon State Department of Transportation, Southern Oregon University, and the City of Jacksonville, as well as on privately-owned land (about half of the plant's current distribution).

Gentner's fritillary is found in three habitats—oak woodlands dominated by Oregon white oak; mixed hardwood forest dominated by California black oak, Oregon white oak, and madrone; and coniferous forests dominated by madrone and Douglas-fir. Gentner's fritillary typically grows in or on the edge of open woodlands with Oregon white oak and madrone as the most common overstory plants. The species can also grow in open chaparral/grassland habitat, which is often found within or adjacent to the mixed hardwood forest type, but always where some wind or sun protection is provided by other shrubs. It does not grow on extremely droughty sites. For unknown reasons, a substantial amount of potential habitat within the species range is unoccupied. Gentner's fritillary often grows in places that have experienced human disturbance and eventually became revegetated (e.g., old road cuts, alongside trails, bulldozer routes, old mounds left from past mining or other earth-moving activities; Rolle 1988). The species seems to require some infrequent, but regular level of disturbance such as the historic pattern of fire frequency in the Rogue and Illinois River valleys. It is not an early colonizer of these sites but eventually takes advantage of the opening or edge effect created. It appears to be a mid-successional species in that it establishes after other plants have colonized a disturbed area, but before taller vegetation becomes established and shades it out.

Gentner's fritillary is a perennial species that reproduces asexually by bulblets. The bulblets break off and form new plants. The flowering season for this species is April-June. However, many of the plants remain dormant for several years and do not produce above-ground stems and flowers. Even though some Gentner's fritillary plants may form fruits and seeds if pollinated, no good evidence exists that the seeds produced are fertile or viable (Guerrant 1997). Hummingbirds or bumblebees are presumed to be the primary pollinators (Guerrant 1998). It is possible that Gentner's fritillary is sterile and that the plant is largely reproducing asexually; however, sexual reproduction of the plant needs to be better documented.

Gentner's fritillary was federally listed as endangered on December 10, 1999. Critical habitat has not been designated. The species is threatened by residential development, agricultural activities, logging, road and trail improvement, OHV use, collection for gardens, and problems associated with small population size. In addition, all three of the habitats in which Gentner's fritillary occurs are threatened by urban and agricultural development and fire suppression.

Ione Manzanita

The primary reference for this section is:

USFWS. 1999e. Determination of Endangered Status for the Plant *Eriogonum apricum* (inclusive of vars. *apricum* and *prostratum*; Ione Buckwheat) and Threatened Status for the Plant *Arctostaphylos myrtifolia* (Ione Manzanita). Federal Register 64(101): 28403-28413.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office. Sacramento, California.

Ione manzanita (*Arctostaphylos myrtifolia*) is found primarily in western Amador County, California, and a few local areas of adjacent northern Calaveras County, in the central Sierra Nevada foothills of California. Most populations occur at elevations between 295 and 918 feet. The species occurs primarily on Ione soils, which have developed along a 40-mile stretch of the Ione Formation. These soils are coarse-textured and exhibit soil properties typical of those produced under tropical climates, such as high acidity, high aluminum content, and low fertility (Singer 1978). In addition, these soils and their associated sedimentary deposits contain large amounts of commercially valuable minerals (quartz sands, kaolinitic clays, lignite [low-grade coal], and possible gold-bearing gravels; Chapman and Bishop 1975).

The vegetation in the Ione area is distinctive enough to receive a special designation as "Ione chaparral" (Holland 1986). This plant community type has been characterized as an ecological island, a relatively small area with climatic and ecological features that differ substantially from the surrounding areas (Stebbins 1993). The entire extent of this community type is estimated at 6,002 acres (California Natural Diversity Database 1997). Because they occur only on very acidic, nutrient-poor, coarse soils, Ione chaparral communities are comprised of low-growing, heath-like shrubs and scattered herbs that are tolerant of these growing conditions (Holland 1986). The dominant shrub is Ione manzanita, which is narrowly endemic to the area.

Ione manzanita is an evergreen shrub of the heath family with a low and spreading appearance. The species depends almost entirely on periodic fire events to promote seed germination (Wood and Parker 1988). As the dominant and characteristic species of Ione chaparral, Ione manzanita occurs in pure stands on outcrops of the Ione Formation. The species also occurs in ecotonal habitat with surrounding taller chaparral types, but does not persist if it is shaded (Woodward 1994). Populations range in elevation from 190 to 1,900 feet, with the largest populations occurring at elevations between 280 and 900 feet (Wood and Parker 1988). It is estimated that Ione manzanita occurs in about 100 individual areas that cover a total of about 1,000 acres (Woodward 1994). Ione manzanita occurs primarily on private or non-federal lands. However, three occurrences are at least partially on public lands, including one occurrence within the Ione Manzanita Area of Critical Environmental Concern. Populations also occur on the state-owned Apricum Hill Ecological Reserve managed by the California Department of Fish and Game (Wood and Parker 1988).

Ione manzanita was federally listed as threatened on May 26, 1999. Critical habitat has not been designated. Factors that threaten populations of Ione manzanita include mining, clearing of vegetation for agriculture and fire protection, habitat fragmentation, residential and commercial development, changes in fire frequency, and ongoing erosion (Bollinger 1994; Wood 1994; California Natural Diversity Database 1997).

Ione Buckwheat
The primary reference for this section is:
USFWS. 1999e. Determination of Endangered Status for the Plant *Eriogonum apricum* (inclusive of vars. *apricum* and *prostratum*; Ione Buckwheat) and Threatened Status for the Plant *Arctostaphylos myrtifolia* (Ione Manzanita). Federal Register 64(101): 28403-28413.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office. Sacramento, California.

Ione buckwheat (*Eriogonum apricum*), like Ione manzanita described above, is found in Ione chaparral communities of the central Sierra Nevada foothills of California, at elevations between 295 and 918 feet. The entire extent of Ione chaparral is estimated at 6,002 acres (California Natural Diversity Database 1997). Because they occur only on very acidic, nutrient-poor, coarse soils, Ione chaparral communities are comprised of low-growing, heath-like shrubs and scattered herbs that are tolerant of these growing conditions (Holland 1986). The dominant shrub is Ione manzanita, which is narrowly endemic to the area.

There are two varieties of Ione buckwheat that occur in Ione chaparral: *Eriogonum apricum* var. *apricum* and *E. apricum* var. *prostratum*. Both varieties are perennial herbs in the buckwheat family. *Eriogonum apricum* var. *apricum* flowers from July to October, and is restricted to occurrences in nine areas occupying a total of approximately 10 acres on otherwise barren outcrops within the Ione chaparral (The Nature Conservancy 1984). This variety occurs primarily on private or non-federal land; however, the BLM administers one area where this species occurs, and another is partially protected by the California Department of Fish and Game (California Natural Diversity Database 1997). *Eriogonum apricum* var. *prostratum* is restricted to otherwise barren outcrops on less than 1 acre of private land in openings of Ione chaparral.

Both varieties of Ione buckwheat were federally listed as endangered on May 26, 1999. Critical habitat has not been designated for either species. Ione buckwheat is threatened by mining, clearing of vegetation for agriculture and for fire protection, habitat fragmentation, increased residential development, and erosion.

Stebbins' Morning-glory

The primary reference for this section is:
USFWS. 1996d. Determination of Endangered Status for Four Plants and Threatened Status for One Plant from the Central Sierran Foothills of California. Federal Register 61(203): 54346-54358.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office. Sacramento, California.

Stebbins' morning-glory (*Calystegia stebbinsii*) occurs in chaparral in western El Dorado County, California. The Pine Hill intrusion, where the species found, is an area of approximately 25,700 acres that ranges in elevation from 453 to 2,060 feet. In addition, Stebbins' morning-glory has a few known isolated occurrences in El Dorado, Nevada, and/or Tuolumne counties, California.

Stebbins' morning-glory is a leafy perennial herb in the morning-glory family with stems that range up to 3.3 feet in length and generally lie flat on the ground. Flowers appear on stalks in May through June, and the fruit is a slender capsule. Most occurrences of this species are discontinuously scattered within two population centers in the northern and southern portions of the Pine Hill intrusion. In El Dorado County, the species is associated with chaparral on gabbro-derived soils. In Nevada County it occurs on serpentine soils. Gabbro-derived soils originate from mafic rocks (gabbrodiorite) that are mildly acidic, are rich in iron and magnesium, and often contain other heavy metals such as chromium (Wilson 1986). Serpentine-derived soils are formed through a similar process, but are derived from ultramafic rocks (e.g., serpentinite, dunite, and peridotite). They tend to have high concentrations of magnesium, chromium, and nickel, and low concentrations of calcium, nitrogen, potassium, and phosphorus (Kruckeberg 1984). Stebbins' morning-glory occurs primarily on privately-owned land, although the BLM administers land harboring some occurrences. Development has extirpated at least one-third of the known occurrences (California Department of Fish and Game 1990).

Stebbins' morning-glory was federally listed as endangered on October 18, 1996. Critical habitat has not been designated for this species. Loss and fragmentation of habitat, and alteration of natural ecosystem processes have resulted from residential and commercial development in the Pine Hill intrusion area. Housing and commercial development, road maintenance, grading, change in fire frequency, unauthorized dumping, OHV use, overgrazing practices, herbicide spraying, mining, competition from invasive, non-native vegetation, and other human-caused conditions threaten the remaining occurrences of these plants. As Stebbin's morning-glory occurs within a fire-adapted plant community, changes in fire frequency have altered natural processes. Historically, fire occurred in chaparral on the average of 3 to 5 times every 100 years (Boyd 1985). Fire is important for seed germination and seedling reestablishment by eliminating competition and shading, as well as replenishing nutrients to the soil. Without periodic fires, chaparral species either do not reproduce by seed or may become shaded by other plants.

Pine Hill Ceanothus

The primary reference for this section is:
USFWS. 1996d. Determination of Endangered Status for Four Plants and Threatened Status for One Plant from the Central Sierran Foothills of California. Federal Register 61(203): 54346-54358.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office. Sacramento, California.

Pine Hill ceanothus (*Ceanothus roderickii*) is another species of the Pine Hill intrusion, where it is found in chaparral communities. This species is restricted to gabbro-derived soil in openings in chaparral or, more infrequently, on previously disturbed sites within chaparral (Wilson 1986). The species is restricted to one localized area of approximately 10 known extant occurrences discontinuously scattered in the Pine Hill intrusion (California Natural Diversity Database 1996). Pine Hill ceanothus occurs primarily on private land. The BLM administers part of one site and the California Department of Forestry administers another site. Residential and commercial development, OHV use, road-widening, changes in fire frequency, and other human-caused conditions are responsible for the decline of this species.

Pine Hill ceanothus is a prostrate evergreen shrub of the buckthorn family that generally grows to about 10 feet in diameter. The branches radiate from a central axis, and root when they come into contact with the ground. Small whitish flowers tinged with blue appear from May through June, and the fruit is a globe-shaped capsule.

Pine Hill ceanothus was federally listed as endangered on October 18, 1996. Critical habitat has not been designated. Loss and fragmentation of habitat, and alteration of natural ecosystem processes have resulted from residential and commercial development in the Pine Hill intrusion area. Housing and commercial development, road maintenance, grading, change in fire frequency, unauthorized dumping, OHV use, overgrazing practices, herbicide spraying, mining, competition from invasive, non-native vegetation, and other human-caused conditions threaten the remaining occurrences of these plants. Pine Hill ceanothus occurs within a fire-adapted plant community, where changes in fire frequency have altered natural processes. Historically, fire occurred in chaparral on the average of 3 to 5 times every 100 years (Boyd 1985). Fire is important for seed germination and seedling reestablishment by eliminating competition and shading, as well as replenishing nutrients to the soil. Without periodic fires, chaparral species either do not reproduce by seed or may become shaded by other plants.

Pine Hill Flannelbush

The primary reference for this section is:
USFWS. 1996d. Determination of Endangered Status for Four Plants and Threatened Status for One Plant from the Central Sierran Foothills of California. Federal Register 61(203): 54346-54358.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office. Sacramento, California.

Pine Hill flannelbush (*Fremontodendron californicum* ssp. *decumbens*) is another species of the Pine Hill intrusion, where it is found in the transition area between chaparral and oak woodland. The taxon occurs on scattered rocky outcrops either in chaparral or in the ecotone between woodland and chaparral. It is only known from one localized area near Pine Hill in western El Dorado County, scattered within an area of approximately 5,000 acres. It occurs primarily on private land, but one site is on BLM-administered land, and the California Department of Forestry and California Department of Fish and Game administers another site.

Pine Hill flannelbush is a branched spreading shrub that grows up to 4 feet tall. This subspecies blooms from late April to early July, bearing showy light-orange to reddish-brown flowers. Its fruit is a capsule. Seeds are dispersed by ants (Boyd 1996), and the plant depends on fire to promote seed germination.

Pine Hill flannelbush was federally listed as endangered on October 18, 1996. Critical habitat has not been designated. Loss and fragmentation of habitat, and alteration of natural ecosystem processes have resulted from residential and commercial development in the Pine Hill intrusion area. Housing and commercial development, road maintenance, grading, change in fire frequency, unauthorized dumping, OHV use, overgrazing practices, herbicide spraying, mining, competition from invasive, non-native vegetation, and other human-caused conditions threaten the remaining occurrences of these plants. Pine Hill flannelbush occurs within a fire-adapted plant community, where changes in fire frequency have altered natural processes. Historically, fire occurred in chaparral on the average of 3 to 5 times every 100 years (Boyd 1985). Fire is important for seed germination and seedling reestablishment by eliminating competition and shading, as well as replenishing nutrients to the soil. Without periodic fires, chaparral species either do not reproduce by seed or may become shaded by other plants.

El Dorado Bedstraw

The primary reference for this section is:
USFWS. 1996d. Determination of Endangered Status for Four Plants and Threatened Status for One Plant from the Central Sierran Foothills of California. Federal Register 61(203): 54346-54358.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

El Dorado bedstraw (*Galium californicum* ssp. *sierrae*) is another species of the Pine Hill intrusion, where it occurs in oak woodland habitat. The taxon is restricted to Pine Hill and the surrounding ridges to the west (Baad and Hanna 1987). El Dorado bedstraw is a perennial herb that flowers in May and June. It grows in oak woodland areas, including sites with ponderosa pine and gray pine (Wilson 1986). El Dorado bedstraw occurs primarily on privately-owned land, although the BLM administers the land where at least one population occurs, and the California Department of Forestry and California Department of Fish and Game administer one site as well.

El Dorado bedstraw was federally listed as endangered on October 18, 1996. Critical habitat has not been designated. Loss and fragmentation of habitat, and alteration of natural ecosystem processes have resulted from residential and commercial development in the Pine Hill intrusion area. Housing and commercial development, road maintenance, grading, change in fire frequency, unauthorized dumping, OHV use, overgrazing practices, herbicide spraying, mining, competition from invasive, non-native vegetation, and other human-caused conditions threaten the remaining occurrences of these plants. Oak woodlands are a fire-adapted plant community, where changes in fire frequency have altered natural processes. Without periodic fires, El Dorado bedstraw may become shaded by other plants.

Layne's Butterweed
The primary reference for this section is:
USFWS. 1996d. Determination of Endangered Status for Four Plants and Threatened Status for One Plant from the Central Sierran Foothills of California. Federal Register 61(203): 54346-54358.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

Layne's butterweed (*Senecio layneae*) is another species of the Pine Hill intrusion, where it occurs in chaparral and oak woodland habitats. The species also has a few known isolated occurrences in El Dorado, Nevada, and/or Tuolumne counties, California. Layne's butterweed grows in open rocky areas within chaparral plant communities, primarily on gabbro-derived soil formations and occasionally on serpentine soils. Most known sites are scattered within a 40,000-acre area in western El Dorado County that includes the Pine Hill intrusion and adjacent areas. A few other colonies occur in the Eldorado National Forest in El Dorado County and in the BLM Red Hills Management Area in Tuolumne County (BioSystems Analysis, Inc. 1989). One site is on land administered by the California Department of Forestry and California Department of Fish and Game, although the species primarily occurs on privately-owned land.

Layne's butterweed is a perennial herb of the aster family that sprouts from a rootstock. It flowers between April and June, each plant producing several orange-yellow flower heads 2 to 3 inches wide.

Layne's butterweed was federally listed as threatened on October 18, 1996. Critical habitat has not been designated for this species. Loss and fragmentation of habitat, and alteration of natural ecosystem processes have resulted from residential and commercial development in the Pine Hill intrusion area. Housing and commercial development, road maintenance, grading, change in fire frequency, unauthorized dumping, OHV use, overgrazing practices, herbicide spraying, mining, competition from invasive, non-native vegetation, and other human-caused conditions threaten the remaining occurrences of these plants. Layne's butterweed occurs within a fire-adapted plant community, where changes in fire frequency have altered natural processes. Fire is important for seed germination and seedling reestablishment by eliminating competition and shading, as well as replenishing nutrients to the soil.

Braunton's Milk-vetch
The primary reference for this section is:
USFWS. 1997b. Determination of Endangered Status for Two Plants and Threatened Status for Four Plants from Southern California. Federal Register 62 (19): 4172-4183.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Ventura Field Office. Ventura, California.

Braunton's milk-vetch (*Astragalus brauntonii*) is a robust, short-lived perennial in the pea family that occurs in the Los Angeles basin. This species is currently known from four general areas in Ventura, Los Angeles, and Orange counties. One population is found along the south slope of the Simi Hills of eastern Ventura and western Los Angeles counties. Two occurrences (one population) are known from Santa Ynez Canyon in the Santa Monica Mountains, Los Angeles County. Two occurrences (one population) are known from Coal and Gypsum canyons in the Santa Ana Mountains, Orange County (Natural Diversity Database 1994). Braunton's milk-vetch is associated with the fire-dependent chaparral habitat dominated by chamise, yucca, and the rare Tecate cypress. The species is considered a limestone endemic, and rarely occurs on non-limestone substrates.

Fire is a natural requirement for the survival of this species. The natural frequency of fire in the habitat of Braunton's milk-vetch is unknown, but estimates range from 20 to over 100 years, with an average of 70-year intervals (Minnich 1989, O'Leary 1990). Higher fire frequencies have resulted from increasing human populations in southern California, mostly in the form of arson-caused fires. This species has a life span of 2 to 3 years, and depending on fire interval, a given population appears only once in 20 to 50 or more years. Because reproduction of Braunton's milk-vetch is stimulated by fire events, the total number of individuals varies with current fire cycles.

Most of the habitat of Braunton's milk-vetch is on private land in areas with expanding development. Four public agencies, the California Department of Parks and Recreation, the Conejo Open Space Conservation Agency, the Rancho Simi Parks and Recreation District, and the National Park Service, have small colonies within their jurisdictions that may not be viable. All of the protected habitat occurs in the immediate vicinity of urban development.

Braunton's milk-vetch was federally listed as endangered on January 29, 1997. Critical habitat has not been designated. This species is threatened by direct loss from urban development, fragmentation of habitat and reduced capabilities for sustained ecological processes, fragmented ownership of single populations resulting in different landscape treatments, alteration of fire cycles, and extinction resulting from naturally occurring events due to small population sizes and low numbers of individuals (Mistretta 1992, Natural Diversity Database 1994).

Nevin's Barberry
The primary reference for this section is:
USFWS. 1998c. Endangered or Threatened Status for Three Plants from the Chaparral and Scrub of Southwestern California. Federal Register 63(197): 54956-54971.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Carlsbad Field Office. Carlsbad, California.

Nevin's barberry (*Berberis nevinii*) occurs in restricted, localized populations in the interior foothills of California. It is found in chaparral and alluvial scrub associated with rocky slopes and sediments and sandy washes in Los Angeles, Riverside and San Bernardino counties (Boyd 1987, Mistretta 1989).

Chaparral habitats of the interior foothill region of southern California are dense shrub associations of moderate height dominated by chamise, California lilac, redberry, manzanita, California scrub oak, sugar bush, laurel sumac, toyon, California buckwheat, and black sage (Holland 1986). Chaparral occurs on many different soil types, but Nevin's barberry typically occurs in clay soils derived from gabbro (mineral) or metavolcanic bedrock (Boyd 1991, Oberbauer 1991, California Natural Diversity Data Base 1997). Clay soils have unique physical and chemical properties that contribute to the disproportionately large number of rare plants found on this substrate, as compared to other soil types.

Alluvial scrub, found in certain floodplain systems in southern California, comprises an open vegetation community of drought-deciduous and evergreen shrubs (Smith 1980; Hanes et al. 1989). Alluvial scrub is characterized by porous, infertile soils subject to periodic intense flooding and erosion associated with the outwash environment (Hanes et al. 1988). This vegetation type includes life-forms of desert and coastal affinities such as California redberry, scalebroom, mountain mahogany, California buckwheat, and occasionally California juniper (Hanes et al. 1988). Urbanization and industrial development are eliminating this plant community (Smith 1980).

Nevin's barberry is a rhizomatous evergreen shrub ranging from 3 to 12 feet in height. It flowers from March through April, and then produces juicy, yellowish to red berries. Nevin's barberry is found in two habitat types: gravelly wash margins in alluvial scrub, and on coarse soils in chaparral (Niehaus 1977, Boyd 1987). The typical elevation range for this species is between 900 and 2,000 feet. The native range of Nevin's barberry currently extends from the foothills of the San Gabriel Mountains of Los Angeles County to near the foothills of the Peninsular Ranges of southwestern Riverside County. The population center for Nevin's barberry is located near Vail Lake in southwestern Riverside County. One of the two largest known populations of Nevin's barberry occurs in this area (Boyd 1987, California Natural Diversity Data Base 1997), and the other large population of Nevin's barberry is in San Francisquito Canyon on the Angeles National Forest in Los Angeles County (Boyd et al. 1989). The majority of Nevin's barberry plants found outside the Vail Lake and Angeles National Forest sites occur as isolated populations in San Bernardino and Los Angeles counties. In 1998, the total number of individuals was reportedly fewer than 1,000 (Boyd 1987), and possibly fewer than 500 (Metropolitan Water District 1991, California Natural Diversity Data Base 1997). The majority of occurrences of this species are on private lands in the Vail Lake region, although a few individuals occur on public lands north of Vail Lake and in the Cleveland National Forest southeast of Vail Lake (Boyd et al. 1989). In Los Angeles County, the species occurs on steep slopes in the Angeles National Forest (Boyd et al. 1989, California Natural Diversity Data Base 1997). Other populations are small and occur on private lands (Boyd 1987, California Natural Diversity Data Base 1997).

Nevin's barberry was federally listed as endangered on October 13, 1998. Critical habitat has not been designated for this species. Nevin's barberry is threatened by destruction, degradation and fragmentation of habitat by urbanization, encroachment by exotic plant species, and OHV use.

Mexican Flannelbush
The primary reference for this section is:
USFWS. 1998c. Endangered or Threatened Status for Three Plants from the Chaparral and Scrub of Southwestern California. Federal Register 63(197): 54956-54971.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Carlsbad Field Office, Carlsbad, California.

Mexican flannelbush (*Fremontodendron mexicanum*), like Nevin's barberry discussed above, occurs in restricted, localized populations in the interior foothills of California. Mexican flannelbush is known from chaparral and closed-cone coniferous forest dominated by Tecate cypress in San Diego County and northwestern Baja California, Mexico. Chaparral habitats of the interior foothill region of southern California are dense shrub associations of moderate height dominated by chamise, California lilac, redberry, manzanita, California scrub oak, sugar bush, laurel sumac, toyon, California buckwheat, and black sage (Holland 1986).

Mexican flannelbush is a small tree or shrub, 5 to 19 feet tall, with evergreen leaves and showy, orange to dark yellow flowers. Native populations of this species occur primarily in closed-cone coniferous forest and southern mixed chaparral, often in association with metavolcanic soils (Oberbauer 1991, Reiser 1996) at elevations between 900 and 3,000 feet. Reliable distribution records for the species indicate that it is currently only known from Cedar Canyon on Otay Mountain in southern San Diego County and at Arroyo Seco, north of San Quintin, Estado de Baja California, Mexico (Wiggins 1980). This species has not been observed during surveys of other historical localities (Ogden Environmental and Energy Services, Inc. 1992, Reiser 1996). The BLM administers most of the Cedar Canyon population. Other historical sites the USFWS considers to have potential for currently supporting or re-establishing populations of Mexican flannelbush are divided in ownership between the BLM and private landowners (California Natural Diversity Data Base 1997).

Mexican flannelbush was federally listed as endangered on October 13, 1998. Critical habitat has not been designated. The species is threatened by destruction, degradation and fragmentation of habitat by urbanization; encroachment by exotic plant species; disruption of normal fire cycles; and OHV use.

San Benito Evening-primrose

The San Benito evening-primrose (*Camissonia benitensis*) is found only on serpentine alluvial terraces in the San Benito Mountain/Clear Creek region of California (Raven 1969; Griffin 1977, 1978a, 1978b; Kiguchi 1983, 1984, 1985; USFWS 1985h, Florence and Kiguchi 1986). It grows in loose alluvial soil in openings in chaparral, under the sparse understory of the San Benito Forest, or in relatively barren deposits of alluvial gravel. Although not found in damp areas along streams, the species occasionally grows in dry soils immediately adjacent to streams (Kiguchi 1983, 1984, 1985; Florence and Kiguchi 1986). Its dependence on riparian influence seems to relate mainly to the deposition of alluvial soil and talus rather than on the aquatic habitat itself. The San Benito Forest is a unique combination of digger pine, Jeffrey pine, Coulter pine, and incense cedar (Griffin 1974). Along alluvial terraces, the forest tends to be sparse, blending in with serpentine chaparral. Throughout the serpentine area, it forms a complex mosaic with chaparral and barren talus slopes.

The San Benito evening-primrose has been found at elevations ranging from approximately 2,500 to 4,600 feet (Kiguchi 1983, 1984, 1985). It seems to prefer relatively flat terraces with slight to moderate slope. Plants grow in open areas, often with full sun exposure throughout the day.

The San Benito evening-primrose is an annual herb (Raven 1969), with a life cycle limited to the period from late winter/early spring (February through March) through early to mid summer (June through July; Kiguchi 1983, 1984, 1985). Flowering occurs from mid-April to early June, and fruiting occurs from late April to mid-June, with seed and fruit dispersal occurring from late May to July (Raven 1969; Griffin 1977, 1978a; Kiguchi 1983, 1984, 1985a; Florence and Kiguchi 1986). Flooding may be an agent in seed dissemination (Florence and Kiguchi 1986). In a large sense, availability of suitable habitat is a limiting factor for this species. However, the presence of potential habitat that does not support evening primrose populations (Kiguchi 1984, 1985) indicates that other factors such as seed dispersal or moisture requirements may also be involved.

The San Benito evening-primrose was federally listed as threatened on February 12, 1985. Critical habitat has not been designated. The San Benito region is mined for gravel, asbestos, and minerals. The Clear Creek Management Area is subject to seasonally heavy use by OHVs and associated impacts of camping (USDI BLM 1983, 1984, 1985a, 1986). The BLM has taken steps to protect the evening-primrose populations on public land, by installing chain fences or pipe barriers around all of the populations to prevent vehicle trespass (Florence and Kiguchi 1986). Future threats to the species are likely to be similar to existing threats, with the additional possibility of interspecific competition from other plants, such as introduced grasses in disturbed areas.

Morro Manzanita

The primary reference for this section is:
USFWS. 1994b. Endangered or Threatened Status for Five Plants and the Morro Shoulderband Snail from Western San Luis Obispo County, California. Federal Register 59 (240): 64613-64623.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Ventura Field Office, Ventura, California.

Morro manzanita (*Arctostaphylos morroensis*) occurs as components of several coastal plant communities in western San Luis Obispo County, California. The distribution of this plant has been tied to the presence of soils derived from ancient sand dunes. These soils, referred to as Baywood fine sands, were deposited during the Pleistocene epoch when sea levels 300 feet lower than current levels allowed large volumes of sand to blow inland into the Los Osos Valley. Morro manzanita is found in association with coastal dune scrub, maritime chaparral, and coast live oak woodland communities in sites with no or low to moderate slopes. On steeper slopes, particularly on the north-facing slopes of the Irish Hills, the species occurs in almost pure stands. Much of the area supporting the required habitat for Morro manzanita has been subject to urban development, and the species now covers an area of approximately 840 to 890 acres. Approximately 65% of the remaining habitat is within private ownership; the remaining 35% is on publicly owned lands within Montana de Oro State Park and two small preserves administered by California Department of Fish and Game.

Morro manzanita is a shrub of the heath family that reaches 5 to 13 feet in height. The seeds of this species require breaking, scratching, or softening of the seed coat to allow germination.

Morro manzanita was federally listed as threatened on December 15, 1994. Critical habitat has not been designated for this species. Morro manzanita occurs in communities that have undergone a number of changes resulting from both human-caused activities and natural occurrences. The rapid urbanization of the surrounding area has already eliminated the species in portions of its range. In addition, the configuration of Morro Bay itself has been altered by the construction of a breakwater and a marina, the deposition of sediments from the Los Osos Creek and Chorro Creek watersheds, and the dredging of waterways within the Bay (Gerdes et al. 1974). Further urban development and other activities such as recreation, grazing, and utility construction threaten the remaining occurrences of Morro manzanita.

Indian Knob Mountain Balm
The primary reference for this section is:
USFWS. 1994b. Endangered or Threatened Status for Five Plants and the Morro Shoulderband Snail from Western San Luis Obispo County, California. Federal Register 59 (240): 64613-64623.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Ventura Field Office, Ventura, California.

Indian Knob mountain balm (*Eriodictyon altissimum*), like Morro manzanita discussed above, occurs as a component of coastal plant communities in western San Luis Obispo County, California. This species occurs within coastal maritime chaparral and oak woodlands and co-occurs with Morro manzanita in several locations. Only six stands of Indian Knob mountain balm are known, ranging from the south end of Morro Bay to Indian Knob, between San Luis Obispo and Arroyo Grande.

Indian Knob mountain balm is a diffusely branched evergreen shrub that reaches a height of about 7 to 13 feet. This species produces small lavender flowers that are arranged in coiled clusters and produce numerous tiny seeds. It is a fire-adapted chaparral species, and produces new growth primarily from rhizomatous suckers.

Indian Knob mountain balm was federally listed as endangered on December 15, 1994. Critical habitat has not been designated. The species occurs in communities that have undergone a number of changes resulting from both human-caused activities and natural occurrences. The rapid urbanization of the surrounding area has already eliminated the Indian Knob mountain balm in a portion of its ranges. In addition, the configuration of Morro Bay itself has been altered by the construction of a breakwater and a marina, the deposition of sediments from the Los Osos Creek and Chorro Creek watersheds, and the dredging of waterways within the Bay (Gerdes et al. 1974). Further urban development and other activities such as recreation, grazing, and utility construction threaten the remaining occurrences of the Indian Knob mountain balm.

Orcutt's Spineflower
The primary reference for this section is:
USFWS. 1998d. Determination of Endangered or Threatened Status for Four Southern Maritime Chaparral Plant Taxa from Coastal Southern California and Northwestern Baja California, Mexico. Federal Register 61(195): 52370-52384.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Carlsbad Field Office, Carlsbad, California.

Orcutt's spineflower (*Chorizanthe orcuttiana*) occurs in southern maritime chaparral, a unique plant association that occurs only in coastal southern California along the immediate coast of San Diego and Orange counties and northwestern Baja California, Mexico. Southern maritime chaparral is a low, fairly open chaparral typically dominated by wart-stemmed ceanothus, mission manzanita, chamise, Nuttall's scrub oak, bush rue, red berry, Mojave yucca, and occasionally bush poppy (Holland 1986; Kehler-Wolf 1993; OGDEN 1993). The distribution of southern maritime chaparral in Orange County is disjunct, and the species composition is slightly different from that found in San Diego County and Mexico (Gray and Bramlet 1992). In 1996, there were an estimated 150 acres of this habitat type in Orange County (Todd Kehler-Wolf 1993) and between 1,500 and 3,700 acres in San Diego County (Oberbauer and Vanderwier 1991, OGDEN 1993, Hogan 1993). Much of the remaining southern maritime chaparral is located on Carmel Mountain, Torrey Pines State Park, and in the cities of Carlsbad and Encinitas in San Diego County.

Orcutt's spineflower is a low, yellow-flowered annual of the buckwheat family (Polygonaceae). It is primarily restricted to weathered sandstone bluffs in association with or in microhabitats within southern maritime chaparral. This species is endemic to south-central and southern coastal San Diego County, California. Historically, the species is known from 10 separate localities from Point Loma near San Diego (including the U.S. Naval Reservation), Del Mar, Kearney Mesa and Encinitas (California Department of Fish and Game 1992). However, plants have not been seen at most of these locations in recent years. The number of individuals in populations often varies widely from year to year because the success of germination is highly dependent on factors such as rainfall, which often differ substantially from one year to the next in southern California.

Orcutt's spineflower was federally listed as endangered on October 7, 1996. Critical habitat has not been designated for the species. The rapid urbanization of southern Orange County and south-central San Diego County has already eliminated a substantial portion of the southern maritime chaparral. In addition, the advent of widespread urbanization and the disruption in natural fire cycles potentially threatens the remaining southern maritime chaparral. Populations of Orcutt's spineflower have been subjected to a considerable degree of fragmentation, and are threatened by trampling by farm workers or recreational activities; fuel modification; competition from non-native plant species; and habitat destruction due to residential, agricultural, commercial, and recreational development.

Encinitis Baccharis

The primary reference for this section is:

USFWS. 1998d. Determination of Endangered or Threatened Status for Four Southern Maritime Chaparral Plant Taxa from Coastal Southern California and Northwestern Baja California, Mexico. Federal Register 61(195): 52370-52384.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Carlsbad Field Office, Carlsbad, California.

Encinitis baccharis (*Baccharis vannesae*), like Orcutt's spineflower discussed above, occurs in southern maritime chaparral of coastal southern California. Southern maritime chaparral is a low, fairly open chaparral typically dominated by wart-stemmed ceanothus, mission manzanita, chamise, Nuttall's scrub oak, bush rue, red berry, Mojave yucca, and occasionally bush poppy (Holland 1986; Kehler-Wolf 1993; OGDEN 1993). The distribution of southern maritime chaparral in Orange County is disjunct, and the species composition is slightly different from that found in San Diego County and Mexico (Gray and Bramlet 1992). In 1996, there were an estimated 150 acres of this habitat type in Orange County (Todd Kehler-Wolf 1993) and between 1,500 and 3,700 acres in San Diego County (Oberbauer and Vanderwier 1991, OGDEN 1993, Hogan 1993). Much of the remaining southern maritime chaparral is located on Carmel Mountain, Torrey Pines State Park, and in the cities of Carlsbad and Encinitas in San Diego County.

Encinitas baccharis is a broom-like shrub that grows to heights of about 2 to 4 feet. The species occurs in southern maritime chaparral in the vicinity of Encinitas, central San Diego County, California, and extends inland to Mount Woodson and Poway, California, where it is associated with dense southern mixed chaparral. There are scattered populations of this species from Encinitas east through the Del Dios highlands and Lake Hodges area to Mount Woodson and south to Poway and Carmel Mountain in San Diego County, California. The majority of the remaining populations of this species are on privately-owned lands.

Encinitis baccharis was federally listed as threatened, on October 7, 1996. Critical habitat has not been designated for the species. The rapid urbanization of southern Orange County and south-central San Diego County has already eliminated a substantial portion of the southern maritime chaparral. In addition, the advent of widespread urbanization and the disruption in natural fire cycles potentially threatens the remaining southern maritime chaparral. Populations of Encinitas baccharis have been subjected to a considerable degree of fragmentation, and are threatened by trampling by farm workers or recreational activities; fuel modification; competition from non-native plant species; and habitat destruction due to residential, agricultural, commercial, and recreational development.

Slender-horned Spineflower

The primary reference for this section is:

U.S. Department of Agriculture Forest Service. No date. Slender-horned Spineflower. Forest Plan Update for Los Padre National Forest, Angeles National Forest, San Bernardino National Forest, and Cleveland National Forest. Available at http://www.r5.fs.fed.us/sccs/species.

The slender-horned spineflower (*Dodecahema leptoceras*) occurs on sandy alluvial benches, and on floodplain terraces with alluvial scrub vegetation. Plants are also found on well-drained slopes in chaparral. Historically, the species occurred in many of the alluvial systems on the coastal side of the transverse range in Los Angeles and San Bernardino counties, California. At present, the species is known from nine occurrences ranging from Bee Canyon in the northeast, west to the Santa Ana River Wash in Redlands, and south to Temescal Canyon, Bautista Canyon, and the Vail Lake area of Riverside County, California.

The slender-horned spineflower is a diminutive annual herb, subject to wide annual variability as a function of amount and seasonality of rainfall, as well as seed set from previous years. The species flowers in April through June, but is most distinct in June and early July after the basal rosette and certain branches have turned a characteristic dark red color.

The slender-horned spineflower was federally listed as endangered on September 28, 1987. Critical habitat has not been designated. The primary threats to this species are loss of habitat through urbanization and flood control projects, and the associated hydrological and geomorphological changes to the alluvial systems that maintain the species' characteristic habitat type. Off-highway vehicle activity and the invasion of exotic species are also threats to some occurrences. Dispersed recreation can lead to trampling of plants. Upstream watershed management, and upstream prescribed fire, can alter downstream hydrology, with adverse (or beneficial) effects.

Santa Ana River Woolly-star

The primary references for this section are:

U.S. Department of Agriculture Forest Service. No date. Santa Ana River Woollystar. Forest Plan Update for Los Padre National Forest, Angeles National Forest, San Bernardino National Forest, and Cleveland National Forest. Available at http://www.r5.fs.fed.us/sccs/species.

and

California Department of Fish and Game. 2000a. The Status of Rare, Threatened, and Endangered Animals and Plants of California, Santa Ana River Woolly-star. California Department of Fish and Game Habitat Conservation Planning Branch. Sacramento, California. Available at http://www.dfg.ca.gov/hcpb/species/.

The Santa Ana River woolly-star (*Eriastrum densifolium* ssp. *sanctoumr*) occurs in the sandy soils of river floodplains or terraced alluvial deposits in the Santa Ana River drainage. The majority of its distribution is on relatively young (20 to 70 year-old) alluvial surfaces supporting early to intermediate phase alluvial scrub vegetation. Historically, the species was known to extend along 60 river miles in Orange, Riverside, and San Diego counties, but now plants occupy only about 18 linear miles of river floodplain along the Santa Ana River, City Creek, and Plunge Creek, California.

The Santa Ana River woolly-star is a perennial shrub species with an expected lifespan of 5 to 10 years. Plants flower from June through August and are most readily detectable during this time. An array of pollinators have been identified, including flies, sphinx moths, digger bees, and hummingbirds.

The Santa Ana River woolly-star was federally listed as endangered on September 28, 1987. Critical habitat has not been designated. The biggest threat to the continued existence of the species stems from the construction of the Seven Oaks dam, which will substantially reduce floodplain areas necessary to support the species. Without habitat-rejuvenating flooding events, open, sandy substrates will eventually become covered with vegetation, which would likely make these areas unsuitable for woolly-stars. Upstream watershed management and upstream prescribed fire can also alter downstream hydrology.

La Graciosa Thistle

The primary references for this section are:

USFWS. 2000b. Final Rule for Endangered Status for Four Plants from South Central Coastal California. Federal Register 65 (54): 14888-14898.

and

USFWS. 2001d. Proposed Designation of Critical Habitat for *Cirsium loncholepis* (La Graciosa Thistle), *Eriodictyon capitatum* (Lompoc yerba santa), and *Deinandra increscens* ssp. *villosa* (Gaviota Tarplant). Federal Register 66 (221): 57559-57600.

References cited in this section are internal to the above-referenced documents. A complete list of these references is available from the USFWS Ventura Field Office, Ventura, California.

La Graciosa thistle (*Cirsium loncholepis*) occurs in a narrow area along the south central California coast, in northern and western Santa Barbara County, southern San Luis Obispo County, and southern Monterey County. This species occurs in sensitive or altered dune habitats (Holland 1986, Schoenherr 1992). The largest coastal dune system in California, the Guadalupe Dune region, is located in southern San Luis Obispo County near Guadalupe, where approximately 8 square miles of active dunes create a series of back dune lakes. These coastal dune habitats are highly disturbed, and have been invaded by non-native plant species. Invasive weeds such as veldt grass, European beach grass, iceplant, and crystalline iceplant are serious threats to the natural ecological processes of coastal sandy habitats (Smith 1976, Zedler and Scheid 1988, Schoenherr 1992).

La Graciosa thistle is restricted to the back dune and coastal wetlands of the Guadalupe Dune complex, with the exception of a small disjunct population in southern Monterey County (California Natural Diversity Database 1998). The species is found in wet soils surrounding the dune lakes and in the moist dune swales, where it is often associated with rush, tule, willow, poison oak, salt grass, and coyote brush (Hendrickson 1990). As of 2000, there were 17 known locations for La Graciosa thistle, many of which were small and isolated, and showed a reduced reproductive vigor. All but one population of La Graciosa thistle, a small population in the Los Padres National Forest in southern Monterey County, occur on private lands. Observed declines in this species are apparently the result of changes in habitat as riparian willows and other vegetation invade the areas that previously supported this wet meadow plant (Chesnut 1998).

La Graciosa thistle is a short-lived member of the Aster family. Plants are from 4 to 40 inches in height, with one to several stems that bear clusters of whitish-purple flowering heads.

La Graciosa thistle was federally listed as endangered on March 20, 2000. On May 7, 2002, the USFWS designated approximately 44,000 acres of land in areas that support La Graciosa thistle as critical habitat. Ongoing threats to this species include groundwater pumping, oil field development, and competition from non-native plants (Hendrickson 1990, California Department of Fish and Game 1992). Cattle grazing in the riparian habitat at the mouth of the Santa Maria River may reduce the competition from other species (Hendrickson 1990), but the long-term effects of livestock use on the habitat are unknown.

Lompoc Yerba Santa

The primary references for this section are:
USFWS. 2000b. Final Rule for Endangered Status for Four Plants From South Central Coastal California. Federal Register 65 (54): 14888-14898.

and

USFWS. 2001d. Proposed Designation of Critical Habitat for *Cirsium loncholepis* (La Graciosa Thistle), *Eriodictyon capitatum* (Lompoc Yerba Santa), and *Deinandra increscens* ssp. *villosa* (Gaviota Tarplant). Federal Register 66 (221): 57559-57600.

References cited in this section are internal to the above-referenced documents. A complete list of these references is available from the USFWS Ventura Field Office, Ventura, California.

Lompoc yerba santa (*Eriodictyon capitatum*), like the La Graciosa thistle discussed in the previous section, occurs in a narrow area along the south central California coast, in sensitive or altered habitats (Holland 1986, Schoenherr 1992). Inland from the active dunes of the Guadalupe Dune region (discussed in the previous species account), a weakly cemented sandstone has weathered to produce a sandy, extremely well drained, and nearly infertile soil (Davis et al. 1988). The habitat that occurs on these sand hills has been called the central coast maritime chaparral (Ferren et al. 1984; Davis et al. 1988; Philbrick and Odion 1988; Davis et al. 1989; Odion et al. 1992). Seven local endemic plant species, including Lompoc yerba santa, and at least 16 other uncommon plant species, are components of this habitat. Central coast maritime chaparral is considered threatened and sensitive by the California Department of Fish and Game's Natural Heritage Division (Holland 1986).

The Lompoc yerba santa occurs in maritime chaparral with bush poppy, scrub oaks, and buck brush, and in southern Bishop pine forests that intergrade with chaparral (manzanita and black sage [Smith 1983]). The four known locations of this species occur in western Santa Barbara County. Two of these locations are on Vandenberg Air Force Base, and the other two are on private land in the oilfields south of Orcutt and at the western end of the Santa Ynez Mountains.

Lompoc yerba santa is a shrub in the waterleaf family with sticky stems up to 10 feet tall. Colonies of this species appear to be multiclonal, where the vegetative spread of the root system of a single plant produces many stems. Lompoc yerba santa is self-incompatible (i.e., it requires pollen from genetically different plants to produce seed), and its fruits appear to be parasitized by an insect (Elam 1994). Plants have been observed to resprout from the base after a prescribed fire, although living stems also die.

Lompoc yerba santa were federally listed as endangered on March 20, 2000. On May 7, 2002, the USFWS designated approximately 8,500 acres of land occupied by Lompoc yerba santa as critical habitat. Factors that threaten the species include fire management practices, invasive non-native plant species, low seed productivity, and naturally occurring catastrophic events.

Monterey Spineflower

The primary reference for this section is:
USFWS. 2001e. Proposed Designation of Critical Habitat for *Chorizanthe pungens* var. *pungens* (Monterey Spineflower). Federal Register 66 (32): 10440-10469.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Ventura Field Office, Ventura, California.

The Monterey spineflower (*Chorizanthe pungens* var. *pungens*) is endemic to sandy soils in coastal areas in southern Santa Cruz and northern Monterey counties, and in the Salinas Valley in interior Monterey County. It is found in a variety of seemingly disparate habitat types, including active coastal dunes, grassland, scrub, chaparral, and woodland types on interior upland sites; and interior floodplain dunes. However, all of these habitat types include microhabitat characteristics that are favored by the taxon. First, all sites are on sandy soils, which may originate from active dunes, interior fossil dunes, or floodplain alluvium. Second, these sites are relatively open and free of other vegetation. Within grassland communities, plants occur along roadsides, in firebreaks, and in other disturbed sites, while in oak woodland, chaparral, and scrub communities, they occur in sandy openings between shrubs. In grassland and oak woodland communities, abundant annual grasses may outcompete the Monterey spineflower, while management of grass species, either through grazing, mowing or fire, may allow the spineflower to persist. In scrub and chaparral communities, the taxon does not occur under dense oak or shrub stands, but will occur between more widely spaced trees and shrubs. Prior to onset of human use of this area, the Monterey spineflower may have been restricted to openings created by wildfires within these communities (USFWS 1998). In addition, at the former Fort Ord, the highest densities of plants are located in the central portion of the firing range, where disturbance is the most frequent.

The Monterey spineflower is generally distributed along the rim of Monterey Bay in southern Santa Cruz and northern Monterey counties, and inland along the coastal plain of the Salinas Valley. At coastal sites ranging from the Monterey Peninsula north to Manresa State Beach, it is found in active coastal dune systems, and on coastal bluffs upon which windblown sand has been deposited. On coastal dunes, the distribution of suitable habitat is subject to dynamic shifts caused by patterns of dune mobilization, stabilization, and successional trends in coastal dune scrub that increase in cover over time. Accordingly, individual colonies of the Monterey spineflower, found in gaps between stands of scrub, shift in distribution and size over time. Native plants associated with the taxon include beach bur, coastal sagewort, mock heather, Monterey Indian paintbrush, and beach pea. At some northern Monterey County locations, the Monterey spineflower occurs in close proximity to the endangered Monterey gilia, Menzies' wallflower, and in areas used by a threatened bird, the snowy plover.

The Monterey spineflower is a short-lived annual species. It germinates during the winter months and flowers from April through June. Although pollination ecology has not been studied for this taxon, it is likely visited by a wide array of pollinators. Each flower produces one seed. Depending on the vigor of an individual plant, dozens, if not hundred of seeds can be produced. The plants turn a rusty hue as they dry through the summer months, eventually shattering during the fall. Seed dispersal is facilitated by the involucral spines, which attach the seed to passing animals. While animal vectors most likely facilitate dispersal between colonies and populations, the prevailing coastal winds undoubtedly play a part in scattering seed within colonies and populations.

The Monterey spineflower was federally listed as threatened on February 4, 1994. On May 29, 2002, the USFWS designated approximately 18,830 acres of land (in Santa Cruz and Monterey counties) at four coastal sites and six inland sites where the taxon is known to occur. Portions of the coastal dune and coastal scrub communities that support the Monterey spineflower have been eliminated or altered by recreational use, industrial and urban development, and military activities. Dune communities have also been altered in composition by the introduction of non-native species, especially sea-fig or iceplant and European beachgrass, in an attempt to stabilize shifting sands. The species is threatened by residential development, agricultural land conversion, sand mining, military activities, and encroachment by non-native plant species.

Howell's Spineflower

The primary reference for this section is:
USFWS. 1992e. Six Plants and Myrtle's Silverspot Butterfly from Coastal Dunes in Northern and Central California Determined to be Endangered. Federal Register Volume 57: 27848-27859.

References cited in this section are internal to the above-referenced documents. They are included in the Bibliography.

Howell's spineflower (*Chorizanthe howellii*) is endemic to the coastal dunes of northern and central California. Within these dune systems, the species is restricted to the coastal foredunes and adjacent sandy habitats occupied by coastal prairie. The foredunes (also referred to as littoral dunes [Barbour and Johnson 1977] or coastal strand [Cooper 1919, Munz and Keck 1950]) are situated immediately above the lower, non-vegetated portion of the beach or littoral strip.

Howell's spineflower is a member of the buckwheat family that flowers from May through July. Restricted to coastal foredunes and adjacent sandy habitats occupied by coastal prairie, the species is discontinuously distributed within the southern portion of the dunes south of Tenmile River. This dune system stretches continuously for about 5 miles from the mouth of Tenmile River to Laguna Point, with isolated dunes as far south as Pudding Creek on the north edge of the community of Fort Bragg.

In the dune systems north of Monterey Bay, sand-stabilizing rhizomatous grasses (e.g., European beachgrass and American dunegrass) generally dominate the vegetation of the foredunes (Barbour and Johnson 1977). European beachgrass is an alien species that has largely replaced the native dunegrass-dominated foredune community. Aside from supplanting the native dunegrass-dominated community in the foredunes, the stabilization of the dunes by European beachgrass has permitted the colonization of formerly active backdune areas with a mixture of native and alien plants (Sauer 1988).

Aside from the beachgrass, many other alien plants have invaded these dune communities. Introduced taxa that are now established include sea-rocket, ice plant or sea-fig, and several annual grasses and forbs generally restricted to wetland habitats within the dunes (Barbour and Johnson 1977, Sauer 1988). In addition to the beachgrass, which has been used in dune stabilization projects along the Pacific Coast since 1869 (Cooper 1967), yellow bush lupine, a shrub native to the dunes of central and southern California, has been planted into the dune systems north of San Francisco Bay since 1900 (Miller 1987). In some cases, these non-native species have outcompeted and largely supplanted the native dune vegetation, including the four plants proposed herein.

Howell's spineflower was federally listed as endangered on June 22, 1992. Critical habitat has not been designated. Aside from the impact of exotic vegetation, many of the areas harboring populations of the species are threatened by proposed commercial and residential development. The historical use of some dune systems by the military has resulted in heavy damage to these systems (Cooper 1967). Off-highway vehicle use has also damaged the fragile plant communities in these dune systems and remains an important threat to rare dune plants on both public and privately-owned lands. Trampling of the native flora by equestrians, hikers (Brown 1987), and perhaps livestock (Clark and Fellers 1986) also threatens plants in the community. Other factors adversely affecting coastal dunes species include sand mining, disposal of dredged material from adjacent bays and waterways, and perhaps stochastic extinction by virtue of the small isolated nature of the remaining populations.

Menzies' Wallflower

The primary reference for this section is:

USFWS. 1992e. Six Plants and Myrtle's Silverspot Butterfly From Coastal Dunes in Northern and Central California Determined to be Endangered. Federal Register Volume 57: 27848-27859.

References cited in this section are internal to the above-referenced documents. They are included in the Bibliography.

Menzies' wallflower (*Erysimum menziesii*) is another species that is endemic to the coastal dunes of northern and central California. The species is discontinuously distributed within the coastal foredune community of four dune systems. The northernmost dune system, known as the Humboldt Bay dune system, stretches from the mouth of the Little River to Centerville Beach south of the Eel River in Humboldt County. Within these dunes, the species is restricted to a 12-mile stretch between the mouths of the Mad River and Humboldt Bay (i.e., Samoa Peninsula). This species also occurs within the Tenmile River dune system in Mendocino County and the Monterey Bay dune system, which ranges from La Selva (north of the mouth of the Pajaro River) to the City of Monterey in Monterey County. Within the Monterey Bay dune system, the species does not occur north of the mouth of the Salinas River. Several small discontinuous populations occur within this 13-mile reach.

Menzies' wallflower is a low, succulent, rosette-forming, biennial to short-lived perennial herb. Throughout most of its range, the species produces dense clusters of bright yellow flowers in the winter and early spring (January to April). However, the populations near Marina in Monterey County flower in early summer (May to June).

Menzies' wallflower was federally listed as endangered on June 22, 1992. Critical habitat has not been designated. As discussed in the previous species account (Howell's spineflower) many non-native plants have invaded these dune communities. In some cases, these non-native species have outcompeted and largely supplanted the native dune vegetation, including Menzies' wallflower. Aside from the impact of exotic vegetation, many of the areas harboring populations of the species are threatened by proposed commercial and residential development. The historical use of some dune systems by the military has resulted in heavy damage to these systems (Cooper 1967). Other OHV use has also damaged the fragile plant communities in these dune systems and remains a major threat to rare dune plants on both public and privately-owned lands. Trampling of the native flora by equestrians, hikers (Brown 1987), and perhaps livestock (Clark and Fellers 1986) also threatens plants in the community. Other factors adversely affecting coastal dunes species include sand mining, disposal of dredged material from adjacent bays and waterways, and perhaps stochastic extinction by virtue of the small isolated nature of the remaining populations.

Monterey Gilia
The primary reference for this section is:
USFWS. 1992e. Six Plants and Myrtle's Silverspot Butterfly From Coastal Dunes in Northern and Central California Determined to be Endangered. Federal Register Volume 57: 27848-27859.

References cited in this section are internal to the above-referenced documents. They are included in the Bibliography.

Monterey gilia (*Gilia tenuiflora ssp. arenaria*), and erect, short, rosette-forming, annual herb, is another species endemic to the coastal dunes of northern and central California. This species is restricted to isolated occurrences within wind-sheltered, sparsely vegetated portions of the Monterey Bay and Monterey Peninsula dune systems in Monterey County. The subspecies typically grows within coastal dune scrub or Flandrian dune habitat (Pavlik et al. 1987). The Monterey Peninsula populations range from Point Pinos to Point Joe.

Coastal dune scrub, characterized as a soft, woody, dense plant community of short shrubs and herbaceous plants, occurs in generally stabilized backdune areas. The following plant species are associated with coastal dune scrub: beach wormwood, coyote brush, California goldenbush, yellow bush lupine, chamisso bush lupine, and California figwort.

Monterey gilia was federally listed as endangered on June 22, 1992. Critical habitat has not been designated. As discussed in the species account for Howell's spineflower, many non-native plants have invaded these dune communities. In some cases, these non-native species have outcompeted and largely supplanted the native dune vegetation, including Monterey gilia. Aside from the impact of exotic vegetation, many of the areas harboring populations of the four plants are threatened by proposed commercial and residential development. The historical use of some dune systems by the military has resulted in heavy damage to these systems (Cooper 1967). Other OHV use has also damaged the fragile plant communities in these dune systems and remains a major threat to rare dune plants on both public and privately-owned lands. Trampling of the native flora by equestrians, hikers (Brown 1987), and perhaps livestock (Clark and Fellers 1986) also threatens plants in the community. Other factors adversely affecting coastal dunes species include sand mining, disposal of dredged material from adjacent bays and waterways, and perhaps stochastic extinction by virtue of the small isolated nature of the remaining populations.

Beach Layia
The primary reference for this section is:
USFWS. 1992e. Six Plants and Myrtle's Silverspot Butterfly from Coastal Dunes in Northern and Central California Determined to be Endangered. Federal Register Volume 57: 27848-27859.

References cited in this section are internal to the above-referenced documents. They are included in the Bibliography.

Beach layia (*Layia carnosa*), a low, glandular winter annual, is another species that is endemic to the coastal dunes of northern and central California. The northernmost occurrences of beach layia are from the Humboldt Bay dune system in Humboldt County. Historically, these populations ranged from near the mouth of the Little River and along the Samoa Peninsula. However, exotic vegetation and highway construction have reportedly eliminated beach layia and the rest of the native plant community from the Little River area. Beach layia occurs in two isolated dune systems: near the mouth of McNutt Gulch and south of the mouth of the Mattole River in Humboldt County. The species has also been collected from near Kehoe Beach and Abbotts Lagoon in the Point Reyes dune system. Within the Monterey Peninsula dune system, two of the four known occurrences have been eliminated. Although suitable habitat remains, the southernmost previously known location of beach layia from near Surf in Santa Barbara County has not been seen since 1929.

Beach layia was federally listed as endangered on June 22, 1992. Critical habitat has not been designated. As discussed in the species account for Howell's spineflower, many non-native plants have invaded these dune communities. In some cases, these non-native species have outcompeted and largely supplanted the native dune vegetation, including beach layia. Aside from the impact of exotic vegetation, many of the areas harboring populations of the four plants are threatened by proposed commercial and residential development. The historical use of some dune systems by the military has resulted in heavy damage to these systems (Cooper 1967). Other OHV use has also damaged the fragile plant communities in these dune systems and remains an important threat to rare dune plants on both public and privately-owned lands. Trampling of the native flora by equestrians, hikers (Brown 1987), and perhaps livestock (Clark and Fellers 1986) also threatens plants in the community. Other factors adversely affecting coastal dunes species include sand mining, disposal of dredged material from adjacent bays and waterways, and perhaps stochastic extinction by virtue of the small isolated nature of the remaining populations.

Western Lily
The primary reference for this section is:
USFWS. 1998e. Recovery Plan for the Endangered Western Lily (*Lilium occidentale*). Portland, Oregon.

The western lily (*Lilium occidentale*) occurs in early successional bogs or coastal scrub on poorly drained soils, usually those underlain by a hard, poorly permeable layer. Currently, the species occurs in widely scattered locations near the Pacific Ocean. Populations occur along a 200-mile stretch of the Pacific Coast, from near Coos Bay in Oregon, south to Humboldt Bay in California. The plants grow at low elevations, from almost sea level to about 300 feet, and from ocean-facing bluffs to about 4 miles inland. Common plant associates include the shrubs salal, western wax myrtle, western spiraea, huckleberry, blackberry, black twinberry, and glandular Labrador tea. Common tree associates include shore pine, Sitka spruce, red alder, Port Orford cedar, and willow. Common herbaceous associates include Pacific reed-grass, slough sedge, bunchberry, staff gentian, bracken fern, peat moss, and western tofieldia.

The western lily appears to require a habitat that maintains a delicate balance between having some shrubbery and having too much. Vegetation less than 3 feet tall can be beneficial to the lily by sheltering juvenile plants from browsing by large mammals, and by providing shelter from the heat in July and August. This protection is most critical during spring and early summer, because seedlings appear to tolerate dieback of aboveground parts later in the growing season. Dense, tall shrub growth reduces reproduction and survivorship, and closure of the forest canopy will eventually eliminate a population entirely.

The western lily is an herbaceous perennial that grows from an unbranched, scaly, bulblike rhizome. The species reproduces primarily by seed, but asexual reproduction is possible from detached bulb scales growing into new plants. Shoots emerge primarily in March and April, although they can emerge as early as January in some locations. Flowers bloom in May to July. Rhizomes may produce one or more flowering shoots per year, each typically with one to three, but up to 25, pendant flowers. Flowers often emerge above the surrounding shrubs, where they are available to pollinators such as hummingbirds. Capsular fruits become erect and may produce over 100 seeds when mature. Seeds are dispersed primarily by wind and gravity, generally within a radius of about 13 feet. Each year the aboveground portion of the plants die back and individuals overwinter underground as rhizomes/bulbs.

The western lily was federally listed as endangered on August 17, 1994. Critical habitat has not been designated. The species is known or assumed to be extirpated in at least nine historical sites, as a result of forest succession, cranberry farm development, livestock grazing, deer and mammal herbivory, highway construction, and other development. These factors continue to threaten the western lily, with development taking a primary role. Populations of the western lily appear to have been maintained in the past by occasional fires, at least at some sites in Oregon, and by grazing. Among the most serious threats to this species is loss of habitat as a result of ecological succession facilitated by aggressive fire suppression.

San Diego Ambrosia

The primary reference for this section is:

USFWS. 1999f. Proposed Endangered Status for *Ambrosia pumila* (San Diego Ambrosia) from Southern California. Federal Register 64(249): 72993-73003.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Carlsbad Field Office. Carlsbad, California.

San Diego ambrosia (*Ambrosia pumila*) is found on upper terraces of rivers and drainages as well as in open grasslands, openings in coastal sage scrub habitat, and dry lake beds. The species may also be found in disturbed sites such as fuel breaks and roadways. Associated native plant taxa include saltgrass, California Orcutt grass, mule-fat, and turkey-mullein. Populations of San Diego ambrosia occur on federal, state, and local government lands, and on private lands in western San Diego County, western Riverside County, and in the northern state of Baja California, Estado de Baja California, Mexico.

San Diego ambrosia is an herbaceous perennial that arises from a branched system of rhizome-like roots. This rhizomatous perennial habit results in groupings of aerial stems, often termed clones, that are, or at least were at one time, all attached to one another. The aerial stems sprout in early spring after the winter rains and deteriorate in late summer. Therefore, the plant may not be evident from late summer to early spring. This species is monoecious, with separate male and female flowers on the same plant, and is wind-pollinated. The male flower clusters are borne at the end of stalks, and the female flower clusters are in the axils of the leaves below the male flower clusters. The fruiting heads are enclosed by cup-like structures. This species flowers from May through October. Because this species is a clonal plant, the numbers of genetically different individuals in an occurrence, especially small occurrences could be very low. It is possible that an occurrence that supports even 1,000 aerial stems may consist of very few plants. This suggests that the low genetic diversity within the smaller occurrences may relegate these occurrences to extinction (Barrett and Kohn 1991). The majority of the occurrences of this species are in San Diego County, with the remainder in western Riverside County.

In San Diego County, two occurrences are protected on the Sweetwater River watershed in the recently established San Diego National Wildlife Refuge. Other occurrences in the Sweetwater River watershed are in vacant lots in El Cajon. There are three occurrences in the San Diego River watershed, the largest of which is in Mission Trails Regional Park administered by the City of San Diego, and on adjacent private land. The adjacent private lands portion of this occurrence is afforded protections under the City of San Diego’s Subarea Plan of the Multiple Species Conservation Program (City of San Diego 1997). There are also small occurrences on the San Luis Rey River watershed near Bonsall. The remaining extant occurrence in San Diego County is at a privately-owned site on the San Dieguito River watershed. The area is degraded and immediately adjacent to a bulldozed area of a development (Wallace 1999).

Two occurrences are known from Riverside County, on privately-owned lands. One is located along Nichols Road, Lake Elsinore, and the other is located at a fenced mitigation area at Skunk Hollow (McMillan 1999).

San Diego ambrosia was proposed for listing as an endangered species on December 29, 1999. The USFWS has determined that future designation of critical habitat for this species is prudent. This species is threatened by the destruction, fragmentation, and degradation of habitat by recreational and commercial development; highway construction and maintenance; construction and maintenance activities associated with a utility easement; competition from non-native plants; trampling by horses and humans; and OHV use.

San Diego Thornmint
The primary reference for this section is:
USFWS. 1998f. Determination of Endangered or Threatened Status for Four Plants from Southwestern California and Baja California, Mexico. Federal Register 63(197): 54937-54956.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Carlsbad Field Office, Carlsbad, California.

San Diego thornmint (*Acanthomintha ilicifolia*) is an annual aromatic herb of the mint family that usually occurs on heavy clay soils in openings within coastal sage scrub, chaparral and native grassland of coastal San Diego County, and in isolated populations south to San Telmo in northern Baja California, Mexico (Beauchamp 1986; Reiser 1996; USFWS, unpublished data). The species is frequently associated with gabbro soils which are derived from igneous rock, and also occurs in calcareous marine sediments. At the time of its listing in 1998, there were 32 known populations of the San Diego thornmint in the United States, ranging from San Marcos east to Alpine and south to Otay Mesa in San Diego County (Reiser 1996, California Native Natural Diversity Data Base 1997, Roberts 1997a), and covering an estimated 400 acres. Four major populations of this species are located within the Multiple Species Conservation Program planning subregion of southern San Diego County, California. The remaining major populations are located either north or east of the Multiple Species Conservation Program subregion, either on lands administered by the Forest Service (on Viejas and Poser mountains), or on privately-owned lands (California Native Natural Diversity Data Base 1997, Roberts 1997a).

San Diego thornmint was federally-listed as threatened on October 13, 1998. Critical habitat has not been designated. Most of the population increases in Southern California have occurred within or near sites historically occupied, in part, by coastal sage scrub. About 220,000 acres of coastal sage scrub remain in San Diego County (USFWS 1996). Habitat destruction or modification adversely affects species native to this area by reducing population densities and contributing to habitat fragmentation. Rapid urbanization and agricultural conversion in Orange and San Diego counties has already affected populations of the San Diego thornmint, and habitat loss and fragmentation is expected to continue as the population expands. The species is also adversely affected by the invasion of non-native plants, OHV use, increased erosion, grazing, and trampling by humans.

Otay Tarplant
The primary reference for this section is:
USFWS. 1998f. Determination of Endangered or Threatened Status for Four Plants from Southwestern California and Baja California, Mexico. Federal Register 63(197): 54937-54956.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Carlsbad Field Office, Carlsbad, California.

The Otay tarplant (*Deinandra conjugens*) is a glandular, aromatic annual in the aster family with a branching stem from 2.0 to 9.8 inches in height and yellow flower heads. The Otay tarplant currently has a limited distribution near Otay Mesa in southern San Diego County, California, and there is also one known population near the United States border in Baja California, Mexico (Morey 1994, California Department of Fish and Game 1994, Reiser 1996, California Native Natural Diversity Data Base 1997, Roberts 1997b). The distribution of this species is highly correlated with the distribution of clay soils or clay subsoils (Morey 1994), and plants are typically found growing in clay soils on slopes and mesas within native and mixed (native and non-native) grassland or open coastal sage scrub habitats. Clay soils offering suitable habitat for the Otay tarplant have been much reduced in acreage, primarily by urbanization and cultivation. The five largest populations of Otay tarplant are Horseshoe Bend-Gobblers Knob (Rancho San Miguel), Rice Canyon, Poggi Canyon, Proctor Valley, and Dennery Canyon (OGDEN 1992a, Morey 1994, Stone 1994, San Diego Gas and Electric 1995, City of San Diego and USFWS 1996b, Roberts 1997b). All populations of this species in the U.S. are on private lands. The Otay tarplant appears to tolerate mild levels of disturbance such as light grazing, which create sites necessary for germination (Tanowitz 1977, Hogan 1990).

Most of the population increases in Southern California have occurred within or near sites historically occupied, in part, by coastal sage scrub. About 220,000 acres of coastal sage scrub remain in San Diego County (USFWS 1996). Habitat destruction or modification adversely affects species native to this area by reducing population densities and contributing to habitat fragmentation.

The Otay tarplant was federally-listed as threatened on October 13, 1998. On December 10, 2002, USFWS designated approximately 2,560 acres in San Diego, California as critical habitat. Most of the population increases in Southern California have occurred within or near sites historically occupied, in part, by coastal sage scrub. Rapid urbanization and agricultural conversion in Orange and San Diego counties has already affected populations of the San Diego thornmint, and habitat loss and fragmentation is expected to continue as the population expands. The species is also adversely affected by the invasion of non-native plants, OHV use, increased erosion, grazing, and trampling by humans.

Otay Mesa-mint
The primary reference for this section is:
USFWS. 1993f. Determination of Endangered Status for Three Vernal Pool Plants and the Riverside Fairy Shrimp. Federal Register 58 (147): 41384-41392.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Carlsbad Field Office, Carlsbad, California.

Otay mesa-mint (*Pogogyne nudiuscula*) occurs in vernal pools from southwestern Riverside County and western San Diego County, California, to northwestern Baja California, Mexico. Vernal pools are specific habitats that form in areas with Mediterranean climates, where slight depressions become seasonally wet or inundated following fall and winter rains. The presence of an impervious layer such as hardpan, clay, or basalt beneath the soil surface causes water to remain in these pools for a few months at a time. In the spring, gradual drying occurs (Holland 1976). The pools form on mesa tops or valley floors and are interspersed among very low hills (Zedler 1987).

Otay mesa-mint is an erect annual that typically blooms from May through June (Munz 1974). A member of the mint family, this plant is aromatic, with bright purple flowers occurring on spikes. The current known distribution of this species is restricted to some of the remaining vernal pools on Otay Mesa.

Otay mesa-mint were federally listed as endangered on August 3, 1993. Critical habitat has not been designated. Agricultural development is widespread and increasing in areas where vernal pool habitat is typically found (Moran 1981). Habitat loss and degradation due to urban and agricultural development, livestock grazing, OHV use, trampling, invasion from weedy non-native plants, and other factors threaten the continued existence of the Otay mesa-mint.

California Orcutt Grass
The primary reference for this section is:
USFWS. 1993f. Determination of Endangered Status for Three Vernal Pool Plants and the Riverside Fairy Shrimp. Federal Register 58 (147): 41384-41392.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Carlsbad Field Office, Carlsbad, California.

California orcutt grass (*Orcuttia californica*), like the Otay mesa-mint discussed in the previous species account, occurs in vernal pools in southern California and Mexico. California orcutt grass occurs in vernal pools on The Nature Conservancy's Santa Rosa Plateau Preserve, in a vernal pool within Salt Creek drainage near Hemet (D. Bramlet 1992), and in the Skunk Hollow pool in Riverside County (Lathrop 1976). In San Diego County, this species is present in pools on Otay Mesa (Bauder 1986). One population of California orcutt grass is present in a vernal pool in Woodland Hills of Ventura County, California.

California orcutt grass is a member of the grass family that is associated with deeper pools of water than Otay mesa-mint. This small annual grass reaches 4 inches in height, is bright green, and secretes sticky droplets that taste bitter. Flowering structures, borne from May through June, are arranged in two rows.

California orcutt grass was federally listed as endangered on August 3, 1993. Critical habitat has not been designated. Agricultural development is widespread and increasing in areas where vernal pool habitat is typically found (Moran 1981). Habitat loss and degradation due to urban and agricultural development, livestock grazing, OHV use, trampling, invasion from weedy non-native plants, and other factors threaten the continued existence of California orcutt grass.

Hairy Orcutt Grass
The primary reference for this section is:
USFWS. 1997c. Determination of Endangered Status for Three Plants and Threatened Status for Five Plants from Vernal Pools in the Central Valley of California. Federal Register 62 (58): 14338-14352.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

Hairy orcutt grass (*Orcuttia pilosa*) is found in the basins and margins of vernal pools of the Central Valley of California. Similar to the vernal pools found in other geographic areas, these pools are typically small, seasonally aquatic ecosystems that are inundated with water in the winter and dry slowly in the spring and summer, creating a harsh, unique environment. Within the Central Valley (a geographic area that consists of the Sacramento Valley in the northern half of the state and the San Joaquin Valley in the southern half), vernal pools are found in four physiographic settings, each possessing an impervious soil layer relatively close to the surface: high terraces with iron-silicate or volcanic substrates, old alluvial terraces, basin rims with claypan soils, and low valley terraces with silica-carbonate claypans. Due to local topography and various geological populations, vernal pools are usually clustered into pool complexes. The vernal pool habitats and the threatened and endangered species found therein occur over a very limited, discontinuous, fragmented area within the Central Valley.

Hairy orcutt grass was federally listed as endangered on March 26, 1997. On August 6, 2003, the USFWS designated approximately 1,184,513 acres of vernal pool habitat as critical habitat for this and other vernal pool species. Factors that imperil the continued existence of Central Valley vernal pool species include habitat loss and degradation as a result of urbanization, agricultural land conversion, livestock grazing, OHV use, a flood control project, a highway project, altered hydrology, landfill projects, and competition from weedy, non-native plants.

Greene’s Tuctoria
The primary reference for this section is:
USFWS. 1997c. Determination of Endangered Status for Three Plants and Threatened Status for Five Plants from Vernal Pools in the Central Valley of California. Federal Register 62 (58): 14338-14352.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

Greene’s tuctoria (*Tuctoria greenei*), like hairy orcutt grass discussed in the previous species account, is found in the basins and margins of vernal pools of the Central Valley of California. Greene’s tuctoria is a tufted, annual grass that grows 2 to 6 inches tall. The present range of this species covers 258 miles, with populations in Butte, Glenn, Merced, Shasta, and Tehama counties. With the exception of one small population of 50 plants on the Sacramento National Wildlife Refuge, all populations are on privately-owned lands, including four on The Nature Conservancy’s Vina Plains Preserve.

Greene's tuctoria was federally listed as endangered on March 26, 1997. On August 6, 2003, the USFWS designated approximately 1,184,513 acres of vernal pool habitat as critical habitat for this and other vernal pool species. Factors that imperil the continued existence of Central Valley vernal pool species include habitat loss and degradation as a result of urbanization, agricultural land conversion, livestock grazing, OHV use, a flood control project, a highway project, altered hydrology, landfill projects, and competition from weedy, non-native plants.

Fleshy Owl's-clover

The primary reference for this section is:
USFWS. 1997c. Determination of Endangered Status for Three Plants and Threatened Status for Five Plants from Vernal Pools in the Central Valley of California. Federal Register 62 (58): 14338-14352.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

Fleshy owl's-clover (*Castilleja campestris* ssp. *succulenta*), like the species discussed in the previous two species accounts, is found in the basins and margins of vernal pools of the Central Valley of California. This taxon, which was formerly more widespread in the Central Valley, discontinuously occurs in the San Joaquin Valley over a range of 66 miles, extending through northern Fresno, western Madera, eastern Merced, southeastern San Joaquin, and Stanislaus counties. One population occurs on lands administered by the Bureau of Reclamation, one on lands administered by the California Department of Transportation, and two populations on land administered by the BLM. The remainder (and majority) of the populations occur on privately-owned lands, and some occur on land where The Nature Conservancy has a conservation easement (California Natural Diversity Database 1996).

Fleshy owl's-clover is a partly parasitic, annual herb with stems that are generally 2 to 10 inches tall. It produces bright yellow to white flowers in May.

Fleshy owl's-clover was federally listed as threatened on March 26, 1997. On August 6, 2003, the USFWS designated approximately 1,184,513 acres of vernal pool habitat as critical habitat for this and other vernal pool species. Factors that imperil the continued existence of Central Valley vernal pool species include habitat loss and degradation as a result of urbanization, agricultural land conversion, livestock grazing, OHV use, a flood control project, a highway project, altered hydrology, landfill projects, and competition from weedy, non-native plants.

Hoover's Spurge

The primary reference for this section is:
USFWS. 1997c. Determination of Endangered Status for Three Plants and Threatened Status for Five Plants from Vernal Pools in the Central Valley of California. Federal Register 62 (58): 14338-14352.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

Hoover's spurge (*Chamaesyce hooveri*), like the species discussed in the previous three species accounts, is found in the basins and margins of vernal pools of the Central Valley of California. It is a prostrate, annual herb that is found in vernal pools on remnant alluvial fans and related depositional stream terraces along a stretch of 240 miles on the eastern margin of the Central Valley. Extant populations occur in Butte, Glenn, Merced, Stanislaus, Tehama, and Tulare counties. All populations are on privately-owned lands, except for the four populations in Glenn County found on the Sacramento National Wildlife Refuge (J. Silveira, Sacramento National Wildlife Refuge, 1994; California Natural Diversity Database 1996).

Hoover's spurge was federally listed as threatened on March 26, 1997. On August 6, 2003, the USFWS designated approximately 1,184,513 acres of vernal pool habitat as critical habitat for this and other vernal pool species. Factors that imperil the continued existence of Central Valley vernal pool species include habitat loss and degradation as a result of urbanization, agricultural land conversion, livestock grazing, OHV use, a flood control project, a highway project, altered hydrology, landfill projects, and competition from weedy, non-native plants.

San Joaquin Valley Orcutt Grass
The primary reference for this section is:
USFWS. 1997c. Determination of Endangered Status for Three Plants and Threatened Status for Five Plants from Vernal Pools in the Central Valley of California. Federal Register 62 (58): 14338-14352.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

San Joaquin Valley orcutt grass (*Orcuttia inaequalis*), like the species discussed in the previous four species accounts, is found in the basins and margins of vernal pools of the Central Valley of California. San Joaquin Valley orcutt grass is a tufted annual that reaches 2 to 6 inches in height. Most of the remaining populations of this species are discontinuously scattered over a 36-mile area in southeastern San Joaquin Valley, in Fresno, Merced, and Madera counties. Two populations are on federal land, one on land administered by the BLM and one transplanted population by the Bureau of Reclamation. The remaining populations are found on privately-owned lands.

San Joaquin Valley orcutt grass was federally listed as threatened on March 26, 1997. On August 6, 2003, the USFWS designated approximately 1,184,513 acres of vernal pool habitat as critical habitat for this and other vernal pool species. Factors that imperil the continued existence of Central Valley vernal pool species include habitat loss and degradation as a result of urbanization, agricultural land conversion, livestock grazing, OHV use, a flood control project, a highway project, altered hydrology, landfill projects, and competition from weedy, non-native plants.

Slender Orcutt Grass
The primary reference for this section is:
USFWS. 1997c. Determination of Endangered Status for Three Plants and Threatened Status for Five Plants from Vernal Pools in the Central Valley of California. Federal Register 62 (58): 14338-14352.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

Slender orcutt grass (*Orcuttia tenuis*), like the species discussed in the previous five species accounts, is found in the basins and margins of vernal pools of the Central Valley of California. Slender orcutt grass is a weakly-tufted annual grass that grows to about 2 to 6 inches in height, producing one to several erect stems that often branch from the upper nodes. Disjunct populations of this species occur in vernal pools on remnant alluvial fans and high stream terraces and recent basalt flows across 220 miles (Stone et al. 1988). Slender orcutt grass is restricted to northern California, with populations in Lake, Lassen, Plumas, Sacramento, Shasta, Siskiyou, and Tehama counties. The majority of these populations are on privately-owned lands. The City of Redding owns lands containing two populations, and the remaining populations are found on land administered by the Forest Service and the BLM.

Slender orcutt grass was federally listed as threatened on March 26, 1997. On August 6, 2003, the USFWS designated approximately 1,184,513 acres of vernal pool habitat as critical habitat for this and other vernal pool species. Factors that imperil the continued existence of Central Valley vernal pool species include habitat loss and degradation as a result of urbanization, agricultural land conversion, livestock grazing, OHV use, a flood control project, a highway project, altered hydrology, landfill projects, and competition from weedy, non-native plants.

Contra Costa Goldfields
The primary reference for this section is:
USFWS. 1997d. Endangered Status for Four Plants from Vernal Pools and Mesic Areas in Northern California. Federal Register 62 (117): 33029-33038.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

Contra Costa goldfields (*Lasthenia conjugens*) grows in and around the margins of vernal pools and in seasonally wet areas in northern California. These vernal pools are typically found in open grassy areas of woodland and valley grassland communities. Contra Costa goldfields typically occurs at elevations up to 700 feet. It is currently known from a total of 13 populations in Alameda, Contra Costa, Napa, and Solano counties (California Native Plant Society 1978, California Natural Diversity Database 1996). The population located at Travis Air Force Base is the only population on federal land; all other populations are on privately-owned lands.

Contra Costa goldfields is a showy spring annual in the aster family that grows 4 to 12 inches tall and is usually branched. This species flowers from March to June, producing yellow flowers in terminal heads.

Contra Costa goldfields was federally listed as endangered on June 18, 1997. On August 6, 2003, the USFWS designated approximately 1,184,513 acres of vernal pool habitat as critical habitat for Contra Costa goldfields and other vernal pool species. The primary factors threatening the continued existence of this plant are habitat loss and degradation. Damage or destruction of vernal pool habitat happens quickly and easily due to the extremely crumbly nature of the soil and the dependency of the pool upon an intact durapan or impermeable subsurface soil layer. Threats to the Contra Costa goldfields are posed by urbanization, agricultural land conversion, drainage, vernal pool and pond construction, ditch construction, OHV use, road maintenance, or random natural events.

Cook's Desert-parsley

The primary reference for this species is:

USFWS. 2002c. Determination of Endangered Status for *Lomatium cookii* (Cook's Lomatium) and *Limnanthes floccosa* ssp. *grandiflora* (Large-Flowered Woolly Meadowfoam) from Southern Oregon. Federal Register 67(216): 68003-68015.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the State Supervisor, USFWS, Oregon Fish and Wildlife Office, Portland, Oregon.

Cook's desert-parsley (*Lomatium cookii*) occurs in vernal pool habitats in a small area of Jackson County, southwestern Oregon. It is also known to occur in seasonally wet habitats at a few sites in Josephine County, the adjacent county to the west. Cook's desert-parsley is known to occur at about 15 sites in Jackson County and at about 21 sites in Josephine County (M. Jones, USDI BLM 2002; Oregon Natural Heritage Information Center Database 2002).

Cook's desert-parsley occurs within a 32-square-mile landform in southwestern Oregon known as the Agate Desert in Jackson County. This landform is characterized by shallow soils, a relative lack of trees, sparse prairie vegetation, and agates commonly found on the soil surface (Oregon Natural Heritage Program 1997). Vernal pools in the Agate Desert vary in size from 3 to 100 feet across, and attain a maximum depth of about 12 inches (Oregon Natural Heritage Program 1997). Common associated native species in these vernal pools include popcorn flower, a rush, navarretia, common woolly meadowfoam, and annual hairgrass.

Cook's desert parsley also occurs in another area of about 4 square miles in adjacent Josephine County. This area, referred to as French Flat, is located within the Illinois Valley near the Siskiyou Mountains. In this area, Cook's desert parsley grows in wet meadow areas underlain with floodplain bench deposits that contain sufficient clay to form a clay pan at 24 to 35 inches below the soil surface (U.S. Department of Agriculture 1983). The clay pan creates seasonally wet areas similar to the vernal pools of the Agate Desert, but mostly lacking in mound-swale topography. Common associated species include California oatgrass, popcorn flower; horkelia; mariposa lily, and trout lily. The surrounding forest contains ponderosa pine and Jeffrey pine.

Cook's desert-parsley is a perennial forb in the carrot family that grows from a slender, twisted taproot. The species is adapted to grow, flower, and set seed during the short time that water is available in the spring, finishing its life cycle before the dry hot summers.

Cook's desert-parsley was federally listed as endangered on November 7, 2002. Designation of critical habitat for this species has been deferred. The primary threat to Cook's desert-parsley is the destruction of vernal pool habitat by industrial and residential development, including road and powerline construction and maintenance. Agricultural conversion, certain grazing practices, and OHV use also contribute to population declines and local extirpations. Recent evidence also indicates that non-native annual grasses, particularly medusahead (*Taeniatherum medusae*), are a greater problem than previously believed. Unlike native perennial bunchgrasses that originally occupied the area, annual grasses die back each year, creating a buildup of thatch from the dead leaves that interferes with the seed germination of native species. Current observations indicate that, without control of annual grasses through mowing, grazing, or prescribed burns, populations tend to decrease over time, and could be extirpated within a relatively short time frame as a result of competition with non-native grasses (Borgias 2002). Additionally, Cook's desert-parsley sites in Josephine County are threatened by habitat alteration associated with gold mining and woody species encroachment resulting from fire suppression.

Large-flowered Woolly Meadowfoam
The primary reference for this species is:
USFWS. 2002c. Determination of Endangered Status for *Lomatium cookii* (Cook's Lomatium) and *Limnanthes floccosa* ssp. *grandiflora* (Large-Flowered Woolly Meadowfoam) from Southern Oregon. Federal Register 67(216): 68003-68015.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the State Supervisor, USFWS, Oregon Fish and Wildlife Office, Portland, Oregon.

Large-flowered woolly meadowfoam (*Limnanthes floccosa* ssp. *grandiflora*), like Cook's desert-parsley discussed in the previous species account, occurs in vernal pool habitats in a small area of Jackson County, southwestern Oregon. The species is known to occur at about 15 sites in Jackson County (M. Jones, USDI BLM 2002; Oregon Natural Heritage Information Center Database 2002).

The large-flowered woolly meadowfoam occurs within the Agate Desert, a landform that was described in the previous species account. This landform is characterized by shallow soils, a relative lack of trees, sparse prairie vegetation, and agates commonly found on the soil surface (Oregon Natural Heritage Program 1997). Vernal pools in the Agate Desert vary in size from 3 to 100 feet across, and attain a maximum depth of about 12 inches (Oregon Natural Heritage Program 1997). Common associated native species in these vernal pools include popcorn flower, a rush, navarretia, common woolly meadowfoam, and annual hairgrass.

The large-flowered woolly meadowfoam is a delicate annual of the meadowfoam family that is covered with short, fuzzy hairs. Like Cook's desert-parsley, plants are adapted to grow, flower, and set seed during the short time that water is available in the spring, finishing their life cycle before the dry hot summers. Each year, plant populations exhibit some natural variation in numbers, related primarily to temperature and rainfall conditions for that year.

The large-flowered woolly meadowfoam was federally listed as endangered on November 7, 2002. Designation of critical habitat has been deferred. The primary threat to the large-flowered woolly meadowfoam is the destruction of vernal pool habitat by industrial and residential development, including road and powerline construction and maintenance. Agricultural conversion, certain grazing practices, and OHV use also contribute to population declines and local extirpations. Recent evidence also indicates that non-native annual grasses, particularly medusahead (*Taeniatherum medusae*), are a greater problem than previously believed, as discussed in the species account for Cook's desert-parsley.

Butte County Meadowfoam

The primary references for this section are:

USFWS. 1992f. Determination of Endangered Status for the Plant *Limnanthes floccosa* ssp. *californica* (Butte County Meadowfoam). Federal Register 57(110): 24192-24199.

and

Sacramento USFWS Office. No Date. Butte County Meadowfoam (*Limnanthes floccosa* ssp. *californica*). Available at: http://sacramento.fws.gov/es/plant_ssp_accts/.

References cited in this section are internal to the above-referenced Federal Register document. They are included in the Bibliography.

The Butte County meadowfoam (*Limnanthes floccosa* ssp. *californica*) is restricted to a narrow, 25-mile strip along the eastern flank of the Sacramento Valley from central Butte County, California, to the northern portion of Chico (Jokerst 1989). The species occurs in three types of seasonal wetland habitats: along the edges of vernal pools and ephemeral streams, and occasionally around the edges of isolated vernal pools. It is generally found on level to gently sloping terrain on poorly drained soils with shallow soil layers that are impermeable to water infiltration. Plants thrive in waterlogged soils and tolerate periodic submergence.

The Butte County meadowfoam is a densely pubescent, herbaceous winter annual. White flowers with dark yellow veins appear in late March through April. The plant is largely self-pollinating because the structure of the flowers prevents them from fully opening. This species has poor seed dispersal, making it poorly equipped to escape chance catastrophes.

The Butte County meadowfoam was federally listed as endangered on June 8, 1992. On August 6, 2003, the USFWS designated approximately 1,184,513 acres of vernal pool habitat as critical habitat for the Butte County meadowfoam and other vernal pool species. The primary threat to this species is urban development in and around the City of Chico. In addition, conversion of the plant's habitat for agricultural purposes is a threat. Road widening or realignment, overgrazing by livestock, garbage dumping, OHV use, competing non-native vegetation and random extinction as a result of the small, isolated nature of the remaining populations all threaten the Butte County meadowfoam to some degree.

Munz's Onion

The primary reference for this section is:

USFWS. 1998g. Determination of Endangered or Threatened Status for Four Southwestern California Plants from Vernal Wetlands and Clay Soils. Federal Register 63 (197): 54975-54994.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Carlsbad Field Office, Carlsbad, California.

Munz's onion (*Allium munzii*) occurs in mesic clay soils in western Riverside County. This species is frequently found in association with southern needlegrass grassland, mixed grassland, and grassy openings in coastal sage scrub or, occasionally, in cismontane juniper woodlands (California Department of Fish and Game 1989, Mistretta 1993).

Munz's onion, a member of the lily family, is a perennial herb, 0.5 to 1.2 feet tall, originating from a bulb. The flower cluster consists of 10 to 35 white flowers that become red with age. In response to rainfall and other factors, perennial bulbs may not produce aerial leaves or flowers in a given year or may produce only leaves. As a result, fluctuations in numbers of observed individuals can be misleading.

Munz's onion was federally listed as endangered on October 13, 1998. On June 4, 2004, the USFWS proposed designating 227 acres of Federal land in western Riverside County, California as critical habitat for the species. Critical habitat has not been designated. Factors that threaten the species include habitat destruction and fragmentation from agricultural and urban development, pipeline construction, alteration of wetland hydrology by draining or excessive flooding, channelization, OHV activity, cattle and sheep grazing, weed abatement, fire suppression practices, and competition from alien plant species

San Jacinto Valley Crownscale
The primary reference for this section is:
USFWS. 1998g. Determination of Endangered or Threatened Status for Four Southwestern California Plants from Vernal Wetlands and Clay Soils. Federal Register 63 (197): 54975-54994.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Carlsbad Field Office, Carlsbad, California.

San Jacinto Valley crownscale (*Atriplex coronata* var. *notatior*) is restricted to the San Jacinto, Perris, Menifee and Elsinore valleys of western Riverside County. It is found only in highly alkaline, silty-clay soils in association with the Traver-Domino-Willows soil association. Common habitat types include areas that are typically flooded by winter rains, such as alkali sink scrub, alkali playa, vernal pools, and, to a lesser extent, annual alkali grassland communities (Bramlet 1993a, Roberts 1993b). The duration and extent of flooding are extremely variable from one year to the next. Populations of the San Jacinto Valley crownscale are primarily associated with the San Jacinto River and Old Salt Creek tributary drainages (Roberts 1993b, Roberts 1997, McMillan 1997). The majority of the population centers are located on privately-owned lands. This plant is not known to occur on federal lands.

The San Jacinto Valley crownscale is an erect annual that grows to a height of 4 to 12 inches. The species germinates after the water has receded. It usually flowers in April and May and sets fruit by May or June (Bramlet 1992).

The San Jacinto Valley crownscale was federally listed as endangered on October 13, 1998. Critical habitat has not been designated. Factors that threaten the species include habitat destruction and fragmentation from agricultural and urban development, pipeline construction, alteration of wetland hydrology by draining or excessive flooding, channelization, OHV activity, cattle and sheep grazing, weed abatement, fire suppression practices, and competition from alien plant species.

Thread-leaved Brodiaea
The primary reference for this section is:
USFWS. 1998g. Determination of Endangered or Threatened Status for Four Southwestern California Plants from Vernal Wetlands and Clay Soils. Federal Register 63 (197): 54975-54994.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Carlsbad Field Office, Carlsbad, California.

Thread-leaved brodiaea (*Brodiaea filifolia*) is a perennial herb that occurs on gentle hillsides, valleys, and floodplains in mesic, southern needlegrass grassland and alkali grassland plant communities in association with clay, loamy sand, or alkaline silty-clay soils (California Department of Fish and Game 1981, Bramlet 1993a). Sites occupied by this species are frequently intermixed with, or near, vernal pool complexes, such as near San Marcos (San Diego County), the Santa Rosa Plateau, and southwest of Hemet in Riverside County. Flowers bloom from May to June and are arranged in a loose umbel. The fruit is a capsule (Munz 1974, Keator 1993). This species is known to hybridize with other brodiaea species in its range (orcutt's brodiaea, dwarf brodiaea, and possibly chaparral brodiaea), where these species coexist (Boyd et. al. 1992; Morey 1995; California Natural Diversity Database 1997).

The historical range of the species extends from the foothills of the San Gabriel Mountains at Glendora (Los Angeles County), east to Arrowhead Hot Springs in the western foothills of the San Bernardino Mountains (San Bernardino County), and south through eastern Orange and western Riverside counties to Carlsbad in northwestern San Diego County, California (Morey 1995, California Natural Diversity Database 1997). At present, its entire range occupies about 825 acres of suitable habitat. Existing populations are clustered in the cities of Vista, San Marcos, and Carlsbad, and in the vicinity of the Santa Rosa Plateau in southwestern Riverside County, or are scattered within the counties of Orange, Los Angeles, Riverside, San Bernardino, and San Diego.

Thread-leaved brodiaea was listed as threatened on October 13, 1998. On December 8, 2004,, the USFWS proposed designating approximately 4,690 acres of land in 10 units in Los Angeles, San Bernardino, Orange, and San Diego counties, California. Thread-leaved brodiaea and its habitat have been substantially reduced by urbanization, agricultural conversion, and disking for fire and weed control. In Riverside County, most of the annual alkaline grassland near the San Jacinto River and southwest of Hemet has been urbanized or converted to dryland farming or more intensive cultivation. Additionally, thread-leaved brodiaea is vulnerable to deep disking or repeated disking. Thus, areas that were disked and have partially recovered after being left fallow for a period of time tend to support reduced and gradually declining populations of the species, if any have survived. The most important threats to this species are urbanization, conversion of habitat to farming, and disking for fire and weed control.

Hartweg's Golden Sunburst

The primary reference for this section is:
USFWS. 1997e. Determination of Endangered Status for *Pseudobahia bahiifolia* (Hartweg's Golden Sunburst) and Threatened Status for *Pseudobahia peirsonii* (San Joaquin Adobe Sunburst), Two Grassland Plants from the Central Valley of California. Federal Register 62 (25): 5542-5551.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

Hartweg's golden sunburst (*Pseudobahia bahiifolia*) is a species of sunflower that is endemic to the non-native grassland and grassland-blue oak woodland community ecotone of the California's Central Valley (southern Sacramento Valley and San Joaquin Valley). Non-native annual grasses and forbs invaded the low elevation plant communities of California during the days of the Franciscan missionaries in the 1700s. These non-native grasses now account for up to 80% or more of the floral composition of the grasslands of California (Heady 1956). Non-native grasses are able to outcompete the native flora because they germinate in late fall, prior to the germination of the native forbs such as Hartweg's golden sunburst. Hartweg's golden sunburst occurs in a distinctive microhabitat within the larger matrix of non-native annual grassland. The top portion of the mounded topography where the grass cover is minimal (Stebbins 1991).

Hartweg's golden sunburst is a few-branched annual 2 to 6 inches tall. The bright yellow flower heads, produced in March or April, are solitary at the ends of branches. The range of this species is strongly correlated with the distribution of the Amador and Rocklin soil series (Stebbins 1991). Both series generally consist of shallow, well-drained, medium-textured soils that exhibit strong mound microrelief. Such topography is characterized by a series of mounds interspersed with shallow basins that may pond water during the rainy season (Bates and Jackson 1987). Hartweg's golden sunburst nearly always occurs on the north or northeast facing slopes of the mounds, with the highest plant densities on upper slopes with minimal grass cover (Stebbins 1991). This plant presently occurs only in the eastern San Joaquin Valley in Stanislaus, Madera, and Fresno counties, a range of approximately 95 miles. One population occurs on land owned and administered jointly by the Bureau of Reclamation and a private owner; the remaining populations all occur on privately-owned property (California Natural Diversity Data Base 1996).

Hartweg's golden sunburst was federally listed as endangered on February 6, 1997. Critical habitat has not been designated. Conversion of native habitat to residential development is the primary threat to the existence of Hartweg's golden sunburst. To a lesser degree, agriculture, competition from aggressive exotic plants, incompatible grazing practices, mining, and other human impacts actions also threaten the species (California Natural Diversity Data Base 1996).

San Joaquin Adobe Sunburst

The primary reference for this section is:
USFWS. 1997e. Determination of Endangered Status for *Pseudobahia bahiifolia* (Hartweg's Golden Sunburst) and Threatened Status for *Pseudobahia peirsonii* (San Joaquin Adobe Sunburst), Two Grassland Plants from the Central Valley of California. Federal Register 62 (25): 5542-5551.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

San Joaquin adobe sunburst (*Pseudobahia peirsonii*), like Hartweg's golden sunburst discussed in the previous species account, is a sunflower species that is endemic to the non-native grassland and grassland-blue oak woodland community ecotone of the California's central valley. The microhabitat preferred by the San Joaquin adobe sunburst is areas of heavy adobe clay soils between mounds, where the water retention properties are high (vernal pools).

The San Joaquin adobe sunburst is an erect annual herb in the aster family, about 4 to 18 inches tall, that produces bright yellow inflorescences. This species occurs only on heavy adobe clay soils over a range of approximately 120 miles through Fresno, Tulare, and Kern counties. This restriction is likely associated with the ability of these clay soils to retain moisture longer into the summer dry season (Stebbins 1991). These soils are mainly distributed in the valleys and flats near the foothills of the southeastern San Joaquin Valley. The San Joaquin adobe sunburst is concentrated in three major locations: east of Fresno in Fresno County; west of Lake Success in Tulare County; and northeast of Bakersfield in Kern County. One population occurs on land administered by the Fresno Flood Control District; two populations occur on land administered by the U. S. Army Corps of Engineers; and all other populations occur on privately-owned land (California Natural Diversity Data Base 1996). The intrusive and aggressive characteristics of herbaceous weedy species appear to be detrimental to habitat quality of this rare plant. Some of the common non-native associates of this species include wild oat, charlock mustard, soft brome, red brome, and redstem stork's bill.

The San Joaquin adobe sunburst was listed as threatened on February 6, 1997. Critical habitat has not been designated. The primary threat to the species is conversion of natural habitat to residential development. In addition, road maintenance projects, recreational activities, competition from non-native plants, agricultural land development, incompatible grazing practices, a flood control project, transmission line maintenance, and other human impacts also may threaten the species.

Purple Amole

The primary reference for this section is:
USFWS. 2000c. Determination of Threatened Status for *Chlorogalum purpureum* (Purple Amole), a Plant from the South Coast Ranges of California. Federal Register 65 (54): 14878-14888.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Ventura Fish and Wildlife Office, Ventura, California.

Purple amole (*Chlorogalum purpureum* var. *purpureum*) is a bulb-forming perennial herb in the lily family. It has a basal rosette of linear leaves and a widely branching stem that supports bluish-purple flowers. Purple amole occurs in grassland, oak woodland, and oak savannah between 1,000 and 2,050 feet in elevation in the south coast ranges of California. It is known from oak woodlands and grasslands at three sites near Jolon in Monterey County on lands administered by the Department of the Army (Fort Hunter Liggett). Historically, appropriate habitat may have existed east of the base, in Jolon Valley, but most of the flat areas in that valley have been converted to cropland, pasture, or vineyards. At Fort Hunter Liggett, the plant occurs on flat or gently sloping terrain with a gravelly surface underlain by clay soils, often where other herbaceous vegetation is sparse.

Reproduction in this species is primarily by seed. Each flower contains six ovules, although not all develop into seeds in the wild (Hoover 1964). The species is reported to be self-compatible, and insect pollination appears to result in increased seed set (Wilken 1998; M. Elvin, USFWS 1998). Clonal reproduction by longitudinal splitting of the bulbs is rare (Hoover 1940). Like other members of the lily family, this species probably forms mycorrhizal relationships (root-hyphae relationships with a fungus), which can aid in nutrient and water uptake by the host plant and can alter growth and competitive interactions between species (Allen 1991).

Purple amole was federally listed as threatened on April 19, 2000. On April 24, 2003, the USFWS designated approximately 20,000 acres of land in Monterey and San Luis Obispo counties, California, as critical habitat for two varieties of purple amole. The primary threats to this plant are the loss, fragmentation, and alteration of habitat and direct elimination of plants from construction and use of military training facilities, military field training activities, displacement by nonnative annual grasses, and potentially by alteration of fire cycles due to military training. Livestock grazing and associated habitat changes may threaten this taxon if grazing is resumed in occupied habitat in the future.

Keck's Checker-mallow
The primary reference for this section is:
USFWS. 2000d. Determination of Endangered Status for *Sidalcea keckii* (Keck's Checker-mallow) from Fresno and Tulare Counties, CA. Federal Register 65 (32): 7757-7764.

Keck's checker-mallow (*Sidalcea keckii*) is a slender, annual herb known from serpentine-derived clay soils in the foothill annual grasslands of the central western Sierra Nevada Mountains. It occurs toward the southern end of the San Joaquin Valley in Tulare and Fresno counties, California. The species grows in relatively open areas on grassy slopes, and like many serpentine species, appears to compete poorly with densely growing non-native annual grasses. One population of the species occurs on 20 to 40% slopes of red or white-colored clay in sparsely-vegetated annual grasslands. This population occurs on a privately-owned parcel of land that is used for livestock grazing.

Keck's checker mallow is a slender, erect annual herb belonging to the mallow family. The species grows 6 to 13 inches tall, and produces deep pink flowers in April and May. The specific requirements for seed germination in the wild, typical germination dates, and how long the seeds remain viable in the soil, are not known. However, it is assumed that this species is able to form a persistent soil seed bank.

Keck's checker-mallow was federally listed as endangered on February 16, 2000. On March 18, 2003, the USFWS designated approximately 1,085 acres in Fresno and Tulare counties as critical habitat for the species. The species is threatened by urban development, agricultural land conversion (particularly to citrus orchards), and competition from non-native grasses. Populations are also vulnerable to random events because they are small in size and number.

California Jewelflower
The primary reference for this section is:
Sandoval, T.M., and E.A. Cypher. 1996. California Jewelflower, *Caulanthus californicus*. Endangered Species Profiles, Species Featured in Recovery Plan for San Joaquin Valley Arid Upland and Riparian Communities. Endangered Species Recovery Program. Fresno, California. Available at: http://arnica.csustan.edu/esrpp.

California jewelflower (*Caulanthus californicus*) is an annual plant that is endemic to California. The species is found in several plant communities, including non-native grassland, Upper Sonoran subshrub scrub, and cismontane juniper woodland and scrub. Herbaceous cover is dense at most California jewelflower sites. Native plant species, such as annual fescue, clovers, red maids, and goldfields comprise a high proportion of the vegetation at many of the known locations. On the Carrizo Plain, California jewelflower occurs primarily on the burrow systems of giant kangaroo rats, another endangered specie. California jewelflower has been reported from elevations ranging from approximately 250 to 2,950 feet and from level terrain to 25% slopes. Primary soil types at known sites are subalkaline, sandy loams. The historical range of the California jewelflower included the floor of the San Joaquin Valley in Fresno, Kern, and Tulare counties; the Carrizo Plain (San Luis Obispo County) and the Cuyama Valley (Santa Barbara and Ventura counties); the Sierra Nevada foothills at the eastern margin of the San Joaquin Valley in Kern County; and foothills west of the San Joaquin Valley, in Fresno, Kern, and Kings counties. As of 1986, all natural occurrences of California jewelflower on the San Joaquin and Cuyama Valley floors had been extirpated. Today, known populations are confined to three areas in hilly terrain west of the San Joaquin Valley: the Carrizo Plain, Santa Barbara Canyon (adjacent to the Cuyama Valley in Santa Barbara County), and the Kreyenhagen Hills (Fresno County). The Carrizo Plain and Kreyenhagen Hills populations are on public land administered by the BLM, as is approximately 10% of the Santa Barbara Canyon population. Additional populations of California jewelflower may persist in the foothills of Fresno, Kern, and Kings counties, where potential habitat remains in private rangeland. Several experimental introductions of California jewelflower have been attempted in Kern, Santa Barbara, and Tulare counties, but none of the populations have persisted.

Seeds of the California jewelflower begin to germinate in the fall when the rainy season begins, but additional seedlings may continue to emerge for several months. The seedlings develop into rosettes during the winter months, and the stem elongates as flower buds begin to appear in February or March. Flowering and seed set continue until the plants die, which may occur as late as May in years of favorable rainfall and temperatures. The flowers are pollinated by insects. Seed-dispersal agents are not known, but may include gravity, seed-eating animals, wind, and water. California jewelflower probably forms a persistent seed bank.

The California jewelflower was federally listed as endangered on July 19, 1990. Critical habitat has not been designated. The primary reason for its decline was habitat destruction by conversion to agriculture and urbanization. Oilfield activity may have eliminated a few sites in the foothills at the western margin of the San Joaquin Valley. Potential threats to the remaining populations include competition from introduced plant species, pesticide effects on pollinators, and small population size. Populations on private land in the upper portion of Santa Barbara Canyon are subject to cattle grazing throughout the growing season, but the magnitude of threat posed by livestock is unknown. Residential development also threatens the privately-owned portion of the Santa Barbara Canyon population.

San Joaquin Woolly-threads

The primary reference for this section is:

Brown, N. L., and E. A. Cypher. 1997a. San Joaquin Woolly-threads, *Lembertia congdonii*. Endangered Species Profiles, Species Featured in Recovery Plan for San Joaquin Valley Arid Upland and Riparian Communities. Endangered Species Recovery Program. Fresno, California. Available at: http://arnica.csustan.edu/esrpp.

San Joaquin woolly-threads (*Lembertia congdonii*) is an annual herb that occurs in non-native grassland, Valley saltbush scrub, Interior Coast Range saltbush scrub, and Upper Sonoran subshrub scrub in California. Historically, the species occurred primarily in the San Joaquin Valley, with a few occurrences in the hills to the west and in the Cuyama Valley of San Luis Obispo and Santa Barbara counties. Since 1986, many new occurrences have been discovered in the hills and plateaus west of the San Joaquin Valley. The largest extant metapopulation occurs on the Carrizo Plain Natural Area in San Luis Obispo County. Much smaller metapopulations are found in Kern County near Lost Hills, in the Kettleman Hills of Fresno and Kings counties, and in the Jacalitos Hills of Fresno County. Isolated occurrences are known from the Panoche Hills in Fresno and San Benito counties, the Bakersfield vicinity in Kern County, and the Cuyama Valley.

San Joaquin woolly-threads occurs on sandy, sandy loam, or silty soils with neutral to subalkaline pH, at elevations ranging from approximately 197 to 2,625 feet. The species typically occupies microhabitats with less than 10% shrub cover, although herbaceous cover may be either sparse or dense. Plant species that often occur with San Joaquin woolly-threads include red brome, red-stemmed filaree, goldfields, Arabian grass, and mouse-tail fescue.

The phenology of San Joaquin woolly-threads varies with weather and site conditions. In years of below-average precipitation, few seeds of this species germinate, and those that do typically produce tiny plants. Seed germination may begin as early as November, but usually occurs in December and January. The species typically flowers between late February and early April, but flowering may continue into early May if conditions are optimal.

Populations in the northern part of the range flower earlier than those farther south. Each plant may have from 1 to more than 400 flower heads. Seed production depends on plant size and the number of flower heads, and can range from 10 to 2,500 seeds per individual. The seeds are shed immediately upon maturity, and all trace of the plants disappears after death in April or May. Seed dispersal agents are unknown, but possible candidates include wind, water, and animals. Seed dormancy mechanisms apparently allow the formation of a substantial seed bank in the soil.

San Joaquin woolly-threads was federally listed as endangered on July 19, 1990. Critical habitat has not been designated. Habitat loss was responsible for the decline of the species, with the majority of the occurrences in the San Joaquin and Cuyama valleys extirpated by intensive agriculture. In addition, several sites in and around Bakersfield were eliminated by urban and intensive oilfield development. Current threats to San Joaquin woolly-threads include commercial and agricultural development, increased intensity of land use in oilfields or pastures, and competition from introduced plants.

Bakersfield Cactus

The primary reference for this section is:

Brown, N.L., and E.A. Cypher. 1997b. Bakersfield Cactus, *Opuntia basilaris* var. *treleasei*. Endangered Species Profiles, Species Featured in Recovery Plan for San Joaquin Valley Arid Upland and Riparian Communities. Endangered Species Recovery Program. Fresno, California. Available at: http://arnica.csustan.edu/esrpp.

Bakersfield cactus (*Opuntia treleasei*) is a perennial plant with fleshy, flattened, green stems (pads), that is endemic to a limited area of central Kern County in the vicinity of Bakersfield, California. The species is characteristic of the Sierra-Tehachapi saltbush scrub plant community, but populations near Caliente are in blue oak woodland, and the Cottonwood Creek population is in riparian woodland. Many sites for Bakersfield cactus support a dense growth of red brome and other annual grasses. Sand Ridge is characterized by sparse vegetation and a preponderance of native species such as California filago and yellow pincushion. Soils supporting the species typically are sandy, although gravel, cobbles, or boulders also may be present. Known populations occur on flood plains, ridges, bluffs, and rolling hills, and occur at elevations between 396 and 1,800 feet. As of 1987, the northern, southern, eastern, and western limits of the known range of this species, respectively, were Granite Station, Comanche Point, Caliente, and Oildale, California.

Few details on the life history of Bakersfield Cactus are available. The fleshy stems, tiny, short-lived leaves, shallow root systems, and specialized physiology are adaptations to growth in arid environments. Known to typically flower in May, the reproductive biology of this taxon has not been studied. Certain other species of *Opuntia* (pricklypear) species require cross-pollination for seed-set, and many are pollinated by bees. Vegetative reproduction is common in Bakersfield cactus and several related species. Fallen pads easily root if sufficient water is available, although the cactus does not survive prolonged inundation. Seeds, which are produced infrequently, require warm, wet conditions to germinate, a combination which is extremely rare in the Bakersfield area. Pads may be dispersed by flood waters, but seed dispersal agents are unknown.

The Bakersfield cactus was federally listed as endangered on July 19, 1990. Critical habitat has not been designated. The primary reason for the decline of the species was habitat loss. The formerly extensive tracts of Bakersfield cactus near Edison and Lamont were destroyed by conversion to row crops and citrus groves; much of the conversion occurred prior to 1931. Residential development, petroleum production, sand and gravel mining, and OHV activity also have contributed to habitat loss and fragmentation of this plant, and continue to threaten the existing populations. Other threats include competition from introduced grasses, air pollution, and low genetic diversity.

Kern Mallow
The primary reference for this section is:
USFWS. 1998h. Recovery Plan for Upland Species of the San Joaquin Valley, California. Portland, Oregon.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The Kern mallow (*Eremalche kernensis*) typically occurs in valley saltbush scrub habitats in the San Joaquin Valley, California. The species grows under and around spiny and common saltbushes and in patches with other herbaceous plants, rather than in the intervening alkali scalds. Associated herbs include red brome, red-stemmed filaree, woolly goldfields, and white Sierran layia. The Kern mallow typically grows on alkaline sandy loam or clay soils at elevations of 315 to 900 feet (Wolf 1938, California Department of Fish and Game 1995), in areas where shrub cover is less than 25% (Talyor and Davilla 1986). The amount of herbaceous cover varies with rainfall and microhabitat.

The Kern mallow has always had a highly-restricted distribution, occurring only in western Kern County, north of McKittrick. At present, the species occurs intermittently within an area of 40 square miles in Lokern, a local name for the area between Buttonwillow and McKittrick (Taylor and Davilla 1986). This occurrence is best described as a single metapopulation.

The seeds of the Kern mallow typically germinate in January and February, and the plants begin flowering in March. Fruit production begins within a few days after flowers appear, and may continue into May if sufficient moisture is available. The seeds fall from the fruit as soon as they are mature. Seeds are capable of germinating in the following growing season, but at least some of them remain ungerminated. The duration of seed viability in the soil is not known. Seed dispersal agents are unknown, but probably include animals and wind (Taylor and Davilla 1986; and Mazer et al. 1993). Additionally, insects are thought to facilitate reproduction of the species. On occasion, Kern mallow has reinvaded disturbed sites when existing populations remained in adjacent areas to provide sources of seed (Mitchell 1989, E. Cypher unpublished observation).

The Kern mallow was federally listed as endangered on July 19, 1990. Critical habitat has not been designated. Loss and degradation of habitat in the Lokern area have been responsible for the decline of the species. Approximately 85% of the Kern mallow habitat in Lokern is privately-owned, and is thus vulnerable to development for many potential uses (Taylor and Davilla 1986, Presley 1994, California Department of Fish and Game 1995). Oil exploration and maintenance of pipelines and utility corridors continue to disturb occupied habitat. Grazing may threaten the population by reducing reproductive output of the species (Mazer et al. 1993), but light to moderate grazing may reduce competition in areas that are dominated by aggressive exotics (Cypher 1994). Another potential threat to the species is the reduction of pollinator populations caused by the use of malathion and other pesticides.

Springville Clarkia
The primary reference for this section is:
USFWS. 1996d. Determination of Threatened Status for Four Plants from the Foothills of the Sierra Nevada Mountains in California. Federal Register 63 (177): 49022-49035.

Springville clarkia (*Clarkia springvillensis*) is found on granitic soils in sunny sites from 1,220 to 3,000 feet in elevation, and grows mostly on the uphill slope of roadbanks, on small decomposing granitic domes, and in openings within the blue oak woodland community in the foothills of the southern Sierra Nevada Mountains of Tulare County, California. All populations of this species but one are found within about a 15-mile range, with the remaining population occurring 16 miles to the northwest. One site is partially protected by the California Department of Fish and Game, one is on public land, eight are on Forest Service-administered land, and five are on privately-owned land. The largest population occurs on the 4.5-acre preserve administered by the California Department of Fish and Game. Populations along roadsides have become restricted to a narrow band just above a zone of herbicide use and just below heavily grazed terrain.

Springville clarkia is an erect annual herb that is approximately 3 feet tall. It produces lavendar-pink flowers, which appear in May to July.

Springville clarkia was federally listed as threatened on September 14, 1998. Critical habitat has not been designated. The species is threatened by urban development, intense livestock grazing, and roadway maintenance activities. Because it has low numbers of populations and small population sizes, it is also vulnerable to extirpation from random events.

Red Hills Vervain

The primary reference for this section is:

USFWS. 1998i. Determination of Threatened Status for Four Plants from the Foothills of the Sierra Nevada Mountains in California. Federal Register 63 (177): 49022-49035.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

The Red Hills vervain (*Verbena californica*) is an erect perennial herb that is restricted to intermittent and perennial streams within serpentine areas of the Red Hills of Tuolumne County, California, at elevations of between 850 and 1,150 feet. Within this narrow range, the total area occupied by the populations is estimated to be 90 acres (California Natural Diversity Database 1997). The majority of populations occur in drainages that feed into Don Pedro Reservoir, primarily Six Bit Gulch and its tributaries. Another population is on Andrew Creek, which feeds into the Tullock Reservoir (California Department of Fish and Game 1993, California Natural Diversity Database 1997). Four populations are wholly on public lands, and two are partially on public lands, although these six sites contain only 15% of the total known number of plants. The remaining 85% of Red Hills vervain plants are on privately-owned lands.

The Red Hills vervain blooms from May through September. Blossoms are white to purple, with one to five flowers growing at the top of each spike.

Red Hills vervain was federally listed as threatened on September 14, 1998. Critical habitat has not been designated. The species is threatened by urbanization, recreational placer gold mining, OHV use, inadequate regulatory mechanisms, dumping, and heavy grazing and trampling. Because there are few populations and low numbers, it is also vulnerable to extirpation from random events.

Cushenbury Milk-vetch

The primary references for this section are:

USFWS. 2002d. Proposed Designation of Critical Habitat for Five Carbonate Plants from the San Bernardino Mountains in Southern California. Federal Register 67 (29): 6577-6612.

and

USDI BLM. 2001c. Biological Evaluation on Effects of California Desert Conservation Area Plan as Amended and Proposed to be Amended by the NEMO and NECO Preferred Alternatives and With Other Interim Measures on Ten Threatened and Endangered Plants. California Desert District, BLM. Riverside, California.

References cited in this section are internal to the above-referenced documents. References from the BLM (2001c) are included in the Bibliography. A complete list of references from USFWS (2002d) is available from the USFWS Carlsbad Fish and Wildlife Office, Carlsbad, California.

The Cushenbury milk-vetch (*Astragalus albens*) is restricted primarily to carbonate deposits and their derived soils in San Bernardino County, California. The area of the San Bernardino Mountains in which the species occurs contains outcrops of carbonate substrates, primarily limestone and dolomite, in several bands running on an east-west axis along desert-facing slopes, and is generally known as the carbonate belt. Occurrences of the Cushenbury milk-vetch are scattered in the northeastern San Bernardino Mountains extending from Dry Canyon southeastward to the head of Lone Valley, a range of 15 miles (Barrows 1988c, California National Park Service 2001, California Natural Diversity Data Base 2001). The species is typically found within singleleaf pinyon-Utah juniper, blackbush scrub, singleleaf pinyon, pinyon woodland, pinyon-juniper woodland, and Joshua tree woodland vegetation communities (Gonella 1994, Gonella and Neel 1995, Neel 2000). Plants closely associated with the Cushenbury milk-vetch include flannelbush, blackbush, mound cactus, desert almond, and Mojave yucca (Gonella 1994, Gonella and Neel 1995). The Cushenbury milk-vetch is typically found on carbonate soils derived directly from decomposing limestone bedrock along rocky washes. It is generally found in areas with an open canopy cover, little accumulation of organic material, rocky substrate cover exceeding 75%, and gentle to moderate slopes (5 to 30%) (Neel 2000). Most occurrences of the species are at elevations between 5,000 and 6,600 feet. Most of the occurrences below about 5,000 feet are found in rocky washes with limestone outwash from erosion (California Natural Diversity Database 2001, San Bernardino National Forest GIS data 2001). Soils at sites supporting the Cushenbury milk-vetch have a higher percentage of calcium than soils that do not support this species (Gonella and Neel 1995).

Cushenbury milk-vetch is an herbaceous member of the pea family (Fabaceae). Little is known of its life history, and it has been described both as an annual and as a short-lived perennial herb (Barneby 1964b, Munz 1974, Greene 1885, Hickman 1993, Skinner and Pavlik 1994). It is not known whether the plants typically flower and fruit the first year, how long they live, or what conditions might cause them to act as annuals in some cases or perennials in other cases. Flowering occurs from late March to mid-June, and flowers occur in racemes on long peduncles. Pods ripen at least as early as May, and become stiff and papery with long hairs as they mature. Pollen vectors are most likely small bees, given the flower shape and color (Faegri and Van der Pijl 1979 *cited in* USDI BLM 2001c). It is thought that most Cushenbury milk-vetch reproduction occurs by seeds, which have been found to have high viability. Vegetative reproduction has never been reported.

Carbonate-restricted plants do not appear to be specifically linked to early vegetation successional stages after disturbance; however, they are found on some surfaces that are naturally disturbed by landslides and substrate upheaval. For the most part, they occur in habitat that is undisturbed by human activities.

The Cushenbury milk-vetch was federally listed on August 24, 1994. On December 24, 2002, the USFWS designated approximately 4,365 acres in San Bernardino County as critical habitat for the species. The primary threat to the Cushenbury milk-vetch and other carbonate plants is limestone mining. Specific threats include population reduction and habitat loss, degradation, and habitat fragmentation from surface mining activities.

Parish's Daisy

The primary references for this section are:

USFWS. 2002d. Proposed Designation of Critical Habitat for Five Carbonate Plants from the San Bernardino Mountains in Southern California. Federal Register 67 (29): 6577-6612.

and

BLM. 2001c. Biological Evaluation on Effects of California Desert Conservation Area Plan as Amended and Proposed to be Amended by the NEMO and NECO Preferred Alternatives and With Other Interim Measures on Ten Threatened and Endangered Plants. California Desert District, BLM. Riverside, California.

References cited in this section are internal to the above-referenced documents. References from the BLM (2001c) are included in the Bibliography. A complete list of references from USFWS (2002d) is available from the USFWS Carlsbad Fish and Wildlife Office, Carlsbad, California.

Parish's daisy (*Erigeron parishii*), like the Cushenbury milk-vetch discussed in the previous species account, is restricted primarily to carbonate deposits and their derived soils in San Bernardino County, California. Parish's daisy is typically associated with singleleaf pinyon-Utah juniper, singleleaf pinyon, pinyon-juniper woodlands, blackbush scrub, and creosote bush-bursage scrub vegetation communities (Neel 2000, Neel and Ellstrand 2001). Plants closely associated with Parish's daisy include singleleaf pinyon, California juniper, Joshua tree, and

Cushenbury milk-vetch (Gonella 1994, Gonella and Neel 1995, California Natural Diversity Database 2001). Parish's daisy typically grows on limestone or dolomite soils occurring on dry, rocky slopes, shallow drainages, and outwash plains. Some plants grow on a granite/limestone interface, usually when granitic parent material has been overlaid with limestone materials washed down from upslope. The species is generally found at elevations between 3,842 and 6,400 feet (the lower elevations of the carbonate belt; Neel 2000).

Parish's Daisy a range that spans approximately 35 miles along the carbonate belt in the northeastern San Bernardino Mountains, extending from Pioneertown in the east to Furnace Canyon in the west. This distribution includes occurrences on Tip Top Mountain and in Arctic, Cushenbury, Arrastre, and Rattlesnake canyons (Krantz 1979, Barrows 1988a, California Natural Diversity Database 2001).

Parish's daisy is an herbaceous perennial with a long simple taproot that extends for a distance of approximately 20 inches into the loose carbonate alluvium that the species favors. The stems are erect or ascending and may be either numerous or rather few on each plant, but on mature plants are typically at least 20 in number. The flower heads are solitary on bracted stalks, commonly with two to four stalks per stem. The total number of heads on a mature plant can easily equal 50 in a given season. The heads bear lavender ray flowers and yellow disk flowers.

The method of pollination is unknown, but is certainly by insects, based on the conspicuously colored flowers. Likely candidates include bees, butterflies or long-tongued flies. Seed dispersal is unstudied, as is the relative importance of seeds versus possible vegetative spread in the maintenance and expansion of populations, though seedlings have been reported at several sites (Krantz 1979 *cited in* USDI BLM 2001c) and are probably the predominant mode of reproduction. Flowering is reported to occur from May to July, but the peak of flowering seems to be from mid May to mid-June. At least in some years a few plants continue flowering into July and some even into August (Provance 1998). Flower heads have been found to be attacked by insect larvae, but the extent and effect of such damage is unknown (Krantz 1979 *cited in* USDI BLM 2001c).

Carbonate-restricted plants do not appear to be specifically linked to early vegetation successional stages after disturbance; however, they are found on some surfaces that are naturally disturbed by landslides and substrate upheaval. For the most part, they occur in habitat that is undisturbed by human activities.

Parish's daisy was federally listed on August 24, 1994. On December 24, 2002, the USFWS designated approximately 4,420 acres in San Bernardino County as critical habitat for the species. The primary threat to Parish's daisy and other carbonate plants is limestone mining. Specific threats include population reduction and habitat loss, degradation, and habitat fragmentation from surface mining activities.

Cushenbury Buckwheat

The primary references for this section are:

USFWS. 2002d. Proposed Designation of Critical Habitat for Five Carbonate Plants from the San Bernardino Mountains in Southern California. Federal Register 67 (29): 6577-6612.

and

BLM. 2001c. Biological Evaluation on Effects of California Desert Conservation Area Plan as Amended and Proposed to be Amended by the NEMO and NECO Preferred Alternatives and With Other Interim Measures on Ten Threatened and Endangered Plants. California Desert District, BLM. Riverside, California.

References cited in this section are internal to the above-referenced documents. References from the BLM (2001c) are included in the Bibliography. A complete list of references from USFWS (2002d) is available from the USFWS Carlsbad Fish and Wildlife Office, Carlsbad, California.

The Cushenbury buckwheat (*Eriogonum ovalifolium* var. *vineum*), like the species discussed in the previous two species accounts, is restricted primarily to carbonate deposits and their derived soils in San Bernardino County, California. The Cushenbury buckwheat occurs in the northeastern San Bernardino Mountains extending from White Mountain in the west to Rattlesnake Canyon in the east, a distance of approximately 25 miles. Included are occurrences in Arctic and Cushenbury Canyons, Terrace and Jacoby Springs, along Nelson Ridge, and southeast to near Onyx Peak (Barrows 1988b, Brown 1992, Tierra Madre Consultants 1992, Gonella and Neel 1995, California Natural Diversity Database 2001). This species inhabits open areas in singleleaf pinyon-Utah juniper, singleleaf pinyon-mountain juniper, singleleaf pinyon, pinyon, pinyon-juniper, Joshua tree woodlands, and blackbush scrub vegetation communities (Gonella 1994, Gonella and Neel 1995, Neel 2000). Plants closely associated with the Cushenbury buckwheat include flannelbush, big-berry manzanita, green-leaf manzanita, Douglas' phacelia, Joshua tree, singleleaf pinyon, Cushenbury milk-vetch, and Parish's daisy (Gonella 1994, Gonella and Neel 1995, California Natural Diversity Database 2001).

The Cushenbury buckwheat typically grows with soils derived from limestone or other carbonate substrates (Hickman 1993, California Natural Diversity Database 2001). It is generally found on gentle slopes between 10 and 25%, mostly with north or west aspects. Other habitat characteristics include open areas with powdery fine soils and little accumulation of organic material, a canopy cover generally less than 15%, and rock cover exceeding 50%. Its elevational range is between 4,600 and 7,900 feet (Neel 2000).

Cushenbury buckwheat plants are very compact with short woody stems spreading a few centimeters over the ground. The foliage mounds seldom rise more than 4 inches above the surrounding rocks or soil. However, when the plants begin flowering, they send up inflorescences 1 to 5 inches above the foliage. The several to many short woody stems spread and ascend over a very small patch of ground from a thick woody base above a deep and well-developed woody taproot. The foliage of the plant is densely covered with tangled, white hairs on both surfaces. The leaves cover the upper parts of the stems, and are densely grouped so that the ground is generally not visible through the plant. This overall plant density is partly caused by the dried leaves, which do not fall from the plant, but simply turn a dark brown color and cling to the older parts of the stem. This adaptation presumably provides insulation for the plant as well as added protection from water loss through the stems.

Cushenbury buckwheat is a perennial of open areas and appears intolerant of extensive shading, preferring full sunlight, and typically occurs between shrubs rather than under them (White 1997 *cited in* USDI BLM 2001c). Although not well adapted to competing for light, the species is very competitive on sites where tall and fast growing species are excluded by moisture deficiencies, wind, winter cold, or nutrient deficiencies. The compact "cushion" habit probably serves to reduce moisture loss on windy ridges as is true for other species of similar life form (Walter 1973 *cited in* USDI BLM 2001c). The short annual growth intervals and consequent low stature makes these plants poor competitors on sites that are capable of supporting tall or dense vegetation. However, sites where moisture stress is combined with high insolation are highly favorable for plants such as this one. The nutrient deficiencies of limestone soil, exacerbated by the high pH which interferes with mineral uptake, doubtless serve to further reduce competition by fast growing species.

The inflorescence consists of a leafless flowering stem, which bears a single head-like umbel of large, cream-white to reddish flowers. Cushenbury buckwheat flowers primarily in May and June, although later flowering sometimes occurs, and fruits from this main flowering period ripen in July. Pollination of this plant has only recently been studied, and small insects are almost certainly its pollinators (Morita 1998). Both wind and birds appear to play a role in dispersing seeds, although given the extremely restricted distribution of Cushenbury buckwheat, long-distance dispersal is uncommon.

Carbonate-restricted plants do not appear to be specifically linked to early vegetation successional stages after disturbance; however, they are found on some surfaces that are naturally disturbed by landslides and substrate upheaval. For the most part, they occur in habitat that is undisturbed by human activities.

The Cushenbury buckwheat was federally listed on August 24, 1994. On December 24, 2002, the USFWS designated approximately 6,955 acres in San Bernardino County as critical habitat for the species. The primary threat to the Cushenbury buckwheat and other carbonate plants is limestone mining. Specific threats include population reduction and habitat loss, degradation, and habitat fragmentation from surface mining activities.

Cushenbury Oxytheca

The primary references for this section are:

USFWS. 2002d. Proposed Designation of Critical Habitat for Five Carbonate Plants from the San Bernardino Mountains in Southern California. Federal Register 67 (29): 6577-6612.

and

BLM. 2001c. Biological Evaluation on Effects of California Desert Conservation Area Plan as Amended and Proposed to be Amended by the NEMO and NECO Preferred Alternatives and With Other Interim Measures on Ten Threatened and Endangered Plants. California Desert District, BLM. Riverside, California.

References cited in this section are internal to the above-referenced documents. References from the USDI BLM (2001c) are included in the Bibliography. A complete list of references from USFWS (2002d) is available from the USFWS Carlsbad Fish and Wildlife Office, Carlsbad, California.

The Cushenbury oxytheca (*Oxytheca parishii* var. *goodmaniana*), like the species discussed in the three previous species accounts, is restricted primarily to carbonate deposits and their derived soils in San Bernardino County, California. The Cushenbury oxytheca is scattered along the carbonate belt in the northeastern San Bernardino Mountains extending from White Mountain in the west to Rattlesnake Canyon in the east. This distribution includes occurrences near Cushenbury Spring; Cushenbury, Marble, Arctic, Wild Rose, and Furnace canyons; Blackhawk, Mineral, and Tip Top mountains; Terrace Springs; Rose Mine and Green Lead Gold Mine (Gonella and Neel 1995, California National Park Service 2001, California Natural Diversity Database 2001).

The Cushenbury oxytheca is typically found in singleleaf pinyon-Utah juniper, singleleaf pinyon-mountain juniper, singleleaf pinyon, and canyon live oak woodland vegetation communities (Neel 2000). Closely associated plant species include mountain mahogany, big-berry manzanita, yellow rabbitbrush, and needlegrass (California Natural Diversity Database 2001). The Cushenbury oxytheca is typically found on soils derived from limestone, dolomite, or a mixture of limestone and dolomite substrates (Tierra Madre Consultants 1992, Neel 2000). It generally occurs in areas with gentle slopes between 10 and 25 degrees with no apparent preference for aspect, and at elevations between 4,724 and 7,782 feet (Neel 2000).

Cushenbury oxytheca germinates in the fall following the first rains and exists as a vegetative rosette through the winter months. The plant has a relatively long, straight taproot, which presumably taps into supplies of soil moisture below the surface. The basal rosette consists of relatively broad leaves, which are followed in the spring by a slender leafless inflorescence. As the inflorescence matures, the leaves wither and dry, so that by the time of late flowering or fruit ripening the plant typically has no living leaves at all. Cushenbury oxytheca flowers in May and June, producing white flowers with a reddish midrib that are apparently pollinated by insects. Specific pollinators, germination requirements, seed longevity, and most other aspects of the biology of this species are largely unknown. Because the Cushenbury oxytheca is an annual, the number of individual plants present fluctuates from year to year, depending on the seed bank dynamics, rainfall, and temperature. It also has few occurrences, and the total number of individuals at some occurrences is often low, possibly making this species more susceptible to extinction from random environmental events than the other three carbonate plant species.

Carbonate-restricted plants do not appear to be specifically linked to early vegetation successional stages after disturbance; however, they are found on some surfaces that are naturally disturbed by landslides and substrate upheaval. For the most part, they occur in habitat that is undisturbed by human activities.

The Cushenbury oxytheca was federally listed on August 24, 1994. On December 24, 2002, the USFWS designated approximately 3,150 acres in San Bernardino County as critical habitat for the species. The primary threat to the Cushenbury oxytheca and other carbonate plants is limestone mining. Specific threats include population reduction and habitat loss, degradation, and habitat fragmentation from surface mining activities.

Yreka Phlox

The primary reference for this section is:
USFWS. 2000e. Determination of Endangered Status for the Plant Yreka Phlox from Siskiyou County, California. Federal Register 65(23): 5268-5275.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

Yreka phlox (*Phlox hirsuta*) is endemic to Siskiyou County, California, where it grows on serpentine slopes in the vicinity of the City of Yreka (California Native Plant Society 1985). Serpentine soils are rocky mineral soils consisting mostly of rocks with unusually large amounts of magnesium and iron. These rocks are found discontinuously throughout California, in the Sierra Nevada and in the Coast Ranges from Santa Barbara County, California, to British Columbia. Serpentine soils have characteristic physical and chemical properties, such as high concentrations of magnesium, chromium, and nickel, and low concentrations of calcium, nitrogen, potassium, and phosphorus. In addition, serpentine soils alter the pattern of vegetation and plant species composition nearly everywhere they occur.

Yreka phlox is a perennial subshrub that grows approximately 2 to 6 inches tall from a stout, woody base. Pink to purple flowers appear from April to June. This species is found on serpentine soils at elevations from 2,800 to 4,400 feet, in association with Jeffrey pine, incense cedar, and junipers (California Native Plant Society 1985, California Department of Fish and Game 1986, California Natural Diversity Database 1997). Yreka phlox is known from only two locations in the vicinity of Yreka, California. One occurrence is on an open ridge in a juniper woodland within the city limits of Yreka, covering an area of about 37 acres (California Native Plant Society 1977, 1985; California Natural Diversity Database 1997). The second occurrence is about 5 to 6 miles southwest of Yreka in an open Jeffrey pine forest (California Native Plant Society 1977, 1985; California Natural Diversity Database 1997) and includes approximately 160 acres of occupied habitat. These two occurrences are found on a mixture of privately-owned, the City of Yreka, and Forest Service-administrated lands (California Natural Diversity Database 1997).

Yreka phlox was federally listed as endangered on February 3, 2000. The USFWS has determined that future designation of critical habitat is prudent, as resources become available. This species is threatened by urbanization at the City of Yreka location and by inadequate State regulatory mechanisms throughout its range. The small number of populations and small range of the species also make it vulnerable to decline or extirpation caused by random events throughout its range.

Metcalf Canyon Jewelflower

The primary reference for this species is:
USFWS. 1995b. Determination of Endangered Status for Ten Plants and Threatened Status for Two Plants from Serpentine Habitats in the San Francisco Bay Region of California. Federal Register 60: 6671-6685.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

The Metcalf Canyon jewelflower (*Streptanthus albidus* ssp. *albidus*) is endemic to serpentine soils in the region of the San Francisco Bay in California. Serpentine soils are found in discontinuous outcrops in the Sierra Nevada and in the Coast Ranges from Santa Barbara County, California, to British Columbia. The chief constituent of the parent rock is some variant of iron-magnesium silicate. Because most serpentine soils are formed in place over the parent rock, they tend to be shallow, rocky, and highly erodible. In addition, they tend to have high concentrations of magnesium, chromium, and nickel and low concentrations of calcium, nitrogen, potassium, and phosphorus (Kruckeberg 1984). These characteristics make serpentine soil inhospitable for the growth of most plants. Nevertheless, serpentine soils often support a high diversity of plants, including many rare species (McCarten 1988). Over 200 taxa in California are endemic to serpentine soils (Kruckeberg 1984).

The Metcalf Canyon jewelflower is an annual herb of the mustard family (Brassicaceae) that reaches up to 3 feet in height. This plant is found on serpentine barrens, areas of minimal soil development and extensive exposed rock that support a distinctive community of a few species, growing at low densities. Because the Metcalf Canyon jewelflower is endemic to these outcrops with little soil development, it has always been rare. It can be locally abundant but its range is limited, extending less than 20 miles from San Jose south to Anderson Lake, which lies northeast of Morgan Hill. Furthermore, the serpentine outcrops on which the subspecies occurs are patchily distributed and comprise only a small percentage of the area within its range.

The Metcalf Canyon jewelflower was federally listed as endangered on February 3, 1995. Critical habitat has not been designated for this species. The human population of the San Francisco Bay region has grown rapidly over the last several decades, and urban development has drastically reduced the amount of serpentine habitat. The increasing numbers of people also place an ever greater strain on undeveloped wildlands, through activities such as pedestrian and OHV traffic, unauthorized garbage dumping, and changes in the pattern of wildland fires. Serpentine habitats, because of their often limited vegetative cover, may appear to the uninitiated as unoccupied space, and so they are especially likely to be subject to disturbances. Recreational activities may directly impact plants; or may result in increased erosion and facilitate the invasion of alien species including many introduced annual grasses that are common in California. The destruction of serpentine habitats as a result of urban development also has increased the fragmentation of rare plant populations, thus increasing the risks of extinction due to chance events such as fire, pest or disease outbreaks, reproductive failure, or other natural or human-caused disaster.

McDonald’s Rock-cress

McDonald’s rock-cress (*Arabis mcdonaldiana*) appears to be restricted to serpentine soils in northern California and immediately adjacent southwestern Oregon. The species occurs at Red Mountain, a dome of red colored rock forming an island of peculiar vegetation protruding through the carpet of mixed evergreen forest indigenous to the Coast Ranges of northern California. The majority of rock-cress populations occupy conspicuously open habitats, scree slopes, rocky ridges, and barren rocky outcrops devoid of competing vegetation and exposed to full sun. This species appears to show long-term stability in open rocky habitats devoid of competition from other plant species. The densest populations occur in areas of north and east exposures or in sheltered saddles, which probably have the most persistent accumulations of snow. Rock-cress roots penetrate rock crevices, and areas of substantial sheet erosion appear to be poor areas of establishment. Temporarily successful at this site, McDonald’s rock-cress is likely a transitional member of this rapidly changing chaparral community (Baad 1985).

The vegetation covering the crest of Red Mountain is notably sparse, consisting of an open forest of sugar pine, ponderosa pine, Jeffrey pine and incense-cedar. An understory of chaparral species forms a patchy mosaic of dense cover alternating with extensive park-like expanses of open forest. Frequent herbaceous associates include Red Mountain buckwheat and Red Mountain stonecrop (Baad 1985). McDonald’s rock-cress is found at elevations of 3,200 to 4,100 feet.

McDonald’s rock-cress is a perennial herb whose aboveground parts remain alive year-round (Rollins 1941, 1973; Baad 1985). Germination commences with fall rains. Flowering occurs from April through June, and fruiting occurs from July through August, with dispersal from August through mid-September (Baad 1985). A number of insect visitors appear to be potential pollinators of rock-cress, including Syrphid flies, solitary bees and bumblebees. Individual plants produce a variable number of fruits, which split open in August.

McDonald’s rock-cress was federally listed as endangered on September 28, 1978. Critical habitat has not been designated. Although approximately two-thirds of the plants occur on public land, all populations are potentially endangered by plans to mine exploitable nickel and chromium deposits occurring within this area. A large-scale surface mining operation immediately adjacent to the total distribution of the species represents a serious threat to the survival of McDonald’s rock-cress.

Chorro Creek Bog Thistle
The primary reference for this section is:
USFWS. 1994b. Endangered or Threatened Status for Five Plants and the Morro Shoulderband Snail From Western San Luis Obispo County, California. Federal Register 59 (240): 64613-64623.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Ventura Field Office, Ventura, California.

Chorro Creek bog thistle (*Cirsium fontinale* var. *obispoense*) is a rugged short-lived perennial herb of the aster family that is restricted to open seep areas on serpentine soil outcrops in western San Luis Obispo County, California. The taxon has probably never been abundant because of these narrow habitat requirements. Currently, the Chorro Creek bog thistle is known from only nine locations; eight are to the south and west of San Luis Obispo, and one is 30 miles to the northwest near San Simeon.

First year plants form a rosette that reaches up to a 3.3 feet in diameter. In the second or third year, the plant produces a branching stalk up to 6.6 feet in height and bearing numerous heads of whitish to pinkish-lavender tinged flowers.

This species was federally listed as endangered on December 15, 1994. Critical habitat has not been designated. Extant populations are threatened by trampling from cattle, proposed water diversions, and road maintenance. Prolonged periods of drought conditions may also cause declines in Chorro Creek bog thistle populations. In addition, two non-native species that are invading bog thistle habitat at several sites—European broom and eucalyptus—may pose a threat to this species (Wikler and Morey 1992).

Marcescent Dudleya
The primary reference for this section is:
USFWS. 1997c. Determination of Endangered Status for Two Plants and Threatened Status for Four Plants from Southern California. Federal Register 62 (19): 4172-4183.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Ventura Field Office, Ventura, California.

Marcescent dudleya (*Dudleya cymosa* ssp. *marcescens*) occurs in the Los Angeles Basin, on the lower slopes of volcanic cliffs in canyons that have perennial moisture. This plant is known from seven occurrences in the Santa Monica Mountains, from Hidden Valley to Malibu Creek State Park, a distance of 15 miles. In 1997, estimates of the number of individuals at each occurrence were between 50 and 200 plants; the total number of individuals was estimated to be less than 1,000. This subspecies can be found on sheer volcanic rock surfaces and canyon walls adjacent to perennial streams. In most locations, the topographic relief has precluded soil formation; therefore, this taxon may be the only vascular plant in a microhabitat otherwise dominated by mosses and lichens (California Natural Diversity Database 1994).

Marcescent dudleya was federally listed as endangered on January 29, 1997. Critical habitat has not been designated. Half of the populations of the subspecies occur on lands administered by the California Department of Parks and Recreation; two locations are administered by the National Park Service—one on an administrative easement where the landowner has drastically altered the native vegetation (pine plantings in a cleared oak grove), and another in an area that receives unsupervised recreational use (boulder hopping and rock climbing). The remaining populations are on lands in private ownership, several of which are threatened by development (California Natural Diversity Database 1994, Skinner and Pavlik 1994). On the California Department of Parks and Recreation and National Park Service lands, the plant is threatened by recreational use, particularly rock climbing, foot traffic, collection, and fire.

Marine Ecoregion Division

The Marine Ecoregion Division includes habitats in the maritime climate of the Cascade and Coast Ranges of western Washington and Oregon along the Pacific Coast. The vegetation in this ecoregion is predominantly coniferous and mixed forests, with non-forested meadows and grasslands occurring over a small area. The TEP plant species found in this ecoregion typically occur in these rarer open habitats, many of which are threatened by forest succession and the encroachment of woody plants.

Bradshaw's Lomatium
The primary reference for this section is:
USFWS. 1993g. *Lomatium bradshawii* (Bradshaw's Lomatium) Recovery Plan. Portland, Oregon.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

Bradshaw's lomatium, or Bradshaw's desert-parsley (*Lomatium bradshawii*) is endemic to the central and southern portions of the Willamette Valley, in western Oregon. It is known from Marion, Linn, Benton, and Lane counties. The majority of the sites and plants occur in and adjacent to the Eugene metropolitan area, with the greatest concentration found in West Eugene. Bradshaw's lomatium occurs in two very distinct habitats. The rarest are the shallow, stream-covered basalt areas found in Marion and Linn counties neat the Santiam River. At these sites, the plants occur in areas with almost no soil, usually in vernal wetlands or along stream channels. The majority of the species' populations occur on seasonally saturated or flooded prairies, which are common by creeks and small rivers in the southern Willamette Valley. They occur in areas with deep, pluvial clays, usually in a matrix with alluvial silts. The slowly permeable clay layer results in a perched water table in winter and spring, so soils are generally saturated to the surface or slightly inundated during the wet season.

This relic wetland prairie has been described as the tufted hairgrass valley prairie, which ranges from fairly wet areas with high sedge and rush cover, to drier bunchgrass prairie. In the wet areas, Bradshaw's lomatium occurs on the edges of tufted-hairgrass or sedge bunches, in patches of bare or open soil. In the drier areas, it is found in low areas, such as small depressions, trails, or seasonal channels, also with open, exposed soils.

Bradshaw's lomatium reproduces entirely by seeds, which are produced on umbels. Flowers are visited by numerous pollinators, and require insects for pollination. The species blooms fairly early in the spring, usually in April or early May. In the Willamette Valley, these are often wet, rainy weeks, when large bees and butterflies are largely absent. The very general nature of the insect pollinators probably buffers the species from population swings of any one pollinator (Kaye 1992). A typical population of Bradshaw's lomatium is composed of many more vegetative plants than reproductive plants. In general, populations that have experienced prescribed fire have a higher probability of survival.

Bradshaw's lomatium was federally listed as endangered on September 30, 1988. Critical habitat has not been designated. The species' habitat is presently being destroyed or modified by a number of factors: invasion of prairie vegetation by trees and shrubs; changes in flooding patterns and water movement (which may be critical to seedling establishment); urban development; and agricultural or rural development. In addition, disease caused by a fungal parasite, and insect predation of plants and fruit may threaten smaller population. Finally, natural factors such as inbreeding depression or limited pollinator availability may reduce fecundity, and therefore reproductive capacity of the species.

Willamette Daisy
The primary reference for this section is:
USFWS. 1997f. Endangered Status for *Erigeron decumbens* var. *decumbens* (Willamette Daisy) and Fender's Blue Butterfly (*Icaricia icarioides fenderi*) and Threatened Status for *Lupinus sulphureus* ssp. *kincaidii* (Kincaid's Lupine). Federal Register 65 (16): 3875-3890.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Oregon State Office, Portland, Oregon.

The Willamette daisy (*Erigeron decumbens* var. *decumbens*) is restricted primarily to the Willamette Valley of Oregon. The valley is an alluvial floodplain that is 130 miles long and 20 to 40 miles wide, with an overall northward gradient (Orr et al. 1992). The valley is narrow and flat at its southern end, widening and becoming hilly near its northern end at the confluence of the Willamette and Columbia rivers. The alluvial soils of the Willamette Valley and southern Washington host a mosaic of grassland, woodland, and forest communities. The Willamette daisy occupies native grassland habitats within the Willamette Valley. The vast majority of Willamette Valley grasslands require natural or human-induced disturbance for their maintenance (Franklin and Dryness 1973), and would likely be forested if left undisturbed (Johannessen et al. 1971).

The Kalapooya Indians cleared and burned lands in the Willamette Valley used for hunting and food gathering. Accounts by early explorers suggest a pattern of annual burning by the Kalapooya Indian tribe resulted in the maintenance of extensive wet and dry prairie grasslands (Johannessen et al. 1971). Although much of the woody vegetation was prevented from becoming established on the grasslands by this treatment, the random survival of young fire-resistant species such as Oregon white oak, accounted for the widely-spaced trees on the margins of the valley (Habeck 1961). After 1848, burning decreased sharply through the efforts of settlers to suppress large-scale fires. Consequently, the open, park-like nature of the valley floor was lost, replaced by agricultural fields, dense oak and fir forests, and scrublands following logging.

The primary habitat for the Willamette daisy is native wetland prairie. This habitat is characterized by the seasonally wet tufted hairgrass community that occurs in low, flat regions of the Willamette Valley where flooding creates anaerobic and strongly reducing soil conditions. This wet prairie community includes rushes and California oatgrass as co-dominant native species, as well as the introduced species tall fescue, Japanese brome and sweet vernal grass (USFWS 1993). Another endangered species, Bradshaw’s lomatium also grows in wet prairie habitat.

The Willamette daisy is a perennial herb, 0.6 to 2.4 inches tall, with erect to sometimes prostrate stems at the base. As with many species in the Aster family, the Willamette daisy produces large quantities of wind-dispersed seeds. Flowering typically occurs in June and July with pollination carried out by flies and bees. Seeds are released in July and August. Although the seeds are wind-dispersed, the short stature of this species likely prevents the long-distance travel of many of these seeds. The Willamette daisy is capable of vegetative spreading and is commonly found in large clumps scattered throughout a site (Clark et al. 1993).

The Willamette daisy was federally listed as endangered on January 25, 2000. At the time of listing, the USFWS indicated that designation of critical habitat was prudent, but that it would be deferred until resources became available to do so. The Willamette daisy likely once occurred over a large distribution throughout the historic native prairie. However, native prairie vegetation in the Willamette Valley was decimated by the rapid expansion of agriculture from the 1850s to the present. In addition, fire suppression allowed shrub and tree species to overtake grasslands, while agricultural practices hastened the decline of native prairie species through habitat loss and increased grazing (Johannessen et al. 1971; Franklin and Dyrness 1973). Currently, the species is threatened by commercial and/or residential development, agriculture, silvicultural practices, road improvement, collection, herbicide use, and naturally occurring demographic and random environmental events.

Kincaid’s Lupine

The primary reference for this section is:

USFWS. 1997f. Endangered Status for *Erigeron decumbens* var. *decumbens* (Willamette Daisy) and Fender’s Blue Butterfly (*Icaricia icarioides fenderi*) and Threatened Status for *Lupinus sulphureus* ssp. *kincaidii* (Kincaid’s Lupine). Federal Register 65 (16): 3875-3890.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Oregon State Office, Portland, Oregon.

Kincaid's lupine (*Lupinus sulphureus* ssp. *kincaidii*), like the Willamette daisy described above, is restricted primarily to the Willamette Valley of Oregon, where it occupies native grassland habitats within the Willamette Valley. The vast majority of Willamette Valley grasslands require natural or human-induced disturbance for their maintenance (Franklin and Dryness 1973), and would likely be forested if left undisturbed (Johannessen et al. 1971).

Kincaid's lupine is typically found in native upland prairie with red fescue and/or Idaho fescues, the dominant species, and Tolmie's mariposa, Hooker's catchfly, broadpetal strawberry, rose checker-mallow, and lomatium species serving as herbaceous indicator species (Hammond and Wilson 1993). At the time of listing in 1997, there were four known occurrences of Kincaid's lupine approximately 38 miles south of the Willamette Valley and within the Umpqua Valley of Douglas County, Oregon. In addition to its Oregon occurrences, this species is known from two small sites in Lewis County, southern Washington, 40 miles north of the Willamette Valley.

Kincaid's lupine is a long-lived perennial species, with a maximum reported age of 25 years (Wilson 1993). Individual plants are capable of spreading by rhizomes, producing clumps of plants exceeding 66 feet in diameter (Hammond 1994). The long rhizomes do not produce adventitious roots (secondary roots growing from stem tissue) and apparently do not separate from the parent clump, and the clumps may be short-lived, regularly dying back to the crown (Kuykendall and Kaye 1993a). Kincaid's lupine is pollinated by solitary bees and flies (Hammond 1994). Seed set and seed production are low, with few (but variable) numbers of flowers producing fruit from year to year, and each fruit containing an average of 0.3 to 1.8 seeds (Liston et al. 1994). Seeds are dispersed from fruits that open explosively upon drying. Kincaid's lupine is the host plant of the federally endangered Fender's blue butterfly.

Kincaid's lupine was federally listed as threatened on January 25, 2000. At the time of listing, the USFWS indicated that designation of critical habitat was prudent, but that it would be deferred until resources became available to do so. Kincaid's lupine likely once occurred over a large distribution throughout the historic native prairie. However, native prairie vegetation in the Willamette Valley was decimated by the rapid expansion of agriculture from the 1850s to the present. In addition, fire suppression allowed shrub and tree species to overtake grasslands, while agricultural practices hastened the decline of native prairie species through habitat loss and increased grazing (Johannessen et al. 1971; Franklin and Dyrness 1973). Currently, Kincaid's lupine is threatened by commercial and/or residential development, agriculture, silvicultural practices, road improvement, collection, herbicide use, and naturally occurring demographic and random environmental events.

Nelson's Checker-mallow
The primary reference for this section is:
USFWS. 1998j. Recovery Plan for the Threatened Nelson's Checker-mallow (*Sidalcea nelsoniana*). Portland, Oregon.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

Nelson's checker-mallow (*Sidalcea nelsoniana*) occurs as scattered populations in two distinct ecological regions: the northern Coast Range and the Willamette Valley of Oregon (includes two outlying populations in the Puget Trough of Washington). The species is not restricted to a single habitat type. Rather, it occupies a broad range of soils that vary in texture, drainage, and disturbance regimes (CH2M Hill 1986b). Plants appear to favor primary drainages, or those that receive mostly ground flow of stormwater runoff, rather than drainages fed by stream sources.

Although occasionally occurring in the understory of woodlands or among woody shrubs, populations of Nelson's checker-mallow in the Willamette Valley usually occupy open habitats that support early successional species (i.e., plants that colonize openings and then disappear as trees shade them out). These habitats are frequently represented by margins of sloughs, ditches, and streams, roadsides, fence rows, drainage swales, native prairie remnants, and fallow fields. Most sites have been densely colonized by invasive weeds, especially introduced forage grasses.

Commonly associated plant species include: tall fescue, rose, common rush, Canada thistle, common St. Johnswort, blackberry, sedge, timothy, velvet grass, yarrow, vetch, western spiraea, bird's-foot trefoil, ox-eye daisy, colonial bent-grass, meadow foxtail, reed canary-grass, Douglas' hawthorn, wild carrot, large-leaved avens, geranium, and Oregon ash (Oregon Department of Agriculture 1995).

Populations of Nelson's checker-mallow in the Coast Range generally occur in open, wet-to-dry meadows, intermittent stream channels, and along the margins of coniferous forests. These areas typically support larger components of native vegetation than the Willamette Valley sites. Commonly associated plant species include tansy ragwort, spear-head senecio, strawberry, velvet grass, timothy, rush, sedge, and yarrow.

Nelson's checker-mallow is an herbaceous perennial plant species in the mallow family. In the Willamette Valley, flowering begins as early as mid-May, and continues through August to early September, depending on the moisture and climatic conditions of each site. Coast Range populations experience a shorter growing season and generally flower later and go dormant earlier. Seeds are deposited locally at or near the base of the parent plant, and may be shed immediately or persist into winter within the dry flower parts that remain attached to the dead stems. Seed dissemination could conceivably be accomplished through ingestion by deer and elk, particularly in the Coast Range. Aboveground portions of the plant die back in the fall, usually followed by some degree of re-greening at the base. It is not uncommon for some plants to continue producing flowers into the fall and early winter. Sexual reproduction appears to be accomplished entirely by insect pollinators.

Nelson's checker-mallow was federally listed as threatened on February 12, 1993. Critical habitat has not been designated. Prior to European settlement, Nelson's checker-mallow habitats were likely maintained and kept free of overgrowth and woody vegetation by natural wildfires, fires set by Native Americans (Johannessen et al. 1971; Franklin and Dyrness 1973; Boyd 1986), and sporadic flooding. The landscape and processes such as flooding and fire have been dramatically altered since the onset of European settlement. Today, no natural prairie remains in the Willamette Valley without evidence of livestock grazing, agriculture, and fire suppression (Moir and Mika 1972). Urbanization and conversion of the native prairies into intensively managed croplands and pastures have eliminated and fragmented grasslands to the extent that Nelson's checker-mallow is now restricted to sparsely distributed patches within narrow highway and country road ROW, undeveloped tracts, ditches, fence rows, abandoned fields, parks, and wildlife refuges. Populations in the Willamette Valley are threatened by roadside maintenance, herbicide application and mowing, soil cultivation, ditching, and other habitat modifications.

Land threats are less extreme in the Coast Range, where the meadows occupied by Nelson's checker-mallow are isolated from agricultural and urban development. Potential threats to these populations include a planned water impoundment project, herbicide application associated with timber harvest, and motorcyclists. Other threats to the species as a whole are competition with invasive plant species, the encroachment of trees and shrubs, limited seed production, and the species' small population size and fragmentation.

Wenatchee Mountains Checker-mallow
The primary references for this section are:
USFWS. 1999g. Determination of Endangered Status for *Sidalcea oregana* var. *calva* (Wenatchee Mountains Checker-mallow). Federal Register 64 (245): 71680-71687;

and

USFWS. 2001f. Final Designation of Critical Habitat for *Sidalcea oregana* var. *calva* (Wenatchee Mountains Checker-mallow). Federal Register 66 (173): 46536-46548.

The Wenatchee Mountains checker-mallow (*Sidalcea oregana* var. *calva*) is endemic to the Wenatchee Mountains of Chelan County in central Washington. The plant is most abundant in moist meadows that have surface water or saturated upper soil profiles during spring and early summer, but it also occurs in open conifer stands dominated by ponderosa pine and Douglas-fir and on the margins of shrub and hardwood thickets. Populations are found at elevations ranging from 1,600 to 3,300 feet. The soils are typically clay loams and silty loams with low moisture permeability. Associated species include quaking aspen, black hawthorn, common snowberry, serviceberry, few-flowered peavine, northern mule's-ear, sticky purple geranium and California false hellebore.

The Wenatchee Mountains checker-mallow is a perennial plant with a stout taproot that branches at the root-crown and gives rise to several stems that are 8 to 60 inches tall. Flowering begins in the middle of June and peaks in the middle to end of July. At the time of listing in 1999, the taxon was known to occur at six sites (populations), three of which had very few individuals. The estimated total number of plants was about 3,600.

The physical and biological habitat features essential to the conservation of this species include open meadows with surface water or saturated upper soil profiles in the spring and early summer, and the hydrologic processes on which these areas depend; open conifer forests dominated by ponderosa pine and Douglas-fir; and the margins of shrub and hardwood thickets. All of these habitats have surface water or saturated soils well into the early summer. The species is generally found on flats or benches, but may also occur in small ravines and occasionally on gently sloping uplands.

The Wenatchee Mountains checker-mallow was federally listed as endangered on December 22, 1999. Approximately 6,135 acres in Chelan County, Washington, were designated as critical habitat on September 6, 2001. The primary threats to the species include habitat fragmentation and destruction caused by alteration of hydrology, rural residential development and associated impacts, conversion of native wetlands to orchards and other agricultural uses, competition from native and non-native plants, recreation, seed and plant collection, and fire suppression and associated activities. To a lesser extent, the species is threatened by livestock grazing, road construction, and timber harvesting and associated impacts, including changes in surface runoff in the small watersheds in which the plant occurs.

Applegate's Milk-vetch

The primary reference for this section is:
Hudson, B., J. Augsburger, M. Hillis, and P. Boehne. 2000. Draft Biological Assessment for the Interior Columbia River Basin Ecosystem Management Project Final Environmental Impact Statement. BLM and Forest Service. Boise, Idaho.

Other references used are cited in the text and included in the Bibliography.

Applegate's milk-vetch (*Astragalus applegatei*) is a narrow endemic, known only from the Lower Klamath Basin near the city of Klamath Falls in southern Oregon. It is restricted to flat-lying, seasonally moist, strongly alkaline soils (USFWS 1997g). Although it is currently replete with introduced grasses and other weeds, the species' habitat was historically characterized by sparse, native bunchgrasses and patches of bare soil. Currently, there are two known populations of the species, which occur over a total area of less than 10 acres, and which form a total metapopulation of fewer than 20,000 individuals. Of the two populations, one is on land leased by The Nature Conservancy and one is on state land. There are no populations on federal lands.

Applegate's milk-vetch appears to be dependent on the seasonal flooding that occurs at sites where it is found, which may limit the dominance of other species and create favorable openings for the establishment of new plants. Applegate's milk vetch hosts an unknown species of beetle larvae, and is pollinated by ground-nesting beetles.

Applegate's milk-vetch was federally-listed as endangered on July 28, 1993. Critical habitat has not been designated. The primary threats to this species include invasion of habitat by exotic species such as quackgrass and downy brome, urban development, and road construction. Low population numbers, loss of habitat, wildlife grazing (rabbits), and management controls that alter natural wildfire and flooding regimes all pose serious threats to this species.

Rough Popcornflower

The primary reference for this species is:

USFWS. 2000f. Endangered Status for the Plant *Plagiobothrys hirtus* (Rough Popcornflower). Federal Register 65(16): 3866-3875.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Oregon State Office, Portland, Oregon.

The rough popcornflower (*Plagiobothrys hirtus*) is endemic to seasonal wetlands (e.g., wet swales and meadows) of the interior valley of the Umpqua River in southwestern Oregon (Amsberry and Meinke 1997b). The plant grows at elevations ranging from 98 to 886 feet, in open microsites within interior valley grasslands. Common associates include one-sided sedge, meadow barley, tufted hairgrass, American slough grass, great camas, water foxtail, baltic rush, wild mint, Willamette downingia, and bentgrass (Gamon and Kagan 1985).

The rough popcornflower is an annual herb on drier sites or a perennial herb on wetter sites (Amsberry and Meinke 1997a). It grows in scattered groups and reproduces largely by insect-aided cross-pollination and partially by self-pollination. The taxon is considered dependent on seasonal flooding and/or fire to maintain open habitat and to limit competition with invasive native and non-native plant species, such as Himalayan blackberry, Oregon ash, teasel, and pennyroyal (Gamon and Kagan 1985, Almasi and Borgias 1996).

A total of 17 habitat patches exist for this species, all of which are located in Douglas County, in the vicinity of Sutherlin and Yoncalla, Oregon. Most populations are small with few individuals. The total estimated number of plants is about 7,000 individuals within a combined area of about 45 acres. Fifteen of the 17 occupied habitat patches occur on private or commercial land. Three of these parcels are owned and managed by The Nature Conservancy. The other 12 have no protective management for the species and are at risk of extirpation from development, incompatible grazing and farming practices, and recreational activities (Kagan 1997, Meinke 1997). The two remaining known sites occur on public land owned by the Oregon Department of Transportation, with a portion of one site partially occurring on private land as well.

The rough popcornflower was federally listed as endangered on January 25, 2000. Critical habitat has not yet been designated for this species. Draining of wetlands for urban and agricultural uses and road and reservoir construction, however, has altered the original hydrology of the valley to such an extent that the total area of suitable habitat for the species has been substantially reduced. In addition to the ongoing threat of direct loss of habitat from conversion to urban and agricultural uses, hydrological alterations, and fire suppression, other threats to the species include spring and summer livestock grazing, roadside mowing, spraying, competition with non-native vegetation, and landscaping (Gamon and Kagan 1985, Kagan 1995).

Showy Stickseed

The primary reference for this species is:

USFWS. 2002e. Determination of Endangered Status for the Washington Plant *Hackelia venusta* (Showy Stickseed). Federal Register 67(25): 5515-5525.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Western Washington Fish and Wildlife Office, Lacey, Washington.

The showy stickseed (*Hackelia venusta*) is a narrow endemic restricted to less than 1 acre of unstable talus, on the lower slopes of Tumwater Canyon, Chelan County, Washington. The species is shade-intolerant (Carr 1998) and grows in openings within ponderosa pine and Douglas-fir forest types. Showy stickseed plants are found on open, steep slopes (minimum of 80% inclination) of loose, well-drained, granitic weathered and broken rock fragmented soils at an elevation of about 1,600 feet. There is currently only one small population of approximately 500 plants, which occurs on land in the Wenatchee National Forest, in an area designated as the Tumwater Botanical Area.

The showy stickseed is a perennial herb of the Borage family. It has large, showy flowers, and its fruit is a nutlet. As the common name suggests, seeds are dispersed by clinging to passing animals. The fruits of the showy stickseed are spurred and covered with stout hairs that cling to the hair and bodies of animals.

The showy stickseed was federally listed as endangered on February 6, 2002. Critical habitat has not been designated for this species. Major threats to the showy stickseed include collection, physical disturbance to the plants and their habitat by humans, competition and shading from native trees and shrubs, encroachment onto the site by non-native noxious weed species, wildfire, fire suppression and associated activities, and low seedling establishment. Highway maintenance activities, such as the spreading of salt and the use of de-icers during the winter months also threaten the species. Application of herbicides may also pose a threat. In addition, reproductive vigor may be depressed because of the plant's small population size and limited gene pool. A single natural or human-caused random environmental disturbance (such as wildfire), could destroy a large percentage of the population.

Marsh Sandwort
The primary reference for this species is:
USFWS. 1998k. Recovery Plan for Marsh Sandwort (*Arenaria paludicola*) and Gambel's Watercress (*Rorippa gambelii*). Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Marsh sandwort (*Arenaria paludicola*) was historically found in scattered locations near the Pacific coast in southern and central California and Washington. The species occurs in freshwater marshes at elevations from sea level to 1,480 feet. Soils in these habitats are saturated, acidic bog soils, that are predominantly sandy and have a high organic content. Presently, there are only two known populations of this species in the United States, both in San Luis Obispo County, California: one of fewer than 10 individuals in Black Lake Canyon, and one of more than 85 individuals at Oso Flaco Lake. The Marsh sandwort has been listed by the Washington Natural Heritage Program as "possibly extirpated" in Washington State. Nonetheless, it is thought that suitable habitat for the species remains in Washington State, and that populations could exist there now or in the future. As this species occurs on the BLM's Washington/Oregon special status species list, but not on the California list, it is unlikely that this species presently occurs on public lands.

Because there are so few individuals of the Marsh sandwort remaining, studying the life history of this species has been difficult. Although plants have been observed flowering and fruiting minimally, and a viable seed bank has been identified, information about the species' pollinators, seed germination and dispersal, and seedling recruitment is lacking.

The Marsh sandwort was federally listed as endangered on August 3, 1993. Critical habitat has not been designated. Threats to the species include encroaching vegetation (both native and non-native) associated with lowered water tables, agricultural and residential development, and OHV use. In addition, the very low number of individuals in the remaining populations puts this species at a great risk of extinction as a result of random, naturally occurring events.

Tundra Ecoregion Division

The Tundra Ecoregion Division includes the northern Continental fringes of North America, where the climate is controlled by arctic air masses. In the U.S., portions of Alaska are included in this ecoregion, which supports vegetation adapted to short, cool summers and long, severe winters. Vegetation in the Tundra Ecoregion Division predominantly consists of grasses, sedges, lichens, and willow shrubs (Bailey 1995). There is only one TEP plant species located in the Tundra ecoregion: the Aleutian shield fern.

Aleutian Shield Fern
The primary reference for this section is:
USFWS. 1992g. Aleutian Shield Fern Recovery Plan. USFWS. Anchorage, Alaska.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The Aleutian shield fern (*Polystichum aleuticum*) is a narrow endemic species that is only known from Atka and Adak islands in the Aleutian Islands of Alaska. At present, it is only known to occur on Adak Island. The species grows in vegetation mats and sod mats on exposed, weathered rock outcrops of Mount Reed (Tande 1989). Associated vegetation includes dwarf willows, sedges, moss, anemone, and arnica. The climate during the growing season is relatively mild, with dense fog blankets common during the summer months.

The fronds (i.e., the aboveground vegetative portion) of this species are present only during the growing season. Spores for reproduction are borne in two rows of sori (masses of spores) along the under-surface of the fronds.

The Aleutian shield fern was federally listed as endangered on February 17, 1988. Critical habitat has not been designated for this species. The factors that contribute to the Aleutian shield fern's rarity are not well known. Given the remote location of the species, human disturbance is minimal. Possible threats to the species include introduced ungulates and natural soil movement and seismic events at the site.

Species in Multiple Ecoregions

Two TEP species addressed by this BA have a large geographic distribution and therefore do not fit into one primary ecoregion category. Both water howellia and Ute ladies'-tresses are wetland species that appear to be more dependent on hydrology and general habitat features than on regional climate.

Water Howellia
The primary reference for this section is:
Hudson, B., J. Augsburger, M. Hillis, and P. Boehne. 2000. Draft Biological Assessment for the Interior Columbia River Basin Ecosystem Management Project Final Environmental Impact Statement. BLM and Forest Service. Boise, Idaho.

Water howellia (*Howellia aquatilis*) is an annual aquatic plant with a scattered distribution in the Pacific Northwest. The species is known to be extant in Idaho, Montana, and Washington, but is also historically known from California and Oregon. Sites in California and Oregon have not been recently relocated, despite intensive field surveys in both states. Within its current range, water howellia is known from a total of 110 occurrences. There are two main centers of distribution within this range: one in the Swan River Valley in Montana, and one in the vicinity of Spokane, Washington. Populations of water howellia in these centers range from one to 1,000 plants, and occur mostly on publicly-owned land, and at elevations of 400 to 2,320 feet. Two occurrences are known in northern Idaho, in private ownership, and two others are found in western Washington. The total known occupied habitat for this species is less than 100 acres.

Water howellia is restricted to small pothole ponds or the quiet water of shallow, abandoned river oxbows. These wetland habitats typically occur in a matrix of dense forest vegetation, and all known sites have at least some deciduous tree cover around a portion of the pond. Ponderosa pine forests typically surround the ponds, and red-osier dogwood is usually present around the perimeters. The bottom surfaces of the wetlands consist of firm, consolidated clay and organic sediments. These wetlands are generally filled by snowmelt runoff and spring rains, but then dry out to varying degrees by late summer or early fall, depending on annual patterns of temperature and precipitation. The ponds are typically shallow, averaging 1 to 2 feet in depth during the middle of summer.

The bloom period of water howellia varies by geographic location, but typically occurs in May and June. The drying of the wetland habitat in late summer is critical to the species' life cycle; the seeds will only germinate if they are exposed to the atmosphere. After the seedlings appear, usually in October, they overwinter under the snowpack. In late spring and early summer, the plants resume growth in the water that accumulates in the ponds. This ecological relationship has a profound influence on the size of occurrences from year to year; the summer climate determines the degree of pond drying, and thus the amount of seed germination in the fall. During years when seed germination is reduced, few plants are present the following summer.

Water howellia was listed as threatened on July 14, 1994, but critical habitat was not designated. The highly specialized ecological adaptations of the species make it vulnerable to both short- and long-term natural environmental changes, such as succession or climate change. Land management activities and habitat destruction have also affected this species. Development, construction of dams, livestock grazing and trampling, timber harvesting, and road building are some of the human activities that alter the habitat of this species. Competition with introduced plant species, such as reed canarygrass and purple loosestrife, is also a threat.

Ute Ladies'-tresses

The primary reference for this section is:
USFWS. 1992h. Final Rule to List the Plant *Spiranthes diluvialis* (Ute Ladies'-tresses) as a Threatened Species. Federal Register 57(12): 2048-2054.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

Ute ladies'-tresses (*Spiranthes diluvialis*) is endemic to moist soils in mesic or wet meadows near springs, lakes, or perennial streams. The species occurs primarily in areas where the vegetation is relatively open and not overly dense, overgrown, or overgrazed (Coyner 1989, 1990; Jennings 1989, 1990). At the time of its listing in 1992, populations of the species were only known to occur in riparian meadows in three geographic areas: near Boulder Creek in Colorado, in the Colorado River drainage of eastern Utah, and in the eastern Great Basin of Western Utah and adjacent Nevada. Since that time, additional populations have been found, and the species is now known to occur in Colorado, Idaho, Montana, Nebraska, Nevada, Utah, Washington, and Wyoming.

Ute ladies'-tresses is a perennial orchid with stems that arise from thickened roots. The bloom consists of 3 to 15 small white or ivory colored flowers clustered into a spike arrangement at the top of the stem. Depending on location, the species may flower as early as early July or as late as early October (Sheviak 1984, Jennings 1989, Coyner 1990). Mature plants may remain dormant for 1 or more growing seasons without producing aboveground shoots, or may exhibit vegetative shoots only. Bumblebees are apparently required for pollination.

Ute ladies'-tresses was federally listed as threatened on January 17, 1992. Critical habitat has not been designated for this species. The species is threatened primarily by habitat loss and modification, though its small populations and low reproductive rate also make it vulnerable to other threats. Urban development and watershed alterations in riparian and wetland habitat adversely affect this species. Exotic plant species, such as purple loosestrife, whitetop, and reed canarygrass may also impact populations of Ute ladies'-tresses. Other potential threats include grazing during periods of flowering or fruiting, and recreational use.

Effects of Vegetation Treatments on Plants

For this analysis, all TEP plant species or their critical habitat located within the project area are considered as a whole. Although the plant species listed in Table 1-1 occupy a wide range of habitat types and account for a wide range of life forms, considering them as a single group is suitable for a programmatic analysis. In general, vegetation treatments have the potential to affect most plant species in much the same way: all are intended to cause mortality or injury to target plants, and may vary in intensity and extent.

All other aspects of analysis being equal, species present in low numbers or that have a limited distribution are the most sensitive to impacts. Information about population size and distribution has been provided in the background section, and should be referred to, as appropriate.

Prescribed Fire Treatments

The potential effects of prescribed fire on TEP plant species would vary depending on a number of factors. The timing of the burn; the area, frequency, and intensity of the burn; the level of resistance or adaptation by individual species to fire; the presence of fire-adapted weeds; and the historical fire disturbance regime of the habitat will all influence the effects of prescribed burning. The ability of a particular plant species to recover after a burn is another important factor to consider.

Direct Effects

Potential direct effects to TEP plant species from fire include mortality and injury caused by burning of plant tissues and crushing caused by equipment used during fire-related activities, and reduced reproductive success caused by damaging the seedbed.

Fire can kill plant tissue, with the greatest damage caused by the hottest fires (Brown and Smith 2000). Direct damage to plants is also dependent on time, with lower temperature fires requiring a longer exposure to plants to cause mortality. Each species of plant has a biologically-based level of susceptibility to fire, with structures such as bark or bud scales providing some amount of protection. In addition, the season in which the burn is conducted can affect the response of plants to fire. Plants that are burned during a period of low carbohydrate storage (such as right after periods of high growth, flowering, or fruiting), may lack the energy reserves required for regrowth after the fire, increasing the duration or permanence of the mortality or injury. In terms of weather, fuel consumption and the spread of fire may be limited during a wet season, causing minimal mortality to plant tissues. During a dry season, however, and especially in a drought, a much higher percentage of the vegetation on a site is likely to be scorched or consumed, with injury to belowground plant parts as well.

In addition to direct harm to plant tissues, fire can severely damage the seedbank, reducing the ability of the plant to recover after a burn. Plants with small seedbanks that are burned before seed dispersal would be expected to have reduced reproductive output in subsequent growing seasons. In all cases, the severity of the fire would influence the amount of damage sustained by TEP plants and their seedbanks. Fires over a large area would also be much more likely to irreparably affect a rare species population than a fire occurring over just a small portion of the habitat.

As a general rule, populations of annual plant species are more capable of tolerating or recovering from a fire than perennial species. Annuals live for a single year, relying on seeds to germinate the following growing season for their persistence. Many annuals produce large quantities of seed, which over time results in a large seedbank. Therefore, populations of annual TEP species would be expected to reappear following a burn, provided that the fire did not damage their seedbank or make the habitat unsuitable for the germination of seeds. The life history strategy of the TEP species considered in this BA are provided in Table 4-1 (under the heading “Life Form”) as a general guideline. Prior to burns, local BLM offices would need to determine the specific degree of risk to these species.

Perennial species, unlike annuals, often require multiple years of growth prior to setting seed. Some species, such as a number of the desert plants considered in this BA, are extremely long-lived species with a low level of reproductive success. These species are adapted to life in harsh environments, where resources are scarce, competition is minimal, and survival is difficult. Therefore, established plants are extremely important for the persistence of the population. The adverse effects resulting from direct mortality or an injury caused by a prescribed burn would likely be severe, and populations would not be expected to recover. In less harsh environments, the effect of burning would be expected to be less severe, depending on multiple factors. Some perennial species have a taproot or woody root, and can resprout vegetatively after a fire. The information in Table

4-1 provides information on the life form, stature, and root type of the TEP species as a general guideline. However, final decisions as to the degree of risks to these species should be made at the local level.

Because the severity of a fire can range from a low intensity understory burn, to a high intensity stand-replacing fire (Brown and Smith 2000), it is reasonable to assume that high intensity fires have a greater potential to adversely affect listed species than fires of lower intensity. Higher intensity fires are most likely to occur in areas where fuel loading has increased beyond the natural range of variability as a result of human fire suppression activities. There is also a greater likelihood for some impacts resulting from high intensity fires to be sustained over a longer time frame than those associated with fires of medium or low intensity. Damage may be severe enough to be considered permanent, or to preclude reoccupation by a species for some time.

Because many TEP species have extremely small populations and/or limited distributions, they could potentially be extirpated if a fire were to burn through habitat.

There are few direct beneficial effects to TEP plant species resulting from prescribed burns. However, low intensity burns that do not cause substantial injury to plant tissues can increase reproductive success during the growing season by increasing flower production. In addition, the seeds of some species require fire to germinate, particularly in chaparral habitats (e.g., Ione manzanita, Stebbins' morning-glory, Pine Hill ceanothus, Pine Hill flannelbush, Layne's butterweed, and Braunton's milk-vetch).

Indirect Effects

Potential indirect effects to TEP plant species resulting from prescribed burns include habitat alteration, an increase or decrease in competitors, and indirect plant mortality.

For many TEP species, the effects of fire on habitat can have long-term benefits. Fire is often beneficial to early successional, disturbance-dependent species that are poor competitors and require open habitats to persist. Fire can increase soil temperature, remove canopy cover and increase the light available to understory species, and increase the availability of soil moisture and soluble nutrients. A prescribed fire program that adequately mimics the historic disturbance regime under which TEP plant species evolved would likely create more hospitable conditions for these species by exposing mineral soil, creating openings in the canopy, and killing later-successional competitors. Numerous TEP species considered in this BA, particularly species found in the Marine Ecoregion, are early-successional species that would be expected to benefit indirectly from prescribed burns. In Table 4-1, the assumed response to fire for these species —"may benefit"—is intended as a general guide. At the local level, determinations would have to be made as to the ability of the population to recover from exposure to fire, and the appropriate time of the year in which to conduct burns. In some cases, populations of TEP plants would need to be protected from fire while the surrounding habitat was burned.

Depending on the ecosystem type and whether it has been substantially altered by fire suppression, prescribed burns away from known populations in critical habitat, or other suitable habitat adjacent to existing rare plant populations, may increase the amount of suitable habitat (e.g., by opening up ponderosa pine forests or oak woodlands; preventing the encroachment of shrubs and woody species in grasslands; and controlling non-native species in and near vernal pools).

Although the removal of competitors such as late successional or fire-intolerant species would be expected to improve habitat for TEP species, prescribed fire can also negatively alter the species composition on a site. In many areas throughout the western United States, non-natives have altered ecosystems so drastically that invasive species will outcompete natives, including TEP plants, after fire in occupying sites that are cleared by burning. In some areas (rangelands, notably), an increase in fire-adapted weeds following a prescribed burn further degrades the quality of the habitat. In addition, because many non-native annuals dry out earlier than native perennials, there is a longer annual flammable period (Hann et al. 2002). Furthermore, the proliferation of some non-native species has increased the density of ground cover to such an extent that a subsequent prescribed burn will burn much hotter than under native conditions. Downy brome and tamarisk are examples of two species whose invasions have increased the frequency of unwanted, damaging wildland fires. Burn treatments followed by reseeding native species to preclude the spread of non-natives on the site would be expected to result in fewer adverse effects to the habitats of TEP plant species.

Over the long-term, prescribed fire would benefit TEP plant species by reducing the risks of a future large-scale wildfire through the reduction of fuel build-up. A naturally-occurring (or human-caused fire) in an area where fires have been suppressed for many years would be expected to burn hotter, and over a larger area than a controlled fire. Such a fire would have an even greater impact on TEP species and their seed banks than a prescribed fire. In addition, activities associated with emergency fire suppression, such as creation of emergency firelines, can harm TEP populations. BLM-prescribed fires would follow guidance and management practices detailed in the *National Fire Plan* (USDI and USDA 1995) to ensure that their intensity and extent would be far less damaging than those of an unmanaged wildfire.

Other indirect effects to plants from prescribed fire include eventual plant mortality caused by post-fire disease, fungus, insects, or drought; soil erosion caused by the removal of vegetation from a site; and reduced infiltration and increase in overland flow. Some wetland plant species rely on adequate groundwater recharge for their survival (e.g., Chorro Creek bog thistle, Pecos sunflower, and Canelo Hills Ladies'-tresses), and others require a specific hydrologic cycle (e.g., water howellia). Other indirect effects to wetland plants may occur during the creation of a wet line or during the mop-up phase of a burn, which typically requires the pumping of water from nearby water sources. Finally, ground-disturbing activities associated with road construction and maintenance and temporary camps (if required) can affect TEP species. In addition, these activities can assist the spread of non-native species into habitats where TEP species are found.

Effects by Habitat Type

Table 4-1 provides an assumed response to fire for all of the TEP plant species considered in this BA. As information on how a prescribed burn would affect populations is not available for all species, determinations were made conservatively, often assuming a negative response to fire if no specific information was available and if it was not apparent that fire would indirectly benefit the species' habitat. Since it is likely that all species would experience some negative effects from direct exposure to fire, and because recovery is dependent on more than just the physiological tolerance of plants to fire (e.g., population size, condition of the site, weather, timing of the burn), the species' habitat type was a major factor in determining the assumed response.

For the most part, plants that occur in communities where fire historically occurred with some regularity are adapted to fires of the same frequency and intensity. Apart from having physical adaptations, which have been discussed in the Direct Effects section, many TEP plants in fire-adapted communities are poor competitors that require frequent disturbance to persist, information that has been provided in Table 4-1, where available. In the absence of fire (or some similar disturbance) in fire-adapted communities, suitable habitat has been lost, and species populations have suffered. Many of these species would be expected to benefit, often indirectly, from fire, as reflected in Table 4-1. Conversely, some TEP plants are long-lived dwellers of communities where fire was never an important component (e.g., many species of cactus). These species are not adapted to fire, and it is assumed that they would be adversely affected by fire treatments. As stated previously, the information in Table 4-1 is intended to provide some information on the degree of adaptation and/or tolerance to fire by the numerous TEP plant species covered by this BA. This information will allow for a general assessment of the potential for these species to be adversely affected by prescribed fire treatments. In all cases, however, final effects determinations should be made at the local level, as many of these species, regardless of their fire-adaptedness, are so reduced in number that populations will still need to be protected from direct exposure to fire treatments.

The majority of desert TEP plants (listed under Temperate Desert and Subtropical Desert on Table 4-1) occur in desert shrub communities. The primary response of these communities to fire is a decrease in shrub cover, and an increase in dominance by grasses. It is believed that fire historically had some role in desert shrublands and grasslands, but for many desert communities there is little detailed information about historical fire frequencies, sizes, and intensities (Brown and Smith 2000). For this reason, the use of fire in desert shrublands is controversial.

In addition, many desert TEP plant species occur in dry, fragile habitats that are too sparse to carry a fire, although the amount of fuel loading may vary from year to year depending on the amount of rainfall a particular site receives. These species are not likely to be adapted to fire, and there is little information about their fire tolerance, since fire is so infrequent. Therefore, it is assumed that most TEP plant species in desert habitats would be adversely affected by fire treatments, pending an assessment of the site at the local level prior to treatment.

It is also assumed that the majority of the TEP plant species occurring in the Subtropical Steppe Ecoregion would be adversely affected by fire, for many of the reasons stated in the previous paragraph. Many of these species are perennials that occur in communities that are highly susceptible to most forms of disturbance, and in many cases are members of stable, climax communities that would not be expected to benefit from the use of prescribed fire.

Many habitats (and the plant species in them) in the Temperate Steppe Ecoregion adapted with fire and grazing, and would generally be expected to respond positively to prescribed fire, as reflected in Table 4-1. The Mediterranean Ecoregion Division also contains a variety of habitat types, such as chaparral, oak woodland, and grasslands, that are fire adapted and would be expected to benefit from the use of prescribed fire. All of the TEP plant species in the Marine Ecoregion Division are also likely to benefit from fire. Despite the assumed responses listed in Table 4-1, however, local BLM offices would still need to make a determination about the possible impacts of fire to TEP populations and their habitats, prior to implementing burn treatments.

Mechanical Treatment Methods

Direct Effects

Potential direct effects to TEP plants from mechanical treatments include injury or mortality to the plants or their seedbanks.

Because mechanical treatment methods are intended to remove entire stands of vegetation, they would likely cause direct injury or mortality to any TEP plants present on the treatment site. Plants removed by the roots would be unable to recover through resprouting or any other form of vegetative regrowth, whereas some plants chopped down above the soil would be able to resprout following treatment. In instances where the top layer of soil was also removed, the seed bank of the species would be negatively impacted. Species with small populations or very limited distributions could be extirpated by such an occurrence. Annual TEP plants, given their short lifespan and reliance on seed, would be able to be killed with few impacts to populations, provided the seedbank and germination conditions were not negatively affected by the treatments.

Indirect Effects

Indirect effects would be expected primarily from habitat alteration.

The potential beneficial effects resulting from mechanical treatments would be similar to those discussed under prescribed fire: early-successional and disturbance-dependent species would benefit from the open conditions, and the removal of fuel sources would decrease the risks of future high-intensity wildfire.

Mechanical treatments can also benefit rare plant populations by removing large tracts of non-native species. In some cases, mechanical treatments could increase the amount of suitable habitat available for a species by improving the quality of habitat adjacent to existing habitat. It is most likely, though certainly not true in all cases, that sites in need of extensive mechanical treatments would be much altered from their original conditions and unlikely to support healthy populations of TEP species in the first place.

Potential adverse effects to plant habitat from mechanical treatments include damage from the use of heavy vehicles, such as soil compaction (which can lead to the puddling of water), scarification, and mixing of soil layers (Spence et al. 1996). Piling of slash can also lead to soil compaction. The reduced infiltration of water in compacted soils can hinder the re-establishment of seedlings or the growth of established vegetation on a treated site. Mixing of mineral and organic layers influences the revegetation process as well (Beschta et al. 1995). Many TEP species occurring in harsh environments are highly sensitive to any activity that disturbs the soil, as indicated in Table 4-1 (under the Additional Information heading). Any use of machinery in habitat for these species would be likely to result in an adverse effect.

Beyond the erosion caused by the removal of vegetation from the site, increased surface erosion would be expected as a result of disturbances to the duff layer and the removal of organic material. These effects would be most severe if the treatments occurred during wet weather. Fuels and other chemicals used in association with heavy equipment could also be released to the environment. Wetland plant species could be impacted by increased surface water runoff, which would alter hydrology, and increased sedimentation.

Over the long term, the suitability of the treatment site for supporting TEP species would depend on the suite of species that became established after the site was cleared. A site cleared, but not replanted or reseeded, would typically favor early successional species, and would be expected to be beneficial for early-successional TEP plants. However, noxious weeds are also well-adapted to disturbed sites, and in many cases can outcompete TEP species. It is expected that mechanical treatment methods would occur on sites with a large amount of undesirable vegetation, and it is likely that propagules of these species would be able to recolonize the site. Thus, it is possible that mechanical treatments alone would have no long-term effect on TEP habitat, or would have a negative effect. However, if replanting or reseeding with native species was also done at the site, long-term effects could be positive, by eventually replacing a site dominated by non-natives species to one dominated by native species.

Manual Treatment Methods

Direct Effects

In general, the effects of manual treatment methods would be minimal, both because of the low level of environmental impact of this method and the limited area in which its use is feasible. Plants could be directly killed or injured if accidentally removed during a treatment, or if trampled by workers treating a site.

Indirect Effects

A long-term beneficial effect to TEP habitat would be expected as a result of manual treatment methods. Removal of competing or unwanted vegetation could increase the health or vigor of existing populations, or increase the suitability of unoccupied sites. Removal of fuel sources would reduce the future risks of damaging wildfires on TEP habitat. Unlike mechanical methods, soil disturbance and risks of erosion would be minimal, unless large areas were cleared of duff and debris, especially on steep slopes.

In general, the negative effects of manual treatment methods on habitat would be minimal. There could be a slight increase in fire hazard after a manual treatment if plant materials were left on the ground in the treatment area. However, this increase would most likely be minimal and temporary. There would be minor risks associated with the use of power hand tools, which may be powered by oil and other fuels. Use of SOPs while operating this equipment would minimize the risk of chemical leaks onto sensitive plants or into their habitats.

Biological Control Treatments

Domestic Animals

There is a wide range of treatments using domestic animals that could be used on public lands. Factors such as timing, area, intensity, frequency, duration, and the species' tolerance to grazing must all be taken into account when predicting the effects of this form of biological control on TEP plant species. The pre-treatment condition of a site and its disturbance history are also important factors to consider when assessing potential impacts.

TABLE 4-1
Attributes of Plant Species Considered in Analysis

Common Name	Habitat	Assumed Response to Fire	Life Form	Stature/ Root Type	Additional Information
Temperate Desert Ecoregion					
Malheur wire-lettuce	Sagebrush/shrubsteppe (Great Basin Desert)	Adversely affected (downy brome invasion)	Annual herb	Upright, to 20 in	Fire lane (buffer zone) maintained around critical habitat.
Desert yellowhead	Sandstone outcrops and sparse vegetation (low cushion plants and Indian ricegrass)	Adversely affected (not adapted)	Perennial herb	Upright, to 12 in Taproot	Occurs in sparsely vegetated areas and threatened by surface disturbance.
Steamboat buckwheat	Desert shrub (Great Basin Desert)	Adversely affected	Perennial herb	Low and densely matted Shallow, rhizomatous system (young) to woody taproot (mature)	Tends to be the most common plant in the specific areas where it occurs, early successional species, and colonizes substrates derived from hot spring deposits.
Slickspot peppergrass	Sagebrush-steppe (microsites called slickspots)	Adversely affected (downy brome invasion)	Annual porbiennial herb	Upright, to 12 in	Occurs in sparsely vegetated areas, highly dependent on seed bank, and displaced by non-native annuals.
Fish Slough milk-vetch	Desert wetland (Mesic alkali meadows adjacent aquatic habitats)	Adversely affected	Perennial herb	Prostrate	Livestock will graze flowering stalks.
Autumn buttercup	Desert wetland (spring-fed wet meadow in the transition to upland)	Adversely affected	Perennial herb	Upright, to 2 ft	Reproduction by seed, trampling/grazing known to be a threat, palatable to livestock/small mammals, and selectively grazed.
Clay-loving wild buckwheat	Clay barrens (Near-barren hills on substrates high in salt and gypsum)	Adversely affected (not adapted)	Perennial subshrub	Low, rounded, to 4 in tall	Occurs in sparsely vegetated areas, flowers produced over a long period, lack of invading species capable of dominating sites, sensitive to surface disturbance, and member of a stable climax association.
Uinta Basin hookless cactus	Desert shrub (alluvial river terraces above the flood plain) and pinyon-juniper	Adversely affected	Perennial succulent	Ovoid or globular, to 2.5 in	Sexual reproduction only, habitat susceptible to surface disturbance, limited grazing beneficial, and moderate to heavy grazing causes physical damage by trampling.
Wright fishhook cactus	Desert shrub (saltbush), desert grassland, and pinyon-juniper	Adversely affected	Perennial succulent	Upright and small	Reproduces by seed, plants rare or absent where cryptogamic crust destroyed or undeveloped, and vulnerable to surface disturbance.

TABLE 4-1 (Cont.)
Attributes of Plant Species Considered in Analysis

Common Name	Habitat	Assumed Response to Fire	Life Form	Stature/ Root Type	Additional Information
Barneby ridge-cress	Pinyon-juniper (shale barrens)	Adversely affected	Perennial herb	Raised clump or cushion, to 6 in Deep woody taproot	Occurs in sparsely vegetated areas and vulnerable to surface disturbance.
Deseret milk-vetch	Pinyon-juniper	Adversely affected	Perennial herb	Upright, to 6 in	Trampling, erosion caused by grazing known to be threats and palatable to livestock.
San Rafael cactus	Pinyon-juniper and mixed desert shrub-grassland	Adversely affected	Perennial Succulent	Subglobose to ovoid, small, to 2.5 in	Shrinks underground during dry or cold seasons and vulnerable to surface disturbance.
Clay reed-mustard	Desert shrub	Adversely affected	Perennial herb	Upright, to 12 in	Vulnerable to surface disturbance and current levels of grazing do not impact the species.
Barneby reed-mustard	Desert shrub	Adversely affected	Perennial herb	Upright, to 15 in	Vulnerable to surface disturbance and current levels of grazing do not impact the species.
Shrubby reed-mustard	Desert shrub	Adversely affected	Perennial herb	Upright, to 12 in	Vulnerable to surface disturbance and current levels of grazing do not impact the species.
Last Chance townsendia	Pinyon-juniper (small barren openings)	Adversely affected	Perennial herb	Low growing and stemless	Vulnerable to surface disturbance.
Maguire daisy	Pinyon-juniper and mountain shrub (partial shade)	Adversely affected	Perennial herb	Decumbent, sprawling, or upright, to 7 in	Vulnerable to surface disturbance.
Maguire primrose	Mountain shrub (cliffs and boulders in cracks or a mat of moss)	Adversely affected (not adapted)	Perennial herb	Upright, to 4 in	Sensitive to surface disturbance.
Clay phacelia	Pinyon-juniper (dwarf) and mountain shrub	Adversely affected	Annual herb	Upright, to 14 in	Long-lived seeds, member of a stable community, and high mortality from grazing by wildlife and livestock.
Heliotrope milk-vetch	Barren outcrops (alpine habitats)	Adversely affected (not adapted)	Perennial herb	Low growing, to 2 in	Occurs in sparsely vegetated areas, current levels of grazing do not appear to adversely affect, and occurs in a fragile ecosystem.
Dudley Bluffs bladderpod	Barren outcrops (white shale and along drainages)	Adversely affected (not adapted)	Perennial herb	Cushion	Occurs in sparsely vegetated areas and vulnerable to surface disturbances.

TABLE 4-1 (Cont.)
Attributes of Plant Species Considered in Analysis

Common Name	Habitat	Assumed Response to Fire	Life Form	Stature/ Root Type	Additional Information
Dudley Bluffs (Piceance) twinpod	Barren outcrops (white shale and along drainages)	Adversely affected (not adapted)	Perennial herb	Upright, to 8 in	Occurs in sparsely vegetated areas and vulnerable to surface disturbances.
Subtropical Desert Ecoregion					
Coachella Valley milk-vetch	Dunes and sandy flats	Adversely affected	Perennial or biennial herb	Upright, to 12 in	Can occur in disturbed areas.
Lane Mountain milk-vetch	Desert shrub (Mojave creosote bush scrub)	Adversely affected	Perennial herb	Zigzagging stems, to 20 in	Buried root crown, may require host or nurse shrub for germination, and fire frequency in habitat has increased.
Peirson's milk-vetch	Sand dunes (Sonoran desert)	Adversely affected (not adapted)	Short-lived perennial herb	Upright, to 27 in Very long taproot and no lateral roots	Occurs in sparsely vegetated areas and seedlings vulnerable to crushing.
Triple-ribbed milk-vetch	Sandy/gravelly soils (in arid canyons at edge of desert; Coachella Valley)	Adversely affected	Annual or perennial herb	Upright or ascending, to 10 in	Requires open soil and tolerant of soil disturbance.
Amargosa niterwort	Desert shrub and saltgrass meadow (springfed saline/alkaline mudflats and sinks)	Adversely affected	Perennial herb	Upright, small, and bushy Heavy underground rootstock	No plants observed in disturbed areas, sensitive to surface disturbance, requires open conditions, member of a climax community and dependent on flows from Ash Meadows aquifer.
Ash Meadows milk-vetch	Desert shrub (barren flats, washes, and knolls of alkaline soils)	Adversely affected	Perennial shrub	Low and mat-forming	No growth observed in areas that have been disturbed, sensitive to surface disturbance, and dependent on flows from Ash Meadows aquifer.
Spring-loving centaury	Desert shrub (riparian areas in mesic saltgrass meadows)	Adversely affected	Annual herb	Upright, to 18 in Slender taproot	Dependent on flows from Ash Meadows aquifer and trampling a known threat.
Ash Meadows ivesia	Desert shrub (spring areas and mesic saltgrass meadow)	Adversely affected	Perennial herb or shrub	Matted, to 2.5 in tall Deep, thick, and woody root	Dependent on flows from Ash Meadows aquifer and trampling a known threat.

TABLE 4-1 (Cont.)
Attributes of Plant Species Considered in Analysis

Common Name	Habitat	Assumed Response to Fire	Life Form	Stature/ Root Type	Additional Information
Ash Meadows gumplant	Desert shrub (saltgrass meadows along streams/pools near ash-screwbean-mesquite woodlands and desert shadscale scrub)	Adversely affected	Biennial and perennial herb	Upright, to 40 in Stout, and woody taproot	Colonizes recently disturbed areas, dependent on flows from Ash Meadows aquifer, and trampling a known threat.
Ash Meadows blazingstar	Desert shrub (shadscale vegetation surrounding spring and seep areas, open areas, and salt-crusted clay soils)	Adversely affected	Biennial herb	Upright, to 20 in Short taproot	Occurs in sparsely vegetated areas, not found on disturbed sites, sensitive to surface disturbance, found in open areas without any vegetation cover, member of a climax community, and dependent on flows from Ash Meadows aquifer.
Ash Meadows sunray	Desert shrub (spring and seep areas and alkaline soils)	Adversely affected	Perennial shrub	Clumps, to 16 in tall Woody root stock	Dependent on flows from Ash Meadows aquifer and trampling a known threat.
Nichol's Turk's head cactus	Desert scrub (talus or bedrock)	Adversely affected (not adapted)	Perennial succulent (slow-growing)	Upright, to 18 in	Occurs in sparsely vegetated areas, member of a climax habitat, and habitat alterations likely to impact the species.
Kearney's blue-star	Desert shrub, semi-desert grassland (Mexican blue oak associations)	Adversely affected	Perennial herb	Upright, to 2.3 ft	Not grazed, but habitat impacted.
Pima pineapple cactus	Desert shrub (Sonoran scrub) and semi-desert grasslands	Tolerates fire	Perennial succulent	Semi-circular, to 18 in	Occurs in sparsely vegetated areas, occurs in a fire-adapted ecosystem, but can be harmed by fire, open microsites may protect, and non-native species have altered habitat.
Huachuca water-umbel	Wetlands (riverine systems, cienegas, springs; semi-aquatic species)	Tolerates fire	Perennial herb	Creeping rhizomes	Reproduction primarily asexual, occupies disturbed habitat after a flood, and persists until outcompeted.
Canelo Hills ladies'-tresses	Wetlands (cienega and streamside habitat in semidesert grassland and oak savannah)	Tolerates fire	Annual and perennial herb	Upright, to 20 in	May favor some form of mild disturbance, such as grazing.
Cochise pincushion cactus	Desert shrub/ semidesert grassland interface	Adversely affected (not adapted)	Perennial succulent	Suborbicular, to 2.4 in	Occurs in sparsely vegetated areas and most of stem underground and occurs in undisturbed soil.

TABLE 4-1 (Cont.)
Attributes of Plant Species Considered in Analysis

Common Name	Habitat	Assumed Response to Fire	Life Form	Stature/ Root Type	Additional Information
Arizona cliff-rose	Desert shrub (Chihuahuan Desert)	Adversely affected	Perennial shrub	Low, straggling, to 6 ft	Long-lived, high reproductive output, low recruitment, and susceptible to soil disturbance and grazing.
Arizona hedgehog cactus	Desert shrub (Interior chaparral), evergreen woodland (Madrean), and desert grassland	Adversely affected	Perennial succulent	Upright, to 16 in	Moderate to high shrub densities preclude establishment, member of a stable climax community, and susceptible to disturbance.
Dwarf bear-poppy	Desert shrub (mixed)	Adversely affected	Perennial herb (evergreen)	Upright, to 3 in	Soil seedbank critical for persistence, member of a stable plant community, and susceptible to disturbance.
Holmgren milk-vetch	Desert shrub (shallow soils; Mojave Desert)	Adversely affected (not adapted)	Perennial herb	Stemless	Occurs in sparsely vegetated areas, vulnerable to surface disturbance, not palatable, and introduction of non-natives, and fire, known to be threats.
Shivwitz milk-vetch	Desert shrub (Mojave Desert)	Adversely affected (not adapted)	Perennial herb	Prostrate or upright, to 40 in	Vulnerable to surface disturbance, extremely palatable to wildlife and domestic livestock, currently overgrazed, and introduction of non-natives, and fire, known to be threats.
Gypsum wild-buckwheat	Desert shrub (Chihuahuan Desert scrub; gypsum soils)	Adversely affected (not adapted)	Perennial herb	Small and upright Persistent woody root crown	Occurs in sparsely vegetated areas, present where hard gypsum crust is broken, and some level of surface disturbance beneficial.
Lee pincushion cactus	Semi-desert grassland (Chihuahuan Desert)	Adversely affected (not adapted)	Perennial succulent (long-lived)	Cushion	Occurs in sparsely vegetated areas and sexual reproduction only.
Sneed pincushion cactus	Semi-desert grassland (Chihuahuan Desert)	Adversely affected (not adapted)	Perennial succulent (long-lived)	Cushion	Occurs in sparsely vegetated areas and sexual reproduction only.
Pecos sunflower	Desert wetlands	Adversely affected	Annual herb	Upright, to 6.5 ft	Livestock will eat.
Subtropical Steppe Ecoregion					
Arizona agave	Desert shrub (open chaparral), desert grassland, grassland, and pinyon-juniper transition zone.	Adversely affected	Perennial succulent	Depressed-globose	Poor reproduction exacerbated by grazing of flowering stalks.
Brady pincushion cactus	Desert shrub (Great Basin; open exposed habitats)	Adversely affected (not adapted)	Perennial succulent	Semiglobose, to 2.5 in tall	Occurs in sparsely vegetated areas, a member of a stable, climax community, and impacted by human disturbances.

TABLE 4-1 (Cont.)
Attributes of Plant Species Considered in Analysis

Common Name	Habitat	Assumed Response to Fire	Life Form	Stature/ Root Type	Additional Information
Peebles Navajo cactus	Desert grassland	Adversely affected	Perennial succulent	Globose, to 1 in tall	Retract into soil during dry weather and sensitive to surface disturbance.
Welsh's milkweed	Sand dunes (surrounded by pinyon-juniper and sagebrush)	Adversely affected (not adapted)	Herb	Upright and tall Rhizomatous	Sensitive to surface disturbance.
Jones cycladenia	Desert shrub (mixed), pinyon-juniper	Adversely affected	Perennial herb (long lived)	Upright, to 6 in Rhizomatous	Occurs in sparsely vegetated areas, low sexual reproductive success, and vulnerable to surface disturbance.
Siler pincushion cactus	Desert shrub, sagebrush-steppe, and pinyon-juniper (gypsum soils)	Adversely affected (not adapted)	Perennial succulent	Globose or cylindrical, to 5 in	Occurs in sparsely vegetated areas and vulnerable to surface disturbance.
Navajo sedge	Pinyon-juniper woodland (Great Basin; hanging garden habitats)	Adversely affected	Perennial herb	Upright, to 16 in Rhizomatous	Palatable to livestock.
Kodachrome bladderpod	Pinyon-juniper (white shale knolls with thin soils)	Adversely affected (not adapted)	Perennial herb	Densely matted and depressed mounds	Occurs in sparsely vegetated areas, vulnerable to surface disturbance, and current level of grazing does not impact species.
Winkler cactus	Desert shrub (saltbush dominated)	Adversely affected	Perennial succulent	Globose, to 2.5 in	Vulnerable to surface disturbance.
Mesa Verde cactus	Desert shrub (low-rolling clay hills)	Adversely affected (not adapted)	Perennial succulent (long-lived)	Ovoid to depressed-globose, to 7 in	Occurs in sparsely vegetated areas, low reproductive potential, and sensitive to disturbance or modification.
Mancos milk-vetch	Sandstone outcrops (sagebrush and pinyon-juniper)	Adversely affected (not adapted)	Perennial herb (long-lived, slow growing)	Small, tufted, to <1 in tall	Occurs in sparsely vegetated areas, competition avoider, highly susceptible to surface disturbance, not grazed, and trampling doesn't affect.
Knowlton cactus	Pinyon-juniper (open spaced)	Adversely affected	Perennial succulent	Very small, to 1.5 in tall	Retracts underground during the dry season.
Zuni fleabane	Pinyon-juniper	Adversely affected	Perennial herb (long-lived)	Large clumps, to 18 in tall Rhizomatous	Does not tolerate surface disturbance, clones common, and establishment of new plants by seed is rare.
Sacramento prickly poppy	Semi-desert grassland and conifer woodland (open areas)	May benefit (habitat)	Perennial herb (short-lived)	Upright, to 5 feet Long taproot in mature plants	Early successional species and young plants more palatable to livestock than mature plants.

TABLE 4-1 (Cont.)
Attributes of Plant Species Considered in Analysis

Common Name	Habitat	Assumed Response to Fire	Life Form	Stature/ Root Type	Additional Information
Kuenzler hedgehog cactus	Pinyon-juniper (lower fringes)	Adversely affected	Perennial succulent	Small, to 6 in	Only sexual reproduction, grass/forb cover are important for catching and hiding seeds, plants not found where surface of soil is disrupted, and grazing a known threat.
Todsen's pennyroyal	Pinyon-juniper (Great Basin, shady areas and openings with thin grasses)	Adversely affected (exact response to fire not known)	Perennial herb	Extensive underground rhizome system; Upright, to 8 in	Low sexual reproduction, expected to resprout after fire, and plants not grazed by livestock, but trampling and soil erosion adversely affect.
Temperate Steppe Ecoregion					
Western prairie fringed orchid	Tallgrass prairie	May benefit (habitat)	Perennial herb	Upright, to 4 ft Tuber	Can occur in disturbed sites, large reproductive potential, and occurs in fire- and grazing-adapted communities.
Blowout penstemon	Dune blowouts	May benefit (habitat)	Perennial herb	Decumbent to upright, to 2 ft Has adventitious roots	Occurs in sparsely vegetated areas, a sand stabilizer, a primary invader of dune blowouts, and does not persist once sites are completely vegetated.
Colorado butterfly plant	Riparian (early to mid-successional habitats)	May benefit (habitat)	Perennial herb	Basal rosette, flowering stems and upright, to 3 ft	Vegetative rosettes little impacted by disturbances, and succession and invasion by non-natives are known threats.
North Park phacelia	Barren outcrops (in a matrix of sagebrush)	Adversely affected (not adapted)	Biennial or short-lived perennial	Upright, to 9 in	Occurs in sparsely vegetated areas, sandy habitat is very friable, vulnerable to disturbance, and poor reproductive success.
Spalding's catchfly	Grasslands (Palouse prairie)	May benefit (habitat)	Perennial herb (long-lived)	Upright, to 24 in	Reproduces only by seed and known to be affected by non-native species and grazing.
Howell's spectacular thelypody	Grassland (moist alkaline meadows)	May benefit (habitat)	Biennial herb	Upright, to 2 ft	High reproductive output, known to be affected by herbicides and grazing, and does not compete well with non-natives.
Macfarlane's four-o'clock	Grassland (bunchgrass)	May benefit (habitat, except in areas where downy brome has invaded)	Perennial herb	Upright Deep-seated, thickened root	Burning while dormant should not harm this species, livestock grazing is a known threat, and herbicides are a known threat.
Osterhout milk-vetch	Sagebrush (open sites)	May benefit (habitat)	Perennial herb	Upright, to 40 in	Vulnerable to surface disturbance.
Penland beardtongue	Sagebrush (open sites)	May benefit (habitat)	Perennial herb	Upright, to 10 in	Vulnerable to surface disturbance.

TABLE 4-1 (Cont.)
Attributes of Plant Species Considered in Analysis

Common Name	Habitat	Assumed Response to Fire	Life Form	Stature/ Root Type	Additional Information
Penland alpine fen mustard	Wetlands (alpine marshes and alpine tundra on peat mats)	Adversely affects	Perennial herb	Upright, to 3 in	Harsh habitat and vulnerable to surface disturbance.
Mediterranean Ecoregion					
Gentner's fritillary	Oak woodland, forest, and chaparral and grassland (open sites)	May benefit (habitat)	Perennial herb	Upright, to 28 in Fleshy bulb	Grows in places that have experienced disturbance, requires some level of disturbance, and reproduces asexually.
Ione manzanita	Chaparral	May benefit (habitat and germination)	Perennial shrub (evergreen)	Low, spreading, to 4 ft.	Depends on fire for germination, and does not sprout after fire (must reproduce by seed).
Ione buckwheat	Chaparral	May benefit (habitat)	Perennial herb	Upright, to 8 in.	Occurs on barren outcrops.
Stebbins' morning-glory	Chaparral (on gabbro-derived soils)	May benefit (habitat and germination)	Perennial herb	Prostrate Extensive root system	Extensive seed bank, shade intolerant, and non-native species and excessive grazing are known threats.
Pine Hill ceanothus	Chaparral (openings)	May benefit (habitat and germination)	Perennial shrub (evergreen)	Prostrate	Does not resprout after fire (depends on seeds), frequent fires adversely affects, and non-native species and excessive grazing are known threats.
Pine Hill flannelbush	Chaparral (rocky outcrops)	May benefit (habitat and germination)	Perennial shrub	Spreading and decumbent	Seeds limited, require fire to germinate, excessive fire frequency adversely affects, and non-native species and excessive grazing are known threats.
El Dorado bedstraw	Oak woodland	May benefit (habitat and germination)	Perennial herb	Prostrate	Non-native species and excessive grazing are known threats.
Layne's butterweed	Chaparral (open rocky areas)	May benefit (habitat and germination)	Perennial herb	Upright Sprouts from a rootstock	Non-native species and excessive grazing are known threats.
Braunton's milk-vetch	Chaparral (limestone outcrops)	May benefit (habitat and reproduction)	Perennial herb (short-lived)	Upright, to 5 feet	Seeds persist in the soil for many years.
Nevin's barberry	Chaparral and alluvial scrub	May benefit (habitat)	Perennial shrub (evergreen)	Upright, to 12 ft Rhizomatous	Habitat being encroached by exotic species.
Mexican flannelbush	Chaparral and closed cone coniferous forest	May benefit (habitat)	Perennial tree or shrub (evergreen)	Upright, to 19 ft	Too frequent fires or fires during the reproductive season may imperil the species.

TABLE 4-1 (Cont.)
Attributes of Plant Species Considered in Analysis

Common Name	Habitat	Assumed Response to Fire	Life Form	Stature/ Root Type	Additional Information
San Benito evening-primrose	Chaparral and forest (openings and barren alluvial soil)	May benefit (habitat)	Annual herb	Upright, <1 in tall	Occurs in sparsely vegetated areas and sensitive to surface disturbance.
Morro manzanita	Chaparral, coastal scrub, and coast live oak	May benefit (habitat)	Perennial shrub (long-lived)	Upright, to 13 ft	Does not resprout after fire and non-native species are a known threat.
Indian Knob mountain balm	Chaparral (coastal maritime) and oak woodlands	May benefit (habitat)	Perennial shrub (evergreen)	Upright, to 13 ft Rhizomatous	New growth from root sprouts.
Orcutt's spineflower	Chaparral (southern maritime)	May benefit (habitat)	Annual herb	Prostrate	Occurs on weathered sandstone bluffs and non-native plant species are a known threat.
Encinitis baccharis	Chaparral (southern maritime)	May benefit (habitat)	Perennial shrub	Broom-like, to 4.5 ft	Non-native plant species are a known threat.
Slender-horned spineflower	Alluvial scrub and chaparral	May benefit (habitat)	Annual herb	Prostrate and small	Does not occur in areas dominated by non-native species and occurs in areas lacking surface disturbance.
Santa Ana River woolly-star	Alluvial scrub (early to intermediate)	May benefit (habitat)	Perennial shrub	Upright, to 3 ft	Does not occur in areas dominated by non-native species.
La Graciosa thistle	Coastal dunes (back dune and coastal wetlands)	May benefit (habitat)	Annual herb	Mound-like or upright, to 40 in	Non-native species are a known threat and weeds must be controlled post fire.
Lompoc yerba santa	Chaparral (maritime) and South Bishop pine forests	May benefit (habitat)	Perennial shrub	Upright, to 10 ft	Low seed productivity, resprouts after fire, non-native species are a known threat, and weeds must be controlled post fire.
Monterey spineflower	Coastal dunes (foredunes, scrub, maritime chaparral, and other areas with sandy soil)	May benefit (habitat)	Annual herb	Prostrate	Occurs in sparsely vegetated areas, found in disturbed areas, trampling of habitat may aid germination, and non-native species are a threat.
Howell's spineflower	Coastal dunes (coastal foredunes and sandy coastal prairie)	May benefit (habitat)	Annual herb	Prostrate, to 4 inches	Occurs in sparsely vegetated areas and non-native plant species can outcompete.
Menzies' wallflower	Coastal dunes (coastal foredune)	May benefit (habitat)	Biennial and perennial herb (succulent)	Low, rosette forming	Occurs in sparsely vegetated areas, seed bank is contained in old standing plants, and non-native species are a threat.

TABLE 4-1 (Cont.)
Attributes of Plant Species Considered in Analysis

Common Name	Habitat	Assumed Response to Fire	Life Form	Stature/ Root Type	Additional Information
Monterey gilia	Coastal dunes (dune scrub, coastal sage scrub, and maritime chaparral)	May benefit (habitat)	Annual herb	Upright	Occurs in moderately disturbed areas and non-native species are a threat.
Beach layia	Coastal dunes	May benefit (habitat)	Annual herb (succulent)	Low, to 6 in	Occurs in sparsely vegetated areas and non-native species are a threat.
Western lily	Coastal dunes (scrub and early successional bogs)	May benefit (habitat)	Perennial herb	Upright, to 8 ft Bulblike rhizome	Populations have been maintained by grazing and benefits from the presence of some low shrubs.
San Diego ambrosia	Drainages (seasonally dry) and open habitats (grassland, coastal sage scrub, and disturbed areas)	May benefit (habitat)	Perennial herb	Upright, to 20 in Rhizomatous	Heavy clonal growth, may be found in disturbed sites, non-native species known to be a threat, mowing and disking known to be a threat, and direct exposure to fire could adversely affect a population.
San Diego thorn-mint	Coastal sage scrub, chaparral, and native grassland (openings)	May benefit (habitat)	Annual herb	Small	Vulnerable to surface disturbance and non-native species a known threat.
Otay tarplant	Grassland (native and mixed) and coastal sage scrub (open)	May benefit (habitat)	Annual herb	Upright, to 10 in	Tolerates/benefits from light grazing and non-native species a known threat.
Otay mesa-mint	Vernal pools (Southern California)	May benefit (habitat)	Annual herb	Upright	Grazing, vehicles, trampling, and non-natives are known threats.
California orcutt grass	Vernal pools (Southern California)	May benefit (habitat)	Annual herb	Upright, to 4 in	Grazing, vehicles, trampling, and non-natives are known threats.
Hairy orcutt grass	Vernal pools (Central Valley)	May benefit (habitat)	Annual herb	Upright, to 8 in	Can tolerate some grazing.
Greene’s tuctoria	Vernal pools (Central Valley)	May benefit (habitat)	Annual herb	Upright, to 6 in	Moderate grazing has little impact and non-natives are a known threat.
Fleshy owl’s-clover	Vernal pools (Central Valley)	May benefit (habitat)	Annual herb	Upright, to 10 in	Partly parasitic and competition from non-natives a known threat.
Hoover’s spurge	Vernal pools (Central Valley)	May benefit (habitat)	Annual herb	Prostrate and mat-former Taprooted	Moderate grazing does not appear to harm and occurs where competition from other species has been reduced.
San Joaquin Valley orcutt grass	Vernal pools (Central Valley)	May benefit (habitat)	Annual herb	Upright, to 6 in	Late spring grazing is a threat and competition with upland non-natives a threat.
Slender orcutt grass	Vernal pools (Central Valley)	May benefit (habitat)	Annual herb	Upright, to 6 in	Moderate grazing does not appear to harm.

TABLE 4-1 (Cont.)
Attributes of Plant Species Considered in Analysis

Common Name	Habitat	Assumed Response to Fire	Life Form	Stature/ Root Type	Additional Information
Contra Costa goldfields	Vernal pools (Northern California; open, grassy areas, woodland and valley grassland)	May benefit (habitat)	Annual herb	Upright, to 12 in	Habitat is highly susceptible to physical damage/destruction.
Cook's lomatium	Vernal pools (Southwest Oregon)	May benefit (competition)	Perennial herb	Slender, and twisted taproot	Competition with non-native grasses, off-road vehicles are known threats, early fall grazing may be beneficial, spring grazing may be detrimental, and fires should occur in early summer.
Large-flowered woolly meadowfoam	Vernal pools (Southwest Oregon; open prairies and wet meadows in a forest matrix)	Tolerates fire	Annual herb	Upright, to 6 in	Competition with non-native grasses, off-road vehicles are known threats, early fall grazing may be beneficial, spring grazing may be detrimental, and fires should occur in early summer.
Butte County meadowfoam	Vernal pools (Northern California)	May benefit (competition)	Annual herb	Prostrate	Poor seed dispersal and non-native species known to be a threat.
Munz's onion	Grassland (needlegrass and mixed); coastal sage scrub and juniper woodlands (grassy openings)	May benefit (habitat)	Perennial herb	Upright, to 1.2 ft Bulb	Fire suppression listed as a threat and grazing known to be a threat.
San Jacinto Valley crownscale	Alkali scrub, alkali playa, vernal pools, and grassland (annual alkali)	May benefit (habitat)	Annual herb	Upright, to 12 in	Fire suppression listed as a threat and grazing known to be a threat.
Thread-leaved brodiaea	Grassland (Southern needlegrass; alkali)	May benefit (habitat)	Perennial herb	Upright, to 16 in	Fire suppression a threat, vulnerable to deep or repeated disking, and grazing known to be a threat.
Hartweg's golden sunburst	Grassland (non-native) and grassland/blue oak woodland ecotone	May benefit (habitat)	Annual herb	Upright, to 6 in	Non-native species known to be a threat and appropriate grazing practices may benefit.
San Joaquin adobe sunburst	Grassland (non-native) and grassland/blue oak woodland ecotone	May benefit (habitat)	Annual herb	Upright, to 18 in	Non-native species known to be a threat.
Purple amole	Grassland, oak woodland	May benefit (habitat)	Perennial herb	Upright Bulb-forming	Occurs in sparsely vegetated areas, known to be impacted by grazing and non-native species, and burning too frequently or during growth and reproduction may impact plants.

TABLE 4-1 (Cont.)
Attributes of Plant Species Considered in Analysis

Common Name	Habitat	Assumed Response to Fire	Life Form	Stature/ Root Type	Additional Information
Keck's checker-mallow	Grassland (annual)	May benefit (habitat)	Annual herb	Upright, to 13 in	Can coexist with some grazing.
California jewelflower	Grassland (annual), juniper woodland, and Upper Sonoran subshrub	May benefit (habitat)	Annual herb	Upright, to 20 in	Persistent seed bank and grazing in the period between the rosette stage and seed set is believed to be detrimental.
San Joaquin woolly-threads	Grassland (annual), saltbush scrub, and subshrub scrub	May benefit (habitat)	Annual herb	Upright to trailing	Forms substantial soil seed bank and may benefit from light to moderate grazing by reducing exotics.
Bakersfield cactus	Valley shrubland (saltbush scrub), oak woodland, and riparian woodland	Adversely affected (not adapted)	Perennial succulent (long-lived)	Upright, to 14 in Shallow root system	Historically occurred in sparsely vegetated areas, vegetative reproduction common, seed production and germination are rare, and direct competition from introduced, annual grasses threatens.
Kern mallow	Valley shrubland (saltbush scrub)	May benefit (habitat)	Annual herb	Upright, to 20 in	Can reinvade disturbed areas and light to moderate grazing may benefit by reducing competition from exotics.
Springville clarkia	Oak woodland (openings, uphill slopes of roadbanks, and small granitic domes)	May benefit (habitat)	Annual herb	Upright, ~3 ft	Heavy grazing may be a threat.
Red Hills vervain	Wetlands (intermittent and perennial streams in grassland/woodland)	May benefit (habitat)	Perennial herb	Upright, to 23 in	Heavy grazing may be a threat.
Cushenbury milk-vetch	Pinyon-juniper and desert shrub (blackbush scrub and Joshua tree woodlands)	Adversely affected	Annual and perennial herb	Prostrate, small	Occurs in habitat that is undisturbed by human activities.
Parish's daisy	Pinyon-juniper and desert shrub (blackbush scrub, and creosote bush-bursage scrub)	Adversely affected	Perennial herb	Upright or ascending, to 12 in Long, simple taproot	Occurs in habitat that is undisturbed by human activities.

TABLE 4-1 (Cont.)
Attributes of Plant Species Considered in Analysis

Common Name	Habitat	Assumed Response to Fire	Life Form	Stature/ Root Type	Additional Information
Cushenbury buckwheat	Pinyon-juniper, desert shrub (Joshua tree woodland and blackbush scrub)	Adversely affected	Perennial herb	Compact, spreading mounds, to 4 in Deep, well-developed woody taproot	Intolerant of shading, poor competitor and occurs in habitat that is undisturbed by human activities.
Cushenbury oxytheca	Pinyon-juniper and live oak woodland	Adversely affected	Annual herb	Basal rosette with upright inflorescence Long, straight taproot	Occurs in habitat that is undisturbed by human activities.
Yreka phlox	Juniper woodland (open ridge) and forest (open); only on serpentine sites	Adversely affected	Perennial subshrub	Upright, to 6 in	Occurs in open habitats.
Metcalf Canyon jewelflower	Grassland (serpentine)	May benefit (habitat)	Annual herb	Upright, to 3 ft.	Grazing/trampling before seed set can harm populations.
McDonald's rock-cress	Chaparral (open, rocky habitats)	May benefit (habitat)	Perennial herb	Flattened rosettes Tap-rooted	Occurs in sparsely vegetated areas and poor competitor.
Chorro Creek bog thistle	Wetlands (seeps and bogs in grassland and chaparral)	May benefit (habitat)	Biennial or short-lived perennial herb	Rosette to upright, to 7 ft	Not usually eaten by cattle and known to be impacted by the seedhead weevil.
Marcescent dudleyea	Rock surfaces (adjacent to perennial streams, areas with little soil formation, and moss/lichen habitat)	Adversely affected	Perennial succulent	Rosette Thickened rootstock	Occurs in sparsely vegetated areas and fire severely reduces population densities and destroys moss substrate.
Marine Ecoregion Division					
Bradshaw's desert-parsley	Grassland (upland bunchgrass prairie)	May benefit (habitat)	Perennial herb	Low and upright Taprooted	Reproduces exclusively by seed and populations that have experienced prescribed fire have a higher probability of survival.
Willamette daisy	Grassland (wetland prairie)	May benefit (habitat)	Perennial herb	Upright to prostrate, to 2.4 in	Member of an early-successional habitat and requires disturbance for persistence.

TABLE 4-1 (Cont.)
Attributes of Plant Species Considered in Analysis

Common Name	Habitat	Assumed Response to Fire	Life Form	Stature/ Root Type	Additional Information
Kincaid's lupine	Grassland (native upland prairie)	May benefit (habitat)	Perennial herb (long-lived)	Upright, to 32 in Rhizomatous	Member of an early-successional habitat and requires disturbance for persistence.
Nelson's checker-mallow	Grassland (Open habitats incl. prairie remnants)	May benefit (habitat)	Perennial herb	Upright, to 5 ft Rhizomatous	Early-successional species, mowing before seed set compromises reproductive output and herbicides are a known threat.
Wenatchee Mountains checker-mallow	Grassland (moist meadows), open conifer stands	May benefit (habitat)	Perennial herb	Upright, to 60 in Stout taproot	Reproduces by seed; high seed output and competition from native and non-native plants a known threat.
Applegate's milk-vetch	Grassland (bunchgrass)	May benefit (habitat and weeds)	Perennial herb	Trailing Taprooted	Reproduction by seeds, grazing by rabbits a threat, and palatable to livestock, absent in grazed areas.
Rough popcornflower	Grassland (interior valley, wet, open microsites)	May benefit (habitat)	Annual or perennial herb	Upright, to 2 ft	Dependent on flooding/fire to maintain habitat, grazing during the spring and early summer causes the most damage, and fall grazing may benefit the species.
Showy stickseed	Ponderosa pine, Douglas-fir forests (openings); unstable talus	May benefit (habitat)	Perennial herb	Upright, to 16 in Slender taproot	Shade-intolerant, non-native species known to be a threat, and fire may increase the risk of landslide.
Marsh sandwort	Wetland (freshwater marshes)	May benefit (habitat)	Perennial herb	Trailing, stems can root at nodes	Competes with other plants for nutrients (dense vegetation).
Tundra Ecoregion Division					
Aleutian shield fern	Rock outcrops	Adversely affected	Fern (perennial)	Tufted, to 6 in Stout rhizome	Grazing by introduced ungulates may be a threat.
Multiple Ecoregions					
Water howellia	Wetland (matrix of dense forest vegetation, often ponderosa pine; aquatic)	Tolerates fire	Annual herb	Submerged or floating stems, to 24 in	Reproduces entirely from seed, requires drawdown for germination, threatened by reed canarygrass, and trampling/grazing can adversely affect.
Ute ladies'-tresses	Wetland and riparian areas (mesic soils or wet meadows near springs, lakes, and perennial streams)	May benefit (habitat)	Perennial herb	Upright, to 20 in Tuberously thickened roots	Occurs in areas where vegetation is relatively open, but not overgrazed., and moderate winter grazing may be beneficial or have no effect.

Direct Effects

Direct effects of weed containment by domestic animals include mortality and injury through browse and trampling, and growth stimulation.

Adverse effects to TEP plant species could occur through direct forage of individual plants, which would be especially likely for species that are palatable to domestic animals. Some plant species, however, are unlikely to be eaten by domestic animals, especially in the presence of more palatable species, or have physical protection against grazing. Grazing typically affects only the aboveground portions of plants, which are ingested by animals. Heavy grazing can cause palatable species to be defoliated (either partially or wholly), which can cause a reduction in plant biomass, plant vigor, and seed production (Kauffman 1988, Heady and Child 1994). The ability to recover from grazing is largely dependent on the extensiveness of the damage and the amount of carbohydrate stores available for plant regrowth. The effects of treatments using domestic animals would be most extensive if TEP plants were browsed before producing seed (reducing the ability of the plant to reproduce), during times of drought or other stress, or if the same plants were grazed repeatedly. Other direct physical damage that could result includes trampling or kicking up plants by hoofed animals.

In some cases, light to moderate grazing can stimulate growth in plants. Removal of plant material that contains carbohydrate reserves may increase photosynthetic activity to replace the lost material. However, the net effect of grazing on plants does not appear to be beneficial (Ellison 1960).

Indirect Effects

A wide range of indirect effects would be expected from using domestic animals to contain weeds on public land. Trampling of soils, especially when wet, can lead to compaction, which decreases soil pore space and reduces the ability of plant roots to penetrate the soil. In addition, loss of plant cover in an area may increase the surface erosion off of a site, especially on steep hillsides. Reduced cover also decreases soil organic matter and soil aggregates, and decreases infiltration rates. In arid and semi-arid regions, trampling by domestic animals breaks up biological soil crusts. These crusts, which can take decades to re-form, have an important role in hydrology and nutrient cycling, and are believed to provide favorable conditions for the germination of vascular plants (Fleischner 1994). In some instances, however, trampling by domestic animals may have a beneficial effect on soil by breaking up impervious surface soils, which can allow for greater water and nutrient infiltration of soils and can aid in covering seeds with soil (Savory 1988).

Other indirect effects to vegetation may occur in wetlands, where TEP species depend on very specific hydrologic conditions to persist. The effects of grazing in riparian areas and other areas adjacent to aquatic habitats include alteration of the flow regime, changes in the routing of water, and incision of the flood channel (see Chapter 4 in PER for more detailed information on the effects of grazing in riparian habitats), all of which can lead to reduced soil moisture in the floodplain/wetland (Spence et al. 1996). Stream downcutting and the resultant lowering of the water table can lead to the encroachment of water-intolerant species into riparian and wetland habitats, and a poorer habitat for rare wetland plant species.

Weed containment by domestic animals would be expected to affect plant habitat by changing the species composition of a site. Domestic animals selectively feed on palatable species, eventually reducing their overall importance in the ecosystem. For example, over time, grazing in desert grassland ecosystems can reduce the dominance of grass species and increase the dominance of shrub species, eventually replacing the grassland community with a desert shrubland community. In upland areas with a history of grazing, the plant species composition has shifted from perennial grasses toward an increased dominance of non-native annuals and weedy species (Heady and Child 1994). In some grazed riparian areas, the shift has been from communities dominated by willows, aspen, sedges, rushes, and grasses to communities that support annual grasses and sagebrush (Spence et al. 1996).

Over the long term, treatments with domestic animals can improve the habitat of some TEP plant species by reducing the cover of non-native or undesirable species. In addition, periodic grazing can help maintain canopy openings and prevent the encroachment of woody species (e.g., ponderosa pine forests, mountain grasslands, desert shrubland). However, grazing has also been linked to the spread of weeds, and can reduce the quality of habitat by spreading propagules (on fur or in dung) throughout treated areas.

Other Biological Control Agents

Direct and Indirect Effects

Biological control agents such as insects and pathogens generally do not have an effect on non-target plant species or habitats. To be approved for use, these agents must highly specific and highly damaging to the target species and able to survive in the target species' habitat. However, some biological control agents have been observed to attack species, in addition to the target plant. The seedhead weevil, for instance, was released to control alien species of thistle, but has also attacked the Chorro Creek bog thistle, a TEP plant species of the same genus. All biocontrol agents utilized by the BLM for vegetation treatments would be tested prior to release to ensure that they are host specific, and would be assessed for potential risks to TEP plant species in the vicinity of their release. As a general rule, it is assumed that biological control agents that attack target species in the same genus as a TEP plant would have an adverse effect on that TEP plant species, unless extensive research has shown otherwise. In addition, biological control agents that attack target species in the same family as a TEP plant may adversely affect that TEP plant species, and should be subject to a high degree of scrutiny prior to a decision that they are safe for use.

Because biological control is a relatively new field, and because biocontrol methods involve complex interactions of pathogens and organisms with other organisms and the environment, it is difficult to determine their potential long-term effects. A biocontrol agent released into the wild would be expected to operate under different conditions than those in a controlled laboratory. And while the introduction of these host-specific agents are carefully studied and planned in advance, there is always a risk of disrupting natural ecosystems. However, as no examples of extensive harm done to natural ecosystems by biocontrol efforts to manage noxious species are known, it is unlikely that use of these agents would have negative long term effects on TEP species and their environments.

Biological control agents would be expected to have long-term positive effects on TEP species by controlling unwanted vegetation in species habitats or in potential habitats. Although biological control agents work slowly and do not eradicate entire populations of weeds, they do weaken a weed's vigor, often reducing its competitive advantage. Thus, rare plant species that are threatened by non-native plant species would be expected to benefit the most from biological control agents. In addition, the reduction in weed vigor on otherwise suitable habitat could provide an increase in suitable habitat for TEP plant species.

Herbicides

The potential effects of herbicide treatments on TEP plant species would vary depending on a number of factors. The location of the application in relation to TEP plant species, and the type of application method utilized, would determine, in part, whether TEP species would be exposed to chemicals. In addition, the type of chemical formulation used (i.e., selective vs. non-selective; pre-emergence vs. post-emergence) and the timing of the application in relation to the phenology of the species of concern would be important factors to consider. Use of herbicides and potential effects to TEP plant species that occur in each proposed treatment area would be considered in detail at the local level prior to initiating an herbicide treatment. At the programmatic level, this BA provides a general analysis of the potential for herbicides currently-approved and proposed for use by the BLM to affect TEP plant species, as determined in ERAs completed by the BLM and Forest Service (see Chapter 2 for more information).

Direct Effects

If herbicide treatments were to occur in habitats where TEP plant species occur, plants could be crushed by trucks and/or ATVs during ground applications. Injury or mortality to plants could occur.

Ecological risk assessments predicted the potential for terrestrial and aquatic TEP plant species to suffer adverse effects as a result of exposure to the herbicides proposed for use by the BLM. Modes of exposure include direct spray of plants, accidental spill of herbicides into a water body with aquatic TEP plants, off-site drift, surface runoff, and wind transport of soils from treatment sites.

In the ERAs, measurable changes in plants as a result of exposure to herbicides included such adverse effects as mortality and reduced growth, reproduction, or other ecologically important sublethal processes (ENSR 2005a-j). It is expected that possible adverse effects to non-target TEP plant species as a result of exposure to herbicides could include one or more of the following: mortality, loss of photosynthetic foliage, reduced vigor, abnormal growth, or reduced reproductive output. Because many TEP plant species have populations that are small, and/or fragmented, they are expected to be more sensitive to many of these effects than plant species with secure populations. One or more of these effects, depending on its extent and severity, could result in the extirpation of a sensitive population. Less severe effects could reduce the size of a population further, reduce its ability to compete with other, more vigorous species, or increase its degree of fragmentation. These population-level effects could in turn reduce the chances of species recovery, or increase the likelihood of a future extirpation due to natural stochastic events, such as catastrophic wildfire or drought. In this discussion, the term "adverse effects," as it pertains to exposure to herbicides, includes any of the above-mentioned effects to individual TEP plants, populations, and/or species.

Direct Spray

According to the ERAs, all of the herbicides proposed for use by the BLM would potentially have adverse effects on terrestrial TEP plant species, should a direct spray of plants occur. In the case of fluridone, 2,4-D, and hexazinone, risk quotients were not calculated because there was a lack of terrestrial plant toxicity testing. Adverse effects to upland TEP plants were assumed as a result of direct spray by one or more of these herbicides.

In aquatic habitats, TEP plant species could be exposed to aquatic herbicides (fluridone, diquat, and certain formulations of 2,4-D, glyphosate, imazapyr, and triclopyr) during the normal application of these herbicides. In the case of diquat, fluridone, glyphosate, and imazapyr direct spray scenarios represent a normal aquatic application. In the case of triclopyr acid, the herbicide is applied directly to the water column to obtain a desired concentration of the herbicide in the water; therefore, normal aquatic application results in a concentration of the herbicide in water that is somewhere in between what would result from a direct spray at the typical application rate and a direct spray at the maximum application rate. Aquatic plants could also be exposed to terrestrial herbicides as a result of an accidental spray of these chemicals into an aquatic habitat.

According to the ERAs for herbicides with aquatic formulations, adverse effects to non-target aquatic plants would potentially occur if they (or their aquatic habitats) were directly sprayed by diquat, imazapyr, triclopyr BEE[1], or triclopyr acid (maximum application rate only), but not if they were directly sprayed by fluridone or glyphosate, or if the water column received the standard aquatic application of triclopyr acid. In addition, since information is not available for 2,4-D, adverse effects to aquatic plants are assumed for this chemical via this exposure pathway. Adverse effects to non-target aquatic plants are also suspected from accidental spray of aquatic habitats by all terrestrial herbicides proposed for use by the BLM, except clopyralid, picloram, and terrestrial formulations of glyphosate and triclopyr acid (typical application rate only).

[1] Risk assessments looked at two forms of triclopyr that are used commercially as herbicides. The triethylamine salt of of triclopyr is referred to as "triclopyr acid," and the butoxyethyl ester of triclopyr is referred to as "triclopyr BEE" (Syracuse Environmental Research Associates, Inc. 2005). Throughout this BA, wherever just "triclopyr" is used, both forms are implied.

Accidental Spill

In the case of an accidental spill of herbicides into an aquatic habitat, nearly all herbicides proposed for use by the BLM, both terrestrial and aquatic, would potentially have adverse effects on non-target aquatic plants, including TEP species. According to the ERAs, however, a spill of picloram would not pose risks to sensitive non-target aquatic plants. Note that adverse effects to TEP aquatic plants are assumed for 2,4-D and hexazinone via this exposure, since the ERAs did not provide the relevant information.

Off-site Drift

Non-target TEP plants could also be exposed to herbicides directly during off-site drift from a nearby treatment site. Off-site drift scenarios under which adverse effects to TEP plants were predicted by ERAs are summarized in Tables 4-2 (terrestrial species) and 4-3 (aquatic species). Note that Forest Service ERAs did not address the potential effects to aquatic plants from off-site drift. For these chemicals, risks to terrestrial plant species are taken to represent the risks to aquatic plant species, unless other information is available (for example, if there is no risk to aquatic plants as a result of direct spray, it is assumed that there is no risk to aquatic plants as a result of off-site drift). As indicated in Tables 4-2 and 4-3, terrestrial plants are more sensitive to off-site drift than aquatic plants; therefore this assumption is conservative.

According to the ERAs, the only herbicide application scenario for which off-site drift would have no risk of causing adverse effects to TEP terrestrial plants at a distance of 25 feet is a ground application of imazapic. Similarly, a helicopter application of imazapic at the typical application rate would have no risk of causing adverse effects to terrestrial TEP plants at a distance of 100 feet. Therefore, this herbicide may be most appropriate for use in treatment areas adjacent to habitats that support terrestrial TEP plant species. For the other herbicides proposed for use by the BLM, adverse effects to terrestrial TEP plants could potentially occur by ground and/or aerial applications at distances ranging from 25 to 1,500 feet. For some herbicides, ERAs were unable assess risks with certainty (i.e., some information was unavailable or drift scenarios did not go out far enough to establish a precise buffer distance), and a conservative buffer distance of ½ mile is assumed. These buffer distances may be changed at the local level if additional information is made available. Conservation measures provided at the end of this section incorporate the buffer distances presented in Tables 4-2 and 4-3.

With the exception of certain applications of chlorsulfuron, clopyralid, diflufuenzopyr, glyphosate, imazapic, Overdrive®, picloram, tebuthiuron, and triclopyr acid, there would be risks to aquatic TEP plants associated with off-site drift of terrestrial herbicides proposed for use by the BLM. Risks were predicted at varying distances, as shown in Table 4-2. Based on this information, minimum buffer distances to protect aquatic TEP plants have been established and incorporated into conservation measures listed at the end of this section. For herbicides for which ERAs were unable to assess risks with certainty, a conservative buffer distance of ½ mile is assumed.

Surface Runoff

Risk assessments analyzed the risks to TEP plant species as a result of exposure to herbicides via surface runoff from an upslope treatment site. Potential effects are summarized in Table 4-4. When considering risks to aquatic plant species as a result of surface runoff, risk assessments completed by the Forest Service presented hazard quotients for just two surface runoff scenarios: runoff in an area with clay soils and an annual rainfall of 100 to 250 inches, and runoff in an area with clay soil and an annual rainfall of 15 inches. Based on the predictions presented in Table 4-4, adverse effects to terrestrial TEP plants would be possible as a result of surface runoff of bromacil, clopyralid, diflufuenzopyr, diuron, imazapyr, metsulfuron methyl, Overdrive®, picloram, sulfometuron methyl, tebuthiuron, or triclopyr under certain site conditions. In addition, since information for 2,4-D and hexazinone is unavailable, it is assumed that adverse effects to terrestrial TEP plants could occur as a result of runoff of these herbicides from an upslope application area under all site conditions.

TABLE 4-2
Potential Effects to Terrestrial Threatened, Endangered, and Proposed Plant Species as a Result of Off-site Drift from Aerial Applications

Herbicide	Ground Application	Aerial Application
2,4-D	Not addressed in ERA.	Not addressed in ERA.
Bromacil	Adverse effects within 1,200 feet.	N/A (herbicide would not be applied aerially)
Chlorsulfuron	Adverse effects within 1,200 feet.	Adverse effects within 1,400 feet.
Clopyralid[1]	Typical application rate: adverse effects within 900 feet. Maximum application rate: adverse effects beyond 900 feet[2].	Typical application rate: adverse effects within 900 feet. Maximum application rate: adverse effects beyond 900 feet[2].
Dicamba	Adverse effects within 25 feet.	Adverse effects within 25 feet.
Diflufenzopyr	Low boom, typical application rate: adverse effects within 100 feet. Low boom, maximum application rate: adverse effects within 900 feet. High boom: adverse effects within 900 feet.	N/A (herbicide would not be applied aerially).
Diquat	Typical application rate: adverse effects within 900 feet. Maximum application rate: adverse effects within 1,000 feet.	Adverse effects within 900 feet.
Diuron	Adverse effects within 1,100 feet.	N/A (herbicide would not be applied aerially).
Fluridone	Effects uncertain.	Effects uncertain.
Glyphosate[1]	Typical application rate: adverse effects within 50 feet. Maximum application rate: adverse effects within 300 feet.	Adverse effects within 300 feet.
Hexazinone	Typical application rate: adverse effects within 300 feet. Maximum application rate: adverse effects within 900 feet.	Not addressed in ERA.
Imazapic	No adverse effects predicted (distances of 25 feet and greater considered).	Helicopter, typical application rate: No adverse effects predicted (distances of 100 feet and greater considered). Helicopter, maximum application rate; or plane, typical application rate: Adverse effects within 300 feet. Plane, maximum application rate: adverse effects within 900 feet.
Imazapyr[1]	Typical application rate: adverse effects within 900 feet. Maximum application rate: adverse effects beyond 900 feet[2].	Typical application rate: adverse effects within 900 feet. Maximum application rate: adverse effects beyond 900 feet[2].
Metsulfuron methyl[1]	Typical application rate: adverse effects within 900 feet. Maximum application rate: adverse effects beyond 900 feet[2].	Typical application rate: adverse effects within 900 feet. Maximum application rate: adverse effects beyond 900 feet[2].

TABLE 4-2 (Cont.)
Potential Effects to Terrestrial Threatened, Endangered, and Proposed Plant Species as a Result of Off-site Drift from Aerial Applications

Herbicide	Ground Application	Aerial Application
Overdrive®	Low boom, typical application rate: adverse effects within 100 feet. Low boom, maximum application rate: adverse effects within 900 feet. High boom: adverse effects within 900 feet.	N/A (herbicide would not be applied aerially).
Picloram[1]	Typical application rate: adverse effects within 900 feet[2]. Maximum application rate: adverse effects beyond 900 feet[2].	Typical application rate: adverse effects beyond 900 feet[2]. Maximum application rate: adverse effects beyond 900 feet[2].
Sulfometuron methyl	Adverse effects within 1,500 feet.	Adverse effects within 1,000 feet.
Tebuthiuron	Low boom, typical application rate: adverse effects within 25 feet. Low boom, maximum application rate: adverse effects within 50 feet. High boom, typical application rate: adverse effects within 50 feet. High boom, maximum application rate: adverse effects within 900 feet.	N/A (herbicide applied in granular form and drift would be minimal).
Triclopyr acid[1]	Typical application rate: adverse effects within 300 feet. Maximum application rate: adverse effects beyond 900 feet[2].	Typical application rate: adverse effects within 500 feet. Maximum application rate: adverse effects beyond 900 feet[2].
Triclopyr BEE[1]	Typical application rate: adverse effects within 300 feet. Maximum application rate: adverse effects beyond 900 feet[2].	Typical application rate: adverse effects within 500 feet. Maximum application rate: adverse effects beyond 900 feet[2].

[1] For these chemicals, ground application scenarios for off-site drift considered use of a low boom only.
[2] For these chemicals, the ERAs did not model spray drift out to a distance at which there would be no risks to TEP plants; therefore, a conservative buffer distance of ½ mile is assumed.
N/A = Not applicable.
Note: The ERAs provided information about the closest distance for which adverse effects were predicted. Buffer distances in this table were determined by extending this distance far enough to sufficiently reduce the likelihood of adverse effects to TEP plant species. To be conservative, in most cases the buffer extends out to the first modeled distance from the application site for which no risks were predicted based on a programmatic-level assessment. Local BLM field offices would be able to use interactive spreadsheets developed for the ERAs to input site-specific characteristics (e.g., soil type, precipitation, vegetation type, treatment method) to develop more precise, and often shorter, distances at which effects could occur.

Risk assessments also predicted risks to aquatic TEP plant species as a result of surface runoff into a water body from an upslope area treated by bromacil, chlorsulfuron, diuron, imazapic, Overdrive®, sulfometuron methyl, tebuthiuron, or triclopyr BEE under various site conditions, as shown in Table 4-4. Adverse effects to TEP aquatic plants were also assumed as a result of surface runoff by 2,4-D and hexazinone, for which relevant risk assessment information was unavailable.

Wind Erosion

Risk assessments analyzed the potential for soil exposed to herbicide treatments to be carried by the wind and affect TEP plant species off site. According to ERAs, there would not be risks to TEP terrestrial plant species as a result of herbicide migration off site in soil at distances of ½ mile or greater from the edge of the application site. Distances closer to the application site were not considered in the ERAs. Risk assessments completed by the Forest

Service looked at quantities of herbicides that could potentially be lost from an application site, but not where eroded soil would land, or how much herbicide would be present in windblown soil within defined distances of the treatment site. Based on the amount of herbicide that could be lost from an application site, the Forest Service ERAs predicted that in areas where wind erosion is likely (i.e., in arid habitats and where the herbicide is incorporated only into the top 1 cm [½ inch] of soil) wind erosion could potentially lead to adverse effects in sensitive plant species. Under more desirable conditions (i.e., relatively deep [10 cm; 4 inches] soil incorporation, low wind speed, and topographic conditions that inhibit wind erosion), wind transport of herbicides from the site would be unlikely. Based on the information provided in BLM and Forest Service ERAs, this BA assumes that in habitats where wind erosion could potentially occur, TEP plant species could suffer adverse effects from wind erosion of soil from treated areas within ½ mile from the edge of the treatment site.

Indirect Effects

Use of herbicides to treat vegetation on public lands could have indirect effects on TEP plant species by altering the species composition of treated areas. Elimination or reduction of non-native species from a site could increase its suitability for TEP plant species, especially those that compete with, have been displaced by, or are otherwise threatened by non-native species. Provided herbicide treatment programs were able to avoid adversely affecting populations of TEP plant species on or near the treatment site, long-term benefits to these populations could potentially occur.

Cumulative Effects

Cumulative effects include the effects of future state, tribal, local, or private actions that are reasonably certain to occur in the action area considered in this BA. 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 ESA.

Private and tribal actions on public lands could affect plant species discussed in this BA. Public activities, including recreation, OHV use, and collection of forest products and other plant materials could impact listed species and species proposed for listing. Direct effects include removal of plants, and trampling or crushing of plants by OHVs, pack horses and mules, hikers, or other recreationists and public land users. Indirect effects include actions occurring in areas away from sites occupied by TEP plant species, but which could harm these species or their habitats. For example, TEP plant species occupying wetland or riparian areas could be impacted if recreational activities were to impact water flows or quality in or near the vicinity of the population. Impacts could result from spills of petroleum products from vehicles, from material found in animal feces, or from other factors entering a nearby water body and impacting the habitat of the TEP plant species.

Livestock grazing on public lands could impact TEP plant species. Livestock could directly affect these species by eating or trampling plants. Indirect effects would include erosion and degradation of water quality in areas of heavy livestock use.

TEP plant species are at risk from private, industrial activities that could occur on public lands, including mining, oil and gas, and ROW development, and timber harvest activities that would potentially disturb large areas of habitat and could result in loss or harm to plant populations. Direct impacts would include loss of habitat and destruction or harm to populations from clearing of land for construction of facilities, surface disturbance associated with timber harvest, and vegetation management at facilities. Air and water pollution, and introduction of noxious weeds and other invasive species, including herbivores, could indirectly affect TEP plant species. If herbicides were used to maintain vegetation on ROW or at facilities, herbicide drift could impact nearby TEP plant populations.

Tribal actions that could harm TEP plant species include the collection and use of these species for traditional lifeway uses. Tribal use of TEP plant species, however, was not identified as a concern requiring discussion in this BA. Indirect effects from tribal actions would be similar to those associated with recreation.

TEP plant species could be indirectly harmed by activities occurring on non-federal lands adjacent to public lands. For example, herbicide treatments on nearby agricultural lands or rangelands could drift onto public lands and harm TEP plant species. A wildfire originating off public lands could spread onto public lands. In addition, impacts to air and water quality, or the spread of weeds, that result from activities that occur off public lands that could affect TEP populations on public lands.

Conservation measures (see below) and SOPs identified in this BA and in the PEIS and PER would reduce the likelihood of TEP plant species being impacted by vegetation treatmetns and by non-federal actions on public lands. The BLM would conduct plant surveys, and analyzie project-level impacts to TEP plants under NEPA as part of the permitting and siting process for activities conducted on public lands. The BLM would conduct local level consultation with the Services, as discussed in Chapter 3, for actions that have potential to affect TEP plant species. The BLM would also coordinate with tribes having an interest in TEP plant species, or potentially affecting these species, on public lands.

Conservation Measures

As dictated in BLM Manual 6840 (*Special Status Species Management*), local BLM offices are required to develop and implement management plans and programs that will conserve listed species and their habitats. In addition, NEPA documentation related to treatment activities (i.e., projects) will be prepared that identify any TEP plant species or their critical habitat that are present in the proposed treatment areas, and that list the measures that will be taken to protect them.

Many local BLM offices already have management plans in place that ensure the protection of these plant species during activities on public land. However, a discussion of these existing plans is outside the scope of this programmatic BA. The following general guidance applies to all management plans developed at the local level.

Required steps include the following:

- A survey of all proposed action areas within potential habitat by a botanically qualified biologist, botanist, or ecologist to determine the presence/absence of the species.
- Establishment of site-specific no activity buffers by a qualified botanist, biologist, or ecologist in areas of occupied habitat within the proposed project area. To protect occupied habitat, treatment activities would not occur within these buffers.
- Collection of baseline information on the existing condition of TEP plant species and their habitats in the proposed project area.
- Establishment of pre-treatment monitoring programs to track the size and vigor of TEP populations and the state of their habitats. These monitoring programs would help in anticipating the future effects of vegetation treatments on TEP plant species.
- Assessment of the need for site revegetation post treatment to minimize the opportunity for noxious weed invasion and establishment.

At a minimum, the following must be included in all management plans:

- Given the high risk for damage to TEP plants and their habitat from burning, mechanical treatments, and use of domestic animals to contain weeds, none of these treatment methods should be utilized within 330 feet of sensitive plant populations UNLESS the treatments are specifically designed to maintain or improve the existing population.
- Off-highway use of motorized vehicles associated with treatments should be avoided in suitable or occupied habitat.
- Biological control agents (except for domestic animals) that affect target plants in the same genus as TEP species must not be used to control target species occurring within the dispersal distance of the agent.
- Prior to use of biological control agents that affect target plants in the same family as TEP species, the specificity of the agent with respect to factors such as physiology and morphology should be evaluated, and a determination as to risks to the TEP species made.
- Post-treatment monitoring should be conducted to determine the effectiveness of the project.

TABLE 4-3
Potential Effects to Aquatic Threatened, Endangered, and Proposed Plant Species as a Result of Off-site Drift

Herbicide[1]	Ground Application	Aerial Application
2,4-D	Not addressed in ERA.	Not addressed in ERA.
Bromacil	Low boom, typical application rate: adverse effects within 100 feet. Low boom, maximum application rate: adverse effects within 900 feet. High boom: adverse effects within 900 feet.	N/A (herbicide would not be applied aerially).
Chlorsulfuron	No adverse effects predicted (distances of 25 feet and greater considered).	Typical application rate: no adverse effects predicted (distances of 100 feet and greater considered). Maximum application rate: adverse effects within 300 feet.
Clopyralid[1]	No adverse effects predicted.	No adverse effects predicted.
Dicamba	No adverse effects predicted (distances of 25 feet and greater considered).	N/A (herbicide would not be applied aerially).
Diflufenzopyr	No adverse effects predicted (distances of 25 feet and greater considered).	N/A (herbicide would not be applied aerially).
Diuron	Low boom, typical application rate: adverse effects within 900 feet. Low boom, maximum application rate: adverse effects within 1,100 feet. Maximum application rate: adverse effects within 1,100 feet.	N/A (herbicide would not be applied aerially).
Glyphosate	No adverse effects predicted.	No adverse effects predicted.
Hexazinone	Typical application rate: adverse effects within 300 feet. Maximum application rate: adverse effects beyond 900 feet.	Not addressed in ERA.
Imazapic	No adverse effects predicted (distances of 25 feet and greater considered).	Typical application rate: no adverse effects predicted (distances of 100 feet and greater considered). Maximum application rate: adverse effects within 300 feet.
Imazapyr[2]	Typical application rate: adverse effects within 900 feet. Maximum application rate: adverse effects beyond 900 feet[3].	Typical application rate: adverse effects within 900 feet. Maximum application rate: adverse effects beyond 900 feet[3].
Metsulfuron methyl[2]	Typical application rate: adverse effects within 900 feet. Maximum application rate: adverse effects beyond 900 feet[3].	Typical application rate: adverse effects within 900 feet. Maximum application rate: adverse effects beyond 900 feet[3].
Overdrive®	No adverse effects predicted (distances of 25 feet and greater considered).	N/A (herbicide would not be applied aerially).
Picloram	No adverse effects predicted.	No adverse effects predicted.
Sulfometuron methyl	Adverse effects within 900 feet.	Adverse effects within 1,500 feet.
Tebuthiuron	No adverse effects predicted (distances of 25 feet and greater considered).	N/A (herbicide would not be applied aerially).
Triclopyr acid[2]	Typical application rate: no adverse effects predicted. Maximum application rate: adverse effects beyond 900 feet[3].	Typical application rate: no adverse effects predicted. Maximum application rate: adverse effects beyond 900 feet[3].

TABLE 4-3 (Cont.)
Potential Effects to Aquatic Threatened, Endangered, and Proposed Plant Species as a Result of Off-site Drift

Herbicide[1]	Ground Application	Aerial Application
Triclopyr BEE[2]	Typical application rate: adverse effects within 300 feet. Maximum application rate: adverse effects beyond 900 feet[3].	Typical application rate: adverse effects within 500 feet. Maximum application rate: adverse effects beyond 900 feet[3].

[1] Note that only terrestrial herbicides are considered for this analysis.
[2] For these chemicals, risks to terrestrial TEP plant species are used to represent risks to aquatic TEP plant species, except for the typical application rate of triclopyr acid (because no risks were associated with direct spray). For these chemicals, ground application scenarios for off-site drift considered use of a low boom only.
[3] For these chemicals ERAs did not model spray drift out to a distance at which there would be no risks to TEP plants; therefore, a conservative buffer distance of ½ mile is assumed.
N/A = Not applicable.
Note: The ERAs provided information about the closest distance for which adverse effects were predicted. Buffer distances in this table were determined by extending this distance far enough to sufficiently reduce the likelihood of adverse effects to TEP plant species. To be conservative, in most cases the buffer extends out to the first modeled distance from the application site for which no risks were predicted based on a programmatic-level assessment. Local BLM field offices would be able to use interactive spreadsheets developed for the ERAs to input site-specific characteristics (e.g., soil type, precipitation, vegetation type, treatment method) to develop more precise, and often narrower buffers.

In addition, the following guidance must be considered in all management plans in which herbicide treatments are proposed to minimize or avoid risks to TEP species. The exact conservation measures to be included in management plans will depend on the herbicide that would be used, the desired mode of application, and the conditions of the site. Given the potential for off-site drift and surface runoff, populations of TEP species on lands not administered by the BLM would need to be considered if they are located near proposed herbicide treatment sites.

- Herbicide treatments should not be conducted in areas where TEP plant species may be subject to direct spray by herbicides during treatments.
- Applicators should review, understand, and conform to the “Environmental Hazards” section on herbicide labels (this section warns of known pesticide risks and provides practical ways to avoid harm to organisms or the environment).
- To avoid adverse effects to TEP plant species from off-site drift, surface runoff, and/or wind erosion, suitable buffer zones should be established between treatment sites and populations (confirmed or suspected) of TEP plant species, and site-specific precautions should be taken (refer to the guidance provided below).
- Follow all instructions and Standard Operating Procedures (SOPs) to avoid spill and direct spray scenarios into aquatic habitats that support TEP plant species.
- Follow all BLM operating procedures for avoiding herbicide treatments during climatic conditions that would increase the likelihood of spray drift or surface runoff.

Please note that the following conservation measures refer to sites where broadcast spraying of herbicides, either by ground or aerial methods, is desired. Manual spot treatment of undesirable vegetation can occur within the listed buffer zones if it is determined by local biologists that this method of herbicide application would not pose risks to TEP plant species in the vicinity. Additional precautions during spot treatments of vegetation within habitats where TEP plant species occur should be considered while planning local treatment programs, and should be included as conservation measures in local-level NEPA documentation.

TABLE 4-4
Potential Effects to Plants as a Result of Surface Runoff

Herbicide	Effects to Terrestrial TEP plants	Effects to Non-target Aquatic Plants
2,4-D	Not addressed in ERA.	Not addressed in ERA.
Bromacil	Maximum application rate: adverse effects in areas with clay soils and where annual precipitation is greater than 50 inches per year. Typical application rate: adverse effects in areas with clay soils and where annual precipitation is greater than 100 inches per year.	Adverse effects where precipitation is greater than 5 inches per year.
Chlorsulfuron	No adverse effects predicted.	Sand and clay soils: adverse effects where precipitation is greater than 10 inches per year. Loam soils: adverse effects where precipitation is greater than 50 inches per year.
Clopyralid	Clay soils: adverse effects where annual precipitation is greater than 10 inches per year.	No adverse effects predicted.
Dicamba	Maximum application rate: adverse effects in areas with clay soils and where annual precipitation is greater than 200 inches per year.	Clay soils: adverse effects where precipitation is greater than 100 inches per year. Loam and sand soils: adverse effects where precipitation is greater than 25 inches per year.
Diflufenzopyr	Clay soils: adverse effects where annual precipitation is greater than 10 inches per year. Silt loam, silt, and clay loam soils: adverse effects where precipitation is greater than 25 inches per year.	No adverse effects predicted.
Diuron	Clay and clay loam soils: adverse effects where annual precipitation is greater than 25 inches per year. Loam soils: adverse effects where annual precipitation is greater than 150 inches per year.	Adverse effects where precipitation is greater than 5 inches per year.
Glyphosate	No adverse effects predicted.	No adverse effects predicted.
Hexazinone	Not addressed in ERA.	No addressed in ERA.
Imazapic	No adverse effects predicted.	Sand: adverse effects where precipitation is greater than 10 inches per year. Clay and clay loam soils: adverse effects where precipitation is greater than 25 inches per year. Loam soils: adverse effects where precipitation is greater than 50 inches per year.

TABLE 4-4 (Cont.)
Potential Effects to Plants as a Result of Surface Runoff

Herbicide	Effects to Terrestrial TEP plants	Effects to Non-target Aquatic Plants
Imazapyr	Clay soils: adverse effects where annual precipitation is greater than 10 inches. Loam soils: adverse effects where annual precipitation is greater than 50 inches.	No adverse effects predicted.
Metsulfuron methyl	Clay soils: adverse effects where annual precipitation is greater than 10 inches. Loam soils: adverse effects where annual precipitation is greater than 50 inches.	No adverse effects predicted.
Overdrive®	Clay soils: adverse effects where annual precipitation is greater than 10 inches. Silt loam, silt, and clay loam soils: adverse effects where annual precipitation is greater than 25 inches.	Clay soils: adverse effects where annual precipitation is greater than 10 inches per year. Sand, silt loam, silt, and clay loam soils: adverse effects where annual precipitation is greater than 25 inches per year. Loam soils: adverse effects where annual precipitation is greater than 150 inches per year (maximum application rates only).
Picloram	Clay soils: adverse effects where annual precipitation is greater than 10 inches. Loam soils: adverse effects where precipitation is between 50 and 200 inches per year.	No adverse effects predicted.
Sulfometuron methyl	Clay soils: adverse effects where annual precipitation is greater than 5 inches per year. Silt loam, silt, and clay loam soils: adverse effects where precipitation is greater than 25 inches per year.	Clay soils: adverse effects where annual precipitation is greater than 5 inches. Sand soils: adverse effects where annual precipitation is greater than 10 inches per year. Loam soils: adverse effects where annual precipitation is greater than 25 inches.
Tebuthiuron	Clay, silt loam, silt, and clay loam soils: adverse effects where annual precipitation is greater than 25 inches per year.	Sand: adverse effects where annual precipitation is greater than 5 inches. Other soil types: adverse effects where annual precipitation is greater than 10 inches.
Triclopyr acid	Clay and loam soils: adverse effects where annual precipitation is greater than 20 inches. Sand: adverse effects where annual precipitation is greater than 25 inches.	No adverse effects predicted.
Triclopyr BEE	Clay soils: adverse effects where annual precipitation is greater than 10 inches per year. Loam and sand soils: adverse effects where annual precipitation is greater than 5 inches.	Adverse effects under certain site conditions (e.g., in areas with clay soils and moderate to high annual rainfall).

The buffer distances provided below are conservative estimates, based on the information provided by ERAs, and are designed to provide protection to TEP plants at the programmatic-level. Some ERAs used regression analysis to predict the smallest buffer distance to ensure no risks to TEP plants. In most cases, where regression analyses were not performed, suggested buffers extend out to the first modeled distance from the application site for which no risks were predicted. In some instances the jump between modeled distances was quite large (e.g., 100 feet to

900 feet). Regression analyses could be completed at the local level using the interactive spreadsheets developed for the ERAs, using information in ERAs and for local site conditions (e.g., soil type, annual precipitation, vegetation type, and treatment method), to calculate more precise, and possibly smaller buffers for some herbicides.

2,4-D

- Because the risks associated with this herbicide were not assessed, do not spray within ½ mile of terrestrial plant species or aquatic habitats where TEP aquatic plant species occur.
- Do not use aquatic formulations in aquatic habitats where TEP aquatic plant species occur.
- Assess local site conditions when evaluating the risks from surface water runoff to TEP plants located within ½ mile downgradient from the treatment area.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Bromacil

- Do not apply within 1,200 feet of terrestrial TEP plant species.
- If using a low boom at the typical application rate, do not apply within 100 feet of an aquatic habitat in which TEP plant species occur.
- If using a low boom at the maximum application rate or a high boom, do not apply within 900 feet of an aquatic habitat in which TEP plant species occur.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Chlorsulfuron

- Do not apply by ground methods within 1,200 feet of terrestrial TEP species.
- Do not apply by aerial methods within 1,500 feet of terrestrial TEP species.
- Do not apply by ground methods within 25 feet of aquatic habitats where TEP plant species occur.
- Do not apply by aerial methods at the maximum application rate within 300 feet of aquatic habitats where TEP plant species occur.
- Do not apply by aerial methods at the typical application rate within 100 feet of aquatic habitats where TEP plant species occur.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Clopyralid

- Since the risks associated with using a high boom are unknown, use only a low boom during ground applications of this herbicide within ½ mile of terrestrial TEP plant species or aquatic habitats in which TEP plant species occur.
- Do not apply by ground methods at the typical application rate within 900 of terrestrial TEP species.
- Do not apply by ground methods at the typical application rate within ½ mile of terrestrial TEP species.
- Do not apply by aerial methods within ½ mile of terrestrial TEP species.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Dicamba

- If using a low boom at the typical application rate, do not apply within 1,050 feet of terrestrial TEP plant species.
- If using a low boom at the maximum application rate, do not apply within 1,050 feet of terrestrial TEP plant species.
- If using a high boom, do not apply within 1,050 feet of terrestrial TEP plant species.
- Do not apply within 25 feet of aquatic habitats where TEP plant species occur.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Diflufenzopyr

- If using a low boom at the typical application rate, do not apply within 100 feet of terrestrial TEP plant species.
- If using a high boom, or a low boom at the maximum application rate, do not apply within 900 feet of terrestrial TEP plant species.
- If using a high boom, do not apply within 500 feet of terrestrial TEP plant species.
- Do not apply within 25 feet of aquatic habitats where TEP plant species occur.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Diquat

- Do not use in aquatic habitats where TEP aquatic plant species occur.
- Do not apply by ground methods within 1,000 feet of terrestrial TEP species at the maximum application rate.
- Do not apply by ground methods within 900 feet of terrestrial TEP species at the typical application rate.
- Do not apply by aerial methods within 1,200 feet of terrestrial TEP species.

Diuron

- Do not apply within 1,100 feet of terrestrial TEP species.
- If using a low boom at the typical application rate, do not apply within 900 feet of aquatic habitats where TEP aquatic plant species occur.
- If using a high boom, or a low boom at the maximum application rate, do not apply within 1,1000 feet of aquatic habitats where TEP aquatic plant species occur.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Fluridone

- Since effects on terrestrial TEP plant species are unknown, do not apply within ½ mile of terrestrial TEP species.

Glyphosate

- Since the risks associated with using a high boom are unknown, use only a low boom during ground applications of this herbicide within ½ mile of terrestrial TEP plant species.
- Do not apply by ground methods at the typical application rate within 50 feet of terrestrial TEP plant species.
- Do not apply by ground methods at the maximum application rate within 300 feet of terrestrial TEP plant species.
- Do not apply by aerial methods within 300 feet of terrestrial TEP plant species.

Hexazinone

- Since the risks associated with using a high boom or an aerial application are unknown, only apply this herbicide by ground methods using a low boom within ½ mile of terrestrial TEP plant species and aquatic habitats that support aquatic TEP species.
- Do not apply by ground methods at the typical application rate within 300 feet of terrestrial TEP plant species or aquatic habitats that support aquatic TEP plant species.
- Do not apply by ground methods at the maximum application rate within 900 feet of terrestrial TEP plant species or aquatic habitats that support aquatic TEP plant species.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Imazapic

- Do not apply by ground methods within 25 feet of terrestrial TEP species or aquatic habitats where TEP plant species occur.
- Do not apply by helicopter at the typical application rate within 25 feet of terrestrial TEP plant species.
- Do not apply by helicopter at the maximum application rate, or by plane at the typical application rate, within 300 feet of terrestrial TEP plant species.

- Do not apply by plane at the maximum application rate within 900 feet of terrestrial TEP species.
- Do not apply by aerial methods at the maximum application rate within 300 feet of aquatic TEP species.
- Do not apply by aerial methods at the typical application rate within 100 feet of aquatic TEP species.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Imazapyr

- Since the risks associated with using a high boom are unknown, use only a low boom for ground applications of this herbicide within ½ mile of terrestrial TEP plant species or aquatic habitats in which TEP plant species occur.
- Do not apply at the typical application rate, by ground or aerial methods, within 900 feet of terrestrial TEP plant species or aquatic habitats in which aquatic TEP species occur.
- Do not apply at the maximum application rate, by ground or aerial methods, within ½ mile of terrestrial TEP plant species or aquatic habitats in which aquatic TEP species occur.
- Do not use aquatic formulations in aquatic habitats where TEP aquatic plant species occur.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Metsulfuron Methyl

- Since the risks associated with using a high boom are unknown, use only a low boom for ground applications of this herbicide within ½ mile of terrestrial TEP plant species or aquatic habitats in which TEP plant species occur.
- Do not apply at the typical application rate, by ground or aerial methods, within 900 feet of terrestrial TEP plant species or aquatic habitats in which aquatic TEP species occur.
- Do not apply at the maximum application rate, by ground or aerial methods, within ½ mile of terrestrial TEP plant species or aquatic habitats in which aquatic TEP species occur.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Overdrive®

- If using a low boom at the typical application rate, do not apply within 100 feet of terrestrial TEP plant species.
- If using a low boom at the maximum application rate, do not apply within 900 feet of terrestrial TEP plant species.
- If using a high boom, do not apply within 900 feet of terrestrial TEP plant species.
- Do not apply within 25 feet of aquatic habitats where TEP plant species occur.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Picloram

- Do not apply by ground or aerial methods, at any application rate, within ½ mile of terrestrial TEP plant species.
- Assess local site conditions when evaluating the risks from surface water runoff to TEP plants located within ½ mile downgradient from the treatment area.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Sulfometuron Methyl

- Do not apply by ground or aerial methods within 1,500 feet of terrestrial TEP species.
- Do not apply by ground methods within 900 feet of aquatic habitats where TEP plant species occur, or by aerial methods within 1,500 feet of aquatic habitats where TEP plant species occur.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Tebuthiuron

- If using a low boom at the typical application rate, do not apply within 25 feet of terrestrial TEP plant species.

- If using a low boom at the maximum application rate or a high boom at the typical application rate, do not apply within 50 feet of terrestrial TEP plant species.
- If using a high boom at the maximum application rate, do not apply within 900 feet of terrestrial TEP plant species.
- Do not apply within 25 feet of aquatic habitats where TEP plant species occur.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Triclopyr Acid

- Since the risks associated with using a high boom are unknown, use only a low boom during ground applications of this herbicide within ½ mile of terrestrial TEP plant species.
- Since the risks associated with using a high boom are unknown, use only a low boom during ground applications at the maximum application rate of this herbicide within ½ mile of aquatic habitats in which TEP plant species occur.
- Do not apply by ground methods at the typical application rate within 300 feet of terrestrial TEP plant species.
- Do not apply by aerial methods at the typical application rate within 500 feet of terrestrial TEP plant species.
- Do not apply by ground or aerial methods at the maximum application rate within ½ mile of terrestrial TEP plant species or aquatic habitats in which TEP plant species occur.
- If applying to aquatic habitats in which aquatic TEP plant species occur, do not exceed the targeted water concentration on the product label.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

Triclopyr BEE

- Since the risks associated with using a high boom are unknown, use only a low boom for ground applications of this herbicide within ½ mile of terrestrial TEP plant species or aquatic habitats in which TEP plant species occur.
- Do not apply by ground methods at the typical application rate within 300 feet of terrestrial TEP plant species or aquatic habitats in which TEP plant species occur.
- Do not apply by aerial methods at the typical application rate within 500 feet of terrestrial TEP plant species or aquatic habitats in which TEP plant species occur.
- Do not apply by ground or aerial methods at the maximum application rate within ½ mile of terrestrial TEP plant species or aquatic habitats in which TEP plant species occur.
- Do not use aquatic formulations in aquatic habitats where TEP aquatic plant species occur.
- In areas where wind erosion is likely, do not apply within ½ mile of TEP plant species.

The information provided in Table 4-1 provides a general guideline as to the types of habitats in which treatments (particularly fire) may be utilized to improve growing conditions for TEP plant species. However, at the local level, the BLM must make a further determination as to the suitability of vegetation treatments for the populations of TEP species that are managed by local offices. The following information should be considered: the timing of the treatment in relation to the phenology of the TEP plant species; the intensity of the treatment; the duration of the treatment; and the tolerance of the TEP species to the particular type of treatment to be used. When information about species tolerance is unavailable or is inconclusive, local offices must assume a negative effect to plant populations, and protect those populations from direct exposure to the treatment in question.

Treatment plans must also address the presence of and expected impacts on noxious weeds on the project site. These plans must be coordinated with BLM weed experts and/or appropriate county weed supervisors to minimize the spread of weeds. In order to prevent the spread of noxious weeds and other unwanted vegetation in occupied or suitable habitat, the following precautions should be taken:

- Cleared areas that are prone to downy brome or other noxious weed invasions should be seeded with an appropriate seed mixture to reduce the probability of noxious weeds or other undesirable plants becoming established on the site.

- Where seeding is warranted, bare sites should be seeded as soon as possible after treatment, and at a time of year when it is likely to be successful.
- In suitable habitat for TEP species, non-native species should not be used for revegetation.
- Certified noxious weed seed free seed must be used in suitable habitat, and preference should be given to seeding appropriate plant species when rehabilitation is appropriate.
- Straw and hay bales used for erosion control in suitable habitat must be certified weed- and seed-free.
- Vehicles and heavy equipment used during treatment activities should be washed prior to arriving at a new location to avoid the transfer of noxious weeds.

When BAs are drafted at the local level for treatment programs, additional conservation measures will be added to this list. Where BLM plans that consider the effects of vegetation treatments on TEP plant species already exist, these plans should be consulted, and incorporated (e.g., any guidance or conservation measures they provide) into local level BAs for vegetation treatments.

Effects Summary

Using the assumption that any of the proposed vegetation treatments could occur anywhere on public lands, the proposed treatment program **is likely to adversely affect** any and all of the TEP plant species and/or their critical habitat listed in Tables 1-1 and 4-1. However, provided the proper precautions were taken at the local level during the formulation of treatment programs, impacts to these species could be avoided. At a minimum, the guidance and conservation measures provided in the previous section, Conservation Measures, must be followed to reduce the likelihood for impacts to these species. However, additional conservation measures would also need to be developed by local offices and incorporated into site-specific BAs in order to ensure a determination of **not likely to adversely affect** at the local level.

CHAPTER 5

AQUATIC ANIMALS

Background Information

This BA considers a total of 78 fish species (including subspecies and Evolutionary Significant Units [ESUs]), 13 mollusks, and 7 aquatic arthropods that are listed as threatened or endangered, or that are proposed for listing. Background information is presented, species by species, in the section that follows. For the most part, species accounts have been arranged by ecoregion. However, some species (e.g., salmonids) travel over a wide geographic area to complete their life history cycles, and therefore may fall into a number of different ecoregions. These species have been discussed separately.

Most of the information contained in this section was obtained directly from Federal Register documents, species recovery plans, biological assessments and evaluations, and other sources of information. Where primary reference(s) was/were used for species background and listing information, full citations are listed in the individual sections for each species. In some instances, citations were used from the primary reference(s), and the complete citations were not available from the primary reference(s) for inclusion in the Bibliography (Chapter 8). In the instances where complete citations were not available, information is listed in the individual sections on where there complete citations can be found (e.g., USFWS Sacramento Field Office, Sacramento, California). If information is not listed on the location of complete citations from the primary reference(s), then the complete citation can be found in the Bibliography.

Marine/Anadromous Species

Fish species that migrate to the ocean to complete a portion of their life cycle are presented here, independent of the ecoregion divisions that are used to group freshwater species. Most of these species migrate through several ecoregions during the completion of their life history cycle.

The primary references for this section are:
National Marine Fisheries Service. No Date. Endangered Species Act Status Reviews and Listing Information. Available at: www.nwr.noaa.gov.

and

Washington State Joint Natural Resources Cabinet. 1999. Statewide Strategy to Recover Salmon. Olympia, Washington.

The life cycles of salmonids vary widely. However, common habitat requirements exist for all species. Freshwater salmonid habitat consists of four major components: habitat for spawning and incubation, juvenile rearing habitat, juvenile and adult migration corridors, and adult holding habitat. Estuarine and marine nearshore areas provide habitats for estuarine and ocean rearing, and for juvenile and adult migration.

Two of the most important features of freshwater habitat for spawning, rearing, and migration are a sufficient quantity of water, and good quality water. Salmon require cool, clean water that is of sufficient depth and velocity to allow passage, migration, and spawning, where floods do not scour channels. In addition, they seek out slow velocity areas adjacent to faster water for feeding, resting, and growing. Temperature affects growth rates and the timing of life history events, and turbidity and sediments can affect the abundance of food, as well as impact spawning and incubation habitats. Salmon also require a high level of dissolved oxygen, and are affected by other chemical aspects of the water.

Salmon life cycles are very sensitive to changes in stream flow, and have adapted over thousands of years to the natural flow regime in their individual watersheds. Natural low flows are important for the establishment of vegetation along stream banks. High flows add gravel, flush sediments from gravel, create new rearing channels, and perform other important functions.

Within the stream channel, salmon require sufficient clean and appropriately sized cobbles and gravel for spawning and incubation. Riffles, rapids, pools, and floodplain connectivity are important for production, rearing, cover, and aeration. Riparian vegetation provides shade, moderates the temperature of the stream, stabilizes banks, and controls soil erosion and sedimentation. It also provides nutrients to the stream and contributes large woody debris, which increases channel complexity, creates backwater habitats, and increases the water depth of pools. Aquatic plants and organic litter provide food for salmon, and can be influenced by riparian vegetation, temperature, streamflow, and substrate. Finally, salmon require unobstructed access both downstream and upstream for migration and feeding. Factors that obstruct passage include physical structures, inadequate streamflow, and high temperatures.

Nearshore marine habitats (e.g., marine tidal marshes, tidal channels, eelgrass beds, and kelp beds) provide salmon with spawning, rearing, and feeding grounds and shelter. They also protect the shoreline from erosion, filter pollutants, and reduce flooding by retaining stormwater during high-flow periods. Estuaries are important habitats for anadromous salmon transitioning from juvenile to adult, and from fresh to salt water and back again. Salmon pass through estuaries as juveniles on their downstream migration to the ocean, and as adults on their upstream migration to spawn. Some species are also dependent on estuaries as rearing areas. There are several important features of estuarine and marine habitats: water quality, especially temperature; adequate food and cover; a saltwater/fresh water transition zone; marine vegetation and algae; adequate river or stream discharge; and migration pathways.

The NOAA Fisheries uses the term ESU to refer to any distinct group of salmon populations, and to further clarify the meaning of subspecies under the ESA. Each salmonid species under the jurisdiction of NOAA Fisheries is divided into several ESUs for the purposes of management, protection, and listing under the ESA.

Coho Salmon

Historically, coho salmon (*Oncorhyncus kisutch*) were distributed throughout the North Pacific Ocean, from Central California to Point Hope, Alaska, through the Aleutian Islands, and from the Anadyr River, Russia south to Hokkaido, Japan. The species probably once inhabited most coastal streams in Washington, Oregon, and northern California. Some populations, now considered extinct, are believed to have migrated hundreds of miles inland to spawn in tributaries of the upper Columbia River in Washington and the Snake River in Idaho. There are six distinct ESUs of coho salmon along the West Coast of the United States, three of which are listed and occur in the project area: Central California, Southern Oregon/Northern California Coast, and Oregon Coast.

Coho salmon are anadromous; adults migrate from a marine environment into the freshwater streams and rivers of the birth. The species spawns only once, and then dies. Coho spend approximately the first half of their life cycle rearing in streams and small freshwater tributaries. The remainder of their life cycle is spent foraging in estuarine and marine waters of the Pacific Ocean, prior to returning to their stream of origin to spawn and die. Most fish return to spawn at 3 years old, although some precocious males may do so at 2 years of age.

Central California Coast

The Central California Coast ESU was federally listed as threatened on October 31, 1996. This ESU includes all naturally-spawned populations of coho salmon from Punta Gorda in northern California south to and including the San Lorenzo River in central California, as well as populations in tributaries to San Francisco Bay, excluding the Sacramento-San Joaquin River system. Critical habitat for this ESU, which was designated on May 5, 1999, includes all accessible river reaches from Punta Gorda to the San Lorenzo River, including Mill Valley (Arroyo Corte Medare Del Presidio) and Corte Maders creeks, which are tributaries to San Francisco Bay. Excluded from this designation are areas above specific dams or above longstanding, naturally impassable barriers, such as natural waterfalls. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 4,152 square miles in California. The following counties lie partially or wholly within these basins: Lake, Marin, Mendocino, San Mateo, Santa Clara, Santa Cruz, and Sonoma.

Southern Oregon/Northern California Coasts

The Southern Oregon/Northern California Coasts ESU was federally listed as threatened on May 6, 1997. This ESU includes all naturally spawned populations occurring in coastal streams between Cape Blanco, Oregon and Punta Gorda, California. Critical habitat (designated on May 5, 1999) includes all accessible reaches within this range, with the exception of areas above specific dams or above longstanding, naturally impassable barriers. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 18,090 square miles in California and Oregon. Counties that lie partially or wholly within watersheds inhabited by this ESU include Del Norte, Glenn, Humboldt, Lake, Mendocino, Siskiyou, and Trinity counties in California, and Coos, Curry, Douglas, Jackson, Josephine, and Klamath counties in Oregon.

Oregon Coast

The Oregon Coast ESU was federally listed as threatened on August 10, 1998. This ESU includes all naturally spawned populations occurring in Oregon coastal streams south of the Columbia River and north of Cape Blanco. Critical habitat for this species (designated on February 16, 2000) has been withdrawn and is being re-evaluated by NOAA Fisheries. The major river basins that contain spawning and rearing habitat for this ESU comprise approximately 10,606 square miles in Oregon. A number of Oregon counties lie partially or wholly within these basins, or contain migration habitat for the species: Benton, Clatsop, Columbia, Coos, Curry, Douglas, Josephine, Lane, Lincoln, Polk, Tillamook, Washington, and Yamhill.

Chinook Salmon

Chinook salmon (*Oncorhynchus tshawtscha*) are found from the Bering Strait south to Southern California. Historically, they ranged as far south as the Ventura River in California. There are 17 ESUs of chinook salmon along the west coast of the United States, which range from southern California to the Canadian border and east to the Rocky Mountains. In the project area, there are eight listed ESUs: Sacramento Winter-run; Snake River Fall-run; Snake River Spring/Summer-run; Lower Columbia River; Upper Willamette River; Upper Columbia River Spring-run; Central Valley Spring-run; and California Coastal.

Chinook salmon are the largest of any salmon, with adults often exceeding 40 pounds. Like coho salmon, they are anadromous and spawn only once before dying. Chinook salmon stocks exhibit considerable variability in size and age of maturation, at least some of which is genetically determined. The relationship between size and length of migration may also reflect the earlier timing of river entry and the cessation of feeding for salmon stocks that migrate to the upper reaches of river systems. Body size, which is correlated with age, may be an important factor in migration and the successful construction of redds (spawning beds).

There are different seasonal runs of chinook salmon, which correspond to the timing of migration from ocean to freshwater. These runs have been identified on the basis of when adults enter freshwater to begin their spawning migration. However, distinct runs also differ in the degree of maturation at the time of river entry, the thermal regime and flow characteristics of their spawning site, and their actual time of spawning.

Adult female chinook prepare spawning beds in stream areas with suitable gravel composition, water depth, and velocity. The female then lays eggs, which she guards for a brief period before dying. Eggs hatch between 90 and 150 days after deposition, depending on water temperatures. The following spring, young salmon fry emerge, and may spend from 3 months to 2 years in freshwater before migrating to estuarine areas as smolts, and then into the ocean to feed and mature. Chinook salmon remain at sea for 1 to 6 years, with the exception of a small number of yearling males that mature in freshwater, or return after 2 to 3 months in salt water.

There are two distinct races of chinook salmon: stream-type and ocean-type. Stream-type chinook have a longer freshwater residency and perform extensive offshore migrations before returning to their natal streams in the spring and summer months. Ocean-type chinook, which are commonly found in coastal streams, typically migrate to sea within the first 3 months of emergence, but may spend up to a year in fresh water prior to emigration. They also spend their ocean life in coastal waters, utilizing estuaries and coastal areas more extensively for juvenile rearing.

Sacramento Winter Run

The Sacramento River winter-run chinook salmon was federally listed as threatened on November 5, 1990 and then reclassified as an endangered species on January 4, 1994. This ESU includes populations of winter-run chinook salmon in the Sacramento River and its tributaries in California.

On June 16, 1993, NOAA National Marine Fisheries Service (NMFS) designated critical habitat for the winter-run chinook from Keswick Dam (Sacramento river mile 302) to the Golden Gate Bridge. The designated habitat includes the area from the Sacramento River at Keswick Dam downstream to the San Francisco Bay. The open ocean was considered important, but was not designated as critical habitat because degradation of the open ocean did not appear to have substantially contributed to the decline of the species. The essential features of the critical habitat include 1) the river water; 2) the river bottom, including those areas used as spawning substrate; 3) the adjacent riparian zone used for rearing; and 4) the estuarine water column and essential foraging habitat and food resources of the Delta and Bay, used for juvenile emigration and adult upmigration. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 9,329 square miles in California. The following counties lie partially or wholly within these basins: Butte, Colusa, Contra Costa, Glenn, Napa, Nevada, Placer, Plumas, Sacramento, Shasta, Solano, Sutter, Tehama, Trinity, Yolo, and Yuba.

Snake River Fall Run

The Snake River Fall-run ESU was federally listed as a threatened species on April 22, 1992. This ESU includes all natural populations occurring in the mainstem Snake River and any of the following subbasins: Tucannon River, Grande Ronde River, Imnaha River, Salmon River, and Clearwater River.

Critical habitat (designated on December 28, 1993) includes all river reaches presently or historically accessible (except reaches above impassable natural falls, and Dworshak and Hells Canyon dams) in the Columbia River, from a straight line connecting the west end of the Clatsop jetty (south jetty, Oregon side) and the west end of the Peacock jetty (north jetty, Washington side). Critical habitat also includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence of the Columbia and Snake rivers. On the Snake River, all reaches from the confluence of the Columbia River, upstream to Hells Canyon Dam are included. Also included are the Palouse River from its confluence with the Snake River upstream to Palouse Falls; the Clearwater River from its confluence with the Snake River upstream to its confluence with Lolo Creek; and the North Fork Clearwater River from its confluence with the Clearwater River upstream to Dworshak Dam. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 13,679 square miles in Idaho, Oregon, and Washington. The following counties lie partially or wholly within these basins: Idaho—Adams, Clearwater, Idaho, Latah, Lemhi, Lewis, and Nez Perce; Oregon—Baker, Union, and Wallowa; and Washington—Adams, Asotin, Columbia, Franklin, Garfield, Walla Walla, and Whitman.

Snake River Spring/Summer Run

The Snake River Spring/Summer-run ESU was federally listed as a threatened species on April 22, 1992. Included in this ESU are all natural populations occurring in the mainstem Snake River and in the subbasins of the Tucannon River, Grande Ronde River, Imnaha River, and Salmon River.

Critical habitat (designated on December 28, 1993) is similar to that for the Snake Fall-run ESU, except that stretches of the Palouse River, Clearwater River, and the North Fork Clearwater are not included. There are a total of 22,390 square miles of major river basins containing spawning and rearing habitat for this ESU in Idaho, Oregon, and Washington. The following counties lie partially or wholly within these basins: Idaho—Adams, Blaine, Custer, Idaho, Lemhi, Lewis, Nez Perce, and Valley; Oregon—Baker, Umatilla, Union, and Wallowa; and Washington—Adams, Asotin, Columbia, Franklin, Garfield, Walla Walla, and Whitman.

Lower Columbia River

The Lower Columbia River ESU was federally listed as threatened on March 24, 1999. Included in this ESU are all naturally-spawned populations occurring in the Columbia River and its tributaries, from its mouth at the Pacific Ocean upstream to a transitional point between Washington and Oregon east of the Hood River and the White Salmon River This ESU also includes populations in the Willamette River to Willamette Falls, Oregon, exclusive of spring-run chinook salmon in the Clackamas River.

On August 15, 2005, NOAA Fisheries filed the final critical habitat designation for this species in Clackamas, Clatsop, Columbia, Hood River, Multnomah, Wasco counties in Oregon; and Clark, Cowlitz, Klickitat, Lewis, Pacific, Pierce, Skamania, Wahkiakum, and Yakima counties in Washington. Major river basins that contain spawning and rearing habitat for this ESU comprise approximately 6,338 square miles in Oregon and Washington. There are approximately 1,311 stream miles and 33 square miles of lake habitat within this ESU that is designated as critical habitat. The following counties lie partially or wholly within these basins, or contain migration habitat for the ESU: Oregon—Clackamas, Clatsop, Columbia, Hood River, Marion, Multnomah, Wasco, and Washington; and Washington—Clark, Cowlitz, Klickitat, Lewis, Pierce, Pacific, Skamania, Wahkiakum, and Yakima.

Upper Willamette River

The Upper Willamette River chinook salmon ESU was federally listed as threatened on March 24, 1999. This ESU includes all naturally-spawned populations occurring in the Clackamas River and in the Willamette River, and its tributaries, above Willamette Falls, Oregon.

NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 1,472 stream miles and 18 square mile of lake habitat has been designated as critical habitat in this ESU. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 8,575 square miles. The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Oregon—Benton, Clackamas, Clatsop, Columbia, Douglas, Lane, Lincoln, Linn, Marion, Multnomah, Polk, Tillamook, Washington, and Yamhill; and Washington—Clark, Cowlitz, Pacific, and Wahkiakum.

Upper Columbia River Spring Run

The Upper Columbia River Spring-run ESU was federally listed as threatened on March 24, 1999. Included in this ESU are all naturally-spawned populations occurring in all accessible river reaches in Columbia River tributaries upstream of the Rock Island Dam and downstream of Chief Joseph Dam in Washington, excluding the Okanogan River. Chinook salmon (and their progeny) from the following hatchery stocks are considered part of the listed ESU: Chiwawa River (spring run); Methow River (spring run); Twisp River (spring run); Chewuch River (spring run); White River (spring run); and Nason Creek (spring run).

NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 974 stream miles and 4 square miles of lake habitat has been designated as critical habitat in this ESU. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 7,003 square miles in Oregon and Washington. The following counties lie partially or wholly within these basins (or contain migration corridors for the species): Oregon—Clatsop, Columbia, Hood River, Gilliam, Morrow, Sherman, Umatilla, and Wasco; and Washington—Benton, Chelan, Clark, Cowlitiz, Douglas, Franklin, Grant, Klickitat, Kittitas, Multnomah, Okanogan, Pacific, Skamania, Wahkiakum, Walla Walla, and Yakima.

Central Valley Spring Run

The Central Valley Spring-run ESU of chinook salmon was federally listed as a threatened species on September 16, 1999. This ESU includes all naturally-spawned populations occurring in the Sacramento River and its tributaries in California.

NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 1,158 stream miles and 254 square miles of estuarine habitat has been designated as critical habitat in Tehama, Butte, Glenn, Shasta, Yolo, Sacramento, Solano, Colusa, Yuba, Sutter, Trinity, Alameda, San Joaquin, and Contra Costa counties, California. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 9,329 square miles in California. The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Alameda, Butte, Colusa, Contra Costa, Glenn, Marin, Napa, Nevada, Placer, Sacramento, San Francisco, San Mateo, Shasta, Solano, Sonoma, Sutter, Tehama, Yolo, and Yuba.

California Coast

The California Coast ESU of chinook salmon was federally listed as threatened on September 16, 1999. This ESU includes all naturally spawned populations occurring in rivers and streams south of the Klamath River to the Russian River, California.

NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 1,475 stream miles and 25 square miles of estuarine habitat has been designated as critical habitat in Humboldt, Trinity, Mendocino, Sonoma, Lake, Napa, Glenn, Colusa, and Tehama counties, California. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 8,061 square miles in California. The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Glenn, Humboldt, Lake, Marin, Mendocino, Sonoma, and Trinity.

Chum Salmon

Chum salmon (*Oncorhynchus keta*) have the widest natural geographic and spawning distribution of any Pacific salmonid, with a range that extends farther along the shores of the Arctic Ocean. Historically, chum salmon were distributed as far south as Monterey, California. Presently, however, major spawning populations are found only as far south as Tillamook Bay on the northern Oregon coast. There are four ESUs of chum salmon along the west coast of the United States, one of which is found in the project area: the Columbia River ESU.

Like coho and chinook salmon, chum salmon are anadromous and spawn only once before dying, primarily in fresh water. They spawn in the lowermost reaches of rivers and streams, typically within about 60 miles of the ocean. Unlike most other salmonids, they migrate almost immediately after hatching to estuarine and ocean waters. Therefore, the survival and growth of juveniles depends less on freshwater conditions than on favorable estuarine and marine conditions. Another behavioral difference between chum salmon and most species that rear extensively in fresh water is that chum salmon form schools, presumably to reduce predation. Most chum salmon mature at between 3 and 5 years of age. The species has only a single form (sea-run) and does not reside in fresh water.

The Columbia River ESU was federally listed as threatened on March 25, 1999. This ESU includes all naturally spawned populations occurring in the Columbia River and its tributaries in Washington and Oregon.

NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 708 stream miles has been designated as critical habitat. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 4,426 square miles in Oregon and Washington. The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Oregon—Clatsop, Columbia, Multnomah, and Washington; and Washington—Clark, Cowlitz, Lewis, Pacific, Skamania, and Wahkiakum.

Sockeye Salmon

Sockeye salmon (*Oncorhyncus nerka*) on the Pacific coast inhabit riverine, marine, and lake environments, from the Columbia River and its tributaries north and west to the Kuskokwim River in western Alaska. There are seven ESUs of sockeye salmon along the west coast of the United States, two of which are federally listed. Of these, only the endangered Snake River ESU is found within the project area.

Like other salmon species, sockeye are anadromous; however, there are non-anadromous life forms of this species. Sockeye salmon exhibit a wide variety of life history patterns that reflect varying dependency on the freshwater environment. With the exception of certain river-type and sea-type populations, the vast majority of sockeye salmon spawn in or near lakes, where the juveniles rear for 1 to 3 years prior to migrating to sea. For this reason, the major distribution and abundance of large sockeye salmon stocks are closely related to the location of rivers that have accessible lakes in their watersheds for juvenile rearing. Occasionally, a proportion of the juveniles in an anadromous sockeye salmon population will remain in their rearing lake environment throughout life, and will be observed on the spawning grounds together with their anadromous siblings.

The Snake River ESU of sockeye salmon was federally listed as endangered on November 20, 1991. This ESU includes populations of sockeye salmon from the Snake River Basin, Idaho (extant populations occur in the Stanley River subbasin).

Critical habitat (designated on December 28, 1993) includes presently or historically accessible river reaches (except reaches above impassable natural falls, and Dworshak and Hells Canyon dams) in the Columbia River, from a straight line connecting the west end of the Clatsop jetty (south jetty, Oregon side) and the west end of the Peacock jetty (north jetty, Washington side) and including all Columbia River estuarine areas and river reaches upstream to the confluence of the Columbia and Snake Rivers. Also included are all Snake River reaches from the confluence of the Columbia River upstream to the confluence of the Salmon River; all Salmon River reaches from the confluence of the Snake River upstream to Alturas Lake Creek; Stanley, Redfish, Yellow Belly, Pettit, and Alturas lakes (including their inlet and outlet creeks); Alturas Lake Creek, and the portion of Valley Creek between Stanley Lake Creek and the Salmon River. Watersheds containing spawning and rearing habitat for this ESU comprise approximately 510 square miles in Idaho. The watersheds lie partially or wholly within Blaine and Custer counties.

Steelhead

Along the west coast, steelhead trout (*Oncorhynchus mykiss*) are distributed across about 15 degrees of latitude from the U.S. Canada border south to the mouth of Malibu Creek, California. In some years, steelhead may be found as far south as the Santa Margarita River in San Diego County. There are 10 listed steehead ESUs, 8 of which are found in the project area: Central California Coast, Upper Columbia River, Snake River Basin, Lower Columbia River, California Central Valley, Upper Willamette, Middle Columbia River, and Northern California.

Steelhead have the greatest diversity of life history patterns of any Pacific salmonid species, including varying degrees of anadromy, differences in reproductive biology, and plasticity of life history between generations. Within the range of West Coast steelhead, spawning migrations occur throughout the year, with seasonal peaks of activity. In any given river basin there may be one or more peaks of migration activity; some rivers may have multiple runs, and fish are divided into either winter, spring, summer, or fall run steelhead. North American steelhead commonly spend 2 years in the ocean before entering fresh water to spawn. Summer steelhead enter fresh water up to a year prior to spawning. Steelhead may spawn more than once. In some cases, the separation between anadromous steelhead and rainbow or redband trout is obscured.

Southern California

The Southern California ESU was federally listed as endangered species on August 18, 1997. This ESU includes all naturally spawned populations of steelhead (and their progeny) in streams from the Santa Maria River to Malibu

Creek, California (inclusive). NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 708 stream miles has been designated as critical habitat in this ESU.

South-central California Coast

The South-central California Coast ESU was federally listed as threatened on August 18, 1997. This ESU includes all naturally-spawned populations of steelhead (and their progeny) in streams from the Pajaro River (inclusive) to, but not including the Santa Maria River, California. NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 1,250 stream miles and 3 square miles of estuarine habitat has been designated as critical habitat in this ESU.

Central California Coast

The Central California Coast ESU was federally listed as threatened on August 18, 1997. This ESU includes all naturally-spawned populations of steelhead (and their progeny) in California streams from the Russian River to Aptos Creek, and the drainages of San Francisco and San Pablo Bays eastward to the Napa River (inclusive), excluding the Sacramento-San Joaquin River Basin. NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 1,465 stream miles and 386 square miles of estuarine habitat has been designated as critical habitat. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 6,516 square miles in California. The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Alameda, Contra Costa, Marin, Mendocino, Napa, San Francisco, San Mateo, Santa Clara, Santa Cruz, Solano, and Sonoma.

Upper Columbia River

The Upper Columbia River ESU was federally listed as endangered on August 18, 1997. This ESU occurs in streams in the Columbia River Basin upstream from the Yakima River, Washington, to the U.S.-Canada border. Wells Hatchery stock steelhead are also part of the listed ESU. NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 1,262 stream miles and 7 square miles of lake habitat has been designated as critical habitat. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 9,545 square miles in Oregon and Washington. The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Oregon—Clatsop, Columbia, Gilliam, Hood River, Morrow, Multnomah, Sherman, Umatilla, and Wasco; and Washington—Benton, Chelan, Clark, Cowlitz, Douglas, Franklin, Gilliam, Grant, Kittitas, Klickitat, Okanogan, Pacific, Skamania, Wahkiakum, Walla Walla, and Yakima.

Snake River

The Snake River ESU of steelhead was federally listed as threatened on August 18, 1997. This ESU occurs in streams in the Snake River Basin of southeast Washington, northeast Oregon, and Idaho. NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 8,049 stream miles and 4 square miles of lake habitat has been designated as critical habitat. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 29,282 square miles in Idaho, Oregon, and Washington. The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Idaho—Adams, Blaine, Boise, Clearwater, Custer, Idaho, Latah, Lemhi, Lewis, Nez Perce, and Valley; Oregon—Baker, Clatsop, Columbia, Hood River, Morrow, Multnomah, Sherman, Umatilla, Union, Wallowa, and Wasco; and Washington—Asotin, Benton, Clark, Columbia, Cowlitz, Franklin, Garfield, Gilliam, Klickitat, Skamania, Wahkiakum, Walla Walla, and Whitman.

Lower Columbia River

The Lower Columbia River ESU was federally listed as threatened on March 19, 1988. This ESU occurs in streams and tributaries to the Columbia River between the Cowlitz and Wind rivers, Washington (inclusive) and the

Willamette and Hood rivers, Oregon (inclusive). Excluded are steelhead in the upper Willamette River Basin above Willamette Falls and steelhead from the Little and Big White Salmon rivers in Washington. NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 2,324 stream miles and 27 square miles of lake habitat has been designated as critical habitat. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 5,017 square miles in Oregon and Washington. The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Oregon—Clackamas, Clatsop, Columbia, Hood River, Marion, Multnomah, and Washington; and Washington—Clark, Cowlitz, Lewis, Pacific, Skamania, and Wahkiakum.

Central Valley, California

The Central Valley, California, ESU was federally listed as threatened on March 19, 1998. This ESU occurs in the Sacramento and San Joaquin rivers and their tributaries. Excluded are steelhead from San Francisco and San Pablo bays and their tributaries. NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 2,308 stream miles and 254 square miles of estuarine habitat has been designated as critical habitat. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 13,096 square miles in California. The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Alameda, Amador, Butte, Calaveras, Colusa, Contra Costa, Glenn, Marin, Merced, Nevada, Placer, Sacramento, San Francisco, San Joaquin, Shasta, Solano, Sonoma, Stanislaus, Sutter, Tehama, Tuolumne, Yolo, and Yuba.

Upper Willamette

The Upper Willamette ESU of steelhead was federally listed as threatened on March 25, 1999. This ESU includes all naturally-spawned populations of winter-run steelhead in the Willamette River, Oregon, and its tributaries upstream from Willamette Falls to the Calapooia River, inclusive. NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 1,276 stream miles and 2 square miles of lake habitat has been designated as critical habitat. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 4,872 square miles in Oregon and Washington. The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Oregon—Benton, Clackamas, Clatsop, Columbia, Lincoln, Linn, Marion, Multnomah, Polk, Tillamook, Washington, and Yamhill; and Washington—Clark, Cowlitz, Pacific, and Wahkiakum.

Middle Columbia River

The Middle Columbia River ESU was federally listed as threatened on March 25, 1999. This ESU occurs in streams from above the Wind River, Washington, and the Hood River, Oregon (exclusive), upstream to, and including, the Yakima River, Washington. Excluded are steelhead from the Snake River Basin. NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 5,815 stream miles has been designated as critical habitat. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 26,739 square miles in Oregon and Washington. The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Oregon—Clatsop, Columbia, Crook, Gilliam, Grant, Harney, Hood River, Jefferson, Morrow, Multnomah, Sherman, Umatilla, Union, Wallowa, Wasco, and Wheeler; and Washington—Benton, Clark, Columbia, Cowlitz, Franklin, Kittitas, Klickitat, Pacific, Skamania, Wahkiakum, Walla Walla, and Yakima.

Northern California

The Northern California ESU was federally listed as threatened on June 7, 2000. This ESU occurs in California coastal river basins from Redwood Creek south to the Gualala River, inclusive. NOAA Fisheries filed final critical habitat designation for this species on August 15, 2005. Approximately 3,028 stream miles and 25 square miles of estuarine habitat has been designated as critical habitat. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 6,672 square miles in California. The following counties lie partially or wholly within these basins: Del Norte, Glenn, Humboldt, Lake, Mendocino, Sonoma, and Trinity.

Threats to Pacific Salmon

Salmonid species on the West Coast of the United States have experienced dramatic declines in abundance during the past several decades as a result of human-induced and natural factors. Water storage, withdrawal, conveyance, and diversions for agriculture, flood control, domestic, and hydropower purposes have greatly reduced or eliminated historically accessible habitat and/or resulted in direct entrainment mortality of juvenile salmonids. Modification of natural flow regimes has resulted in increased water temperatures; changes in fish community structures; and a depletion of the flows necessary for migration, spawning, rearing, flushing of sediments from spawning gravels, gravel recruitment, and transport of large woody debris. Physical features of dams, such as turbines and sluiceways, have resulted in increased mortality of both adults and juvenile salmonids. Attempts to mitigate adverse impacts of these structures have, to date, met with limited success.

Natural resource use and extraction leading to habitat modification can have substantial direct and indirect impacts to salmon populations. Land use activities associated with logging, road construction, urban development, mining, agriculture, and recreation have significantly altered fish habitat quantity and quality. Impacts associated with these activities include: alteration of streambanks and channel morphology; alteration of ambient stream water temperatures; degradation of water quality; reduction in available food supply; elimination of spawning and rearing habitat; fragmentation of available habitats; elimination of downstream recruitment of spawning gravels and large woody debris; removal of riparian vegetation resulting in increased stream bank erosion; and increased sedimentation input into spawning and rearing areas, resulting in the loss of channel complexity, pool habitat, suitable gravel substrate, and large woody debris. In most western states, about 80 to 90% of the historic riparian habitat has been eliminated. In has also been estimated that Washington and Oregon’s wetlands have been diminished by one third, and that California has experienced a 91% loss of its wetland habitat.

Other factors that have led to the decline of salmon and continue to threaten remaining populations include loss of spatial and temporal connectivity and complexity, recreational and commercial fishing, introduction of non-native species, and natural environmental conditions (e.g., floods, drought, climatic shifts) that exacerbate the problems associated with degraded and altered riverine and estuarine habitats.

Bull Trout

The primary reference for this section is:
USFWS. 1999h. Determination of Threatened Status for Bull Trout in the Coterminous United States Final Rule. Federal Register 64(210): 58909-58933.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Snake River Basin Field Office. Boise, Idaho.

Bull trout (*Salvelinus confluentus*) are native to the Pacific Northwest and western Canada. They historically occurred in major river drainages in the Pacific Northwest, from the southern limits in the McCloud River in northern California and the Jarbidge River in Nevada, north to the headwaters of the Yukon River in Northwest Territories, Canada (Cavender 1978, Bond 1992). To the west, the range of the bull trout includes the Puget Sound, and various coastal rivers of Washington, British Columbia, Canada, and southeast Alaska (Bond 1992, Leary and Allendorf 1997). Bull trout are relatively dispersed throughout tributaries of the Columbia River Basin, including its headwaters in Montana and Canada. Bull trout also occur in the Klamath River Basin of south-central Oregon. East of the Continental Divide, they are found in the headwaters of the Saskatchewan River in Alberta and the MacKenzie River system in Alberta and British Columbia (Cavender 1978, Brewin and Brewin 1997).

Bull trout exhibit both resident and migratory life-history strategies through much of their current range (Rieman and McIntyre 1993). Resident bull trout complete their life cycles in the tributary streams in which they spawn and rear. Migratory bull trout spawn in tributary streams, and juvenile fish rear from 1 to 4 years before migrating to either a lake (adfluvial), river (fluvial), or in certain coastal areas, saltwater (anadromous), to mature (Fraley and Shepard 1989, Goetz 1989). Anadromy is the least studied life-history stage in bull trout, and some biologists believe the existence of true anadromy in bull trout is still uncertain (McPhail and Baxter 1996). Resident and migratory forms may be found together, and bull trout may produce offspring exhibiting either resident or migratory behavior (Rieman and McIntyre 1993).

Compared to other salmonids, bull trout have more specific habitat requirements (Rieman and McIntyre 1993) that appear to influence their distribution and abundance. Critical parameters include water temperature, cover, channel form and stability, valley form, spawning and rearing substrates, and migratory corridors (Oliver 1979; Pratt 1984, 1992; Fraley and Shepard 1989; Goetz 1989; Hoelscher and Bjornn 1989; Sedell and Everest 1991; Howell and Buchanan 1992; Rieman and McIntyre 1993, 1995; Rich 1996; Watson and Hillman 1997). Watersheds must have specific physical characteristics to provide the necessary habitat requirements for bull trout spawning and rearing, although these characteristics are not necessarily ubiquitous throughout watersheds in which bull trout occur. Because bull trout exhibit a patchy distribution, even in undisturbed habitats (Rieman and McIntyre 1993), fish would not likely occupy all available habitats simultaneously (Rieman et al. 1997).

Bull trout are typically associated with the colder streams in a river system, although fish can occur throughout larger river systems (Fraley and Shepard 1989; Rieman and McIntyre 1993, 1995; Buchanan and Gregory 1997; Rieman et al. 1997). Spawning areas are often associated with cold-water springs, groundwater infiltration, and the coldest streams in a given watershed (Pratt 1992; Rieman and McIntyre 1993; Rieman et al. 1997). All life history stages of bull trout are associated with complex forms of cover, including large woody debris, undercut banks, boulders, and pools (Oliver 1979, Fraley and Shepard 1989, Goetz 1989, Hoelscher and Bjornn 1989, Sedell and Everest 1991, Pratt 1992, Thomas 1992, Rich 1996, Sexauer and James 1997, Watson and Hillman 1997). Maintaining bull trout populations requires stream channel and flow stability (Rieman and McIntyre 1993). Juvenile and adult bull trout frequently inhabit side channels, stream margins, and pools with suitable cover (Sexauer and James 1997). These areas are sensitive to activities that directly or indirectly affect stream channel stability and alter natural flow patterns.

Preferred spawning habitat generally consists of low gradient stream reaches, which are often found in high gradient streams that have loose, clean gravel (Fraley and Shepard 1989) and water temperatures of 41 to 48 °F in late summer to early fall (Goetz 1989). The size and age of maturity for bull trout is variable depending upon life-history strategy. Growth of resident fish is generally slower than that of migratory fish; resident fish tend to be smaller at maturity and less fecund (productive; Fraley and Shepard 1989, Goetz 1989). Bull trout normally reach sexual maturity in 4 to 7 years, and can live 12 or more years. Biologists report repeat and alternate year spawning, although repeat spawning frequency and post-spawning mortality are not well known (Leathe and Graham 1982, Fraley and Shepard 1989, Pratt 1992, Rieman and McIntyre 1996). Bull trout typically spawn from August to November during periods of decreasing water temperatures. However, migratory bull trout may begin spawning migrations as early as April, and move upstream as far as 155 miles to spawning grounds in some areas of their range (Fraley and Shepard 1989, Swanberg 1997). Depending on the water temperature, egg incubation is normally 100 to 145 days (Pratt 1992), and juveniles remain in the substrate after hatching. Fry normally emerge from early April through May, depending on water temperatures and increasing stream flows (Pratt 1992, Ratliff and Howell 1992).

Bull trout are opportunistic feeders, with food habits primarily a function of size and life-history strategy. Resident and juvenile bull trout prey on terrestrial and aquatic insects, macro-zooplankton, amphipods, mysids, crayfish, and small fish (Wyman 1975, Rieman and Lukens 1979 *cited in* Rieman and McIntyre 1993, Boag 1987, Goetz 1989, Donald and Alger 1993). Adult migratory bull trout are primarily piscivorous, known to feed on various trout and salmon species, whitefish, yellow perch and sculpin (Fraley and Shepard 1989, Donald and Alger 1993).

The bull trout was federally listed as threatened throughout its entire range in the coterminous United States on November 1, 1999. On October 6, 2004, approximately 1,748 miles of streams and 61,235 acres of lakes and reservoirs in Oregon, Washington, and Idaho were designated as critical habitat for the Klamath River and Columbia River populations of bull trout. However, the USFWS is currently re-evaluating this designation. The decline of bull trout is primarily attributable to habitat degradation and fragmentation, blockage of migratory corridors, poor water quality, past fisheries management practices, and the introduction of non-native species.

Temperate Desert Ecoregion

The Temperate Desert Ecoregion is a cool desert region, with low precipitation and a relatively high elevation. This region more or less corresponds to the Great Basin and the Colorado Plateau. Much of this area is made up of separate interior basins; only a small part of it drains to the sea (Bailey 1995). The lower parts of many basins have heavy accumulations of alkaline and saline salts. Streams are rare and few are permanent. Important aquatic habitats include terminal lakes (e.g., Mono Lake and the Great Salt Lake), marshes, or sinks that are warm and saline (Moyle 1976). The northern half of this ecoregion division also includes portions of the Snake, Columbia, Yakima, and Platte rivers.

Foskett Speckled Dace
The Foskett speckled dace (*Rhinichthys osculus* ssp.) is endemic to Foskett Spring in south-central Oregon, a small spring system in the Coleman Basin on the west side of Warner Valley. Habitat is a small springhole and overflow rivulets that occur in what appears to be mixed rangeland at the edge of an alkali playa. The wet areas at the spring, along the course of the rivulets, and at the sump on the edge of the playa supports grasses and some aquatic vegetation, including cattails. The main population is in the springhole, which is about 6 feet in diameter and mostly 6 to 12 inches deep. Individuals also live in tiny outflow rivulets that are at times only a few inches wide and deep. Some are found in cattle tracks into which water seeps continuously (Bond 1974). Cover utilized includes overhanging bank edges, grass, exposed grass roots, and filamentous algae. Water in the spring is clear, and the current is slow. The bottom is primarily mud. The dace has also been introduced into Dace Spring, an excavated area at a spring source located on public land about 1 mile south of Foskett Spring. This artificial habitat is muddy and well-vegetated (Armantrout 1985). Although individuals have been collected from shallow water habitats associated with filamentous algae, exposed grass roots, and emergent aquatic vegetation, this habitat is not believed to be optimal. Based on conditions under which other speckled dace live, it is likely that deeper water with moderate vegetative cover would be better habitat.

The Foskett speckled dace appears to feed primarily on invertebrates. Extensive migration is not known, but larval and early juvenile dace have been observed only in the marsh at the edge of the lake bed (Hayes 1980), so there is either a migration of adults downstream to spawn, or a migration of the hatched larvae from the spring hole or rivulets to the marsh (a distance of about 6 to 12 feet). Like other dace populations, it is likely that the Foskett speckled dace requires some kind of hard substrate for egg deposition (Moyle 1976). Reproduction apparently occurs in the second year of age, and spawning is believed to occur between late May and early July (Hayes 1980).

The Foskett speckled dace was federally listed as threatened on March 28, 1985. Critical habitat has not been designated. The subspecies apparently became isolated in Foskett Spring about 9,000 to 10,000 years ago, when Lake Warner went dry (Hubbs and Miller 1948a). Its main natural habitat has been overrun by vegetation or heavily trampled by cattle. Future perceived threats are essentially the same as the past reasons for decline, although the dace population seems to have stabilized to a point compatible with present use of the area by cattle. A spring to which the dace was transplanted by the BLM is fenced to exclude cattle (Armantrout and Bond 1981), and the main threat at this site is the encroachment of vegetation (cattails and possible rushes), and the resulting decrease in dissolved oxygen. Pumping of groundwater or channelization (via heavy equipment, such as a backhoe) at either site could impact the habitat as well (USFWS 1985i). Both springs that contain the dace are in a known geothermal area, so there is also a potential future threat of energy development.

Warner Sucker
The primary reference for this section is:
USFWS. 1998l. Recovery Plan for the Native Fishes of the Warner Basin and Alkali Subbasin. Portland, Oregon.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The Warner sucker (*Catostomus warnerensis*) is endemic to the Warner Basin of southeastern Oregon. The probable historic range of this species includes the main Warner Lakes (Pelican, Crump, and Hart), and other accessible standing or flowing water in the Warner Valley, as well as the low-to-moderate gradient reaches of the tributaries that drain into the valley. Studies conducted between 1977 and 1991 indicate that when adequate water is present, Warner suckers may inhabit all the lakes, sloughs, and potholes in the Warner Valley. Stream resident populations are found in Honey Creek, Snyder Creek, Twentymile Creek, and Twelvemile Creek.

There are two phenotypic variations, or morphs of the Warner sucker, which correspond to the two generally continuous aquatic habitat types provided by the Warner Basin. Stream morphs occur in the temporally stable stream environments, and lake morphs occur in the temporally less stable lake environments. Individual fish can opportunistically change from one morph to another based on the types of habitat that are available. The exact nature of the relationship between lake and stream morphs is not well studied, and remains poorly understood.

The feeding habitats of the Warner sucker depend to a large degree on habitat and life history stage, with adult suckers becoming more generalized than juveniles and young-of-year. Larvae have terminal mouths and short digestive tracts, enabling them to feed selectively in midwater or at the surface. Invertebrates, particularly planktonic crustaceans, make up most of their diet. As the suckers grow, they develop subterminal mouths and longer digestive tracts, and gradually become benthic feeders, eating diatoms, filamentous algae, and detritus. Adult stream morph suckers forage nocturnally over a wide variety of substrates, such as boulders, gravel, and silt. Adult lake morph suckers are thought to have a similar diet, though they feed over predominantly muddy substrates (Tait and Mulkey 1993a, b).

Spawning usually occurs in April and May in streams, although variations in water temperature and stream flows may result in either earlier or later spawning. Temperature and flow cues appear to trigger spawning, with most taking place at 57 to 68 °F when stream flows are relatively high. Suckers spawn in sand or gravel beds in slow pools (White et al. 1990, 1991; Kennedy and North 1993). In years when access to stream spawning areas is limited by low flow or by physical in-stream blockages, suckers may attempt to spawn on gravel beds along the lake shorelines.

Larvae are found in shallow backwater pools or on stream margins where there is no current, often among or near macrophytes. Young-of-year are often found over still, deep water from midwater to the surface, but also move into faster flowing water near the heads of pools (Coombs et al. 1979). Juveniles (1 to 2 years old) are usually found at the bottom of deep pools or in other habitats that are relatively cool or permanent, such as near springs. In general, adults use stretches of streams where the gradient is sufficiently low to allow the formation of long (167 feet or longer) pools. These pools tend to have undercut banks, large beds of aquatic macrophytes, root wads or boulders, a surface to bottom temperature differential of at least 36 °F, a maximum depth greater than 5 feet, and overhanging vegetation.

The Warner sucker was federally listed as threatened on September 27, 1985, with critical habitat designated at the time of listing. Critical habitat for this species includes the following areas: 1) Twentymile Creek from the confluence of Twelvemile and Twentymile Creeks upstream for about 4 miles; 2) Twentymile Creek starting about 9 miles upstream of the confluence of Twelvemile and Twentymile Creeks and extending downstream for about 18 miles; 3) Spillway Canal north of Hart Lake and continuing about 2 miles downstream; 4) Snyder Creek from the confluence of Snyder and Honey Creeks upstream for about 3 miles; and 5) Honey Creek from the confluence of Hart Lake upstream 16 miles.

The Warner sucker is threatened by human-induced stream channel and watershed degradation; irrigation diversion practices that block its spawning migration routes and reduce stream flows below the points of diversion; and predation by and competition with non-native game fish such as crappie, bullhead catfish, and bass that were previously stocked in Warner Basin lakes.

June Sucker
The primary reference for this section is:
USFWS. 1999i. June Sucker (*Chasmistes liorus*) Recovery Plan. Denver, Colorado.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The June sucker is a lakesucker that is endemic to Utah Lake, which is located about 45 miles south of the Great Salt Lake, Utah. Historically, the species was found in Utah Lake throughout the year, and in Provo River and other tributaries to Utah Lake during its annual spawning migration. In addition to the small remaining Utah Lake population, five other locations have been stocked with June suckers for the purposes of study and increasing overall population numbers. The historic habitat of Utah Lake was typified by relatively stable water levels, which allowed for long-term maintenance of macrophyte beds. These macrophyte beds are commonly uses as nursery habitat by native fish species. Spawning habitat preferred by the June sucker is riverine habitat with braided, slow, meandering channels, providing a diversity of habitat conditions for different age-classes of fish.

Like other lakesuckers, the June sucker is thought to be a mid-water planktivore (Miller and Smith 1981; Scoppetone et al. 1986a). However, food habits of this species in Utah Lake are difficult to verify.

June suckers are thought to be a long-lived species, maturing at 5 to 10 years of age. During the reproductive period, beginning in April and May, adults concentrate in and around the mouth of the Provo River (Radant and Hickman 1984). In the second and third weeks of June, the spawning migration typically begins. The exact date of migration is dependent on environmental conditions. Most spawning is completed within a span of 5 to 8 days. After hatching, emergent June sucker larvae drift downstream in the river during nighttime hours (Modde and Muirhead 1990; Crowl and Thomas 1997; Keleher et al. 1998). During the larval stage, abundant aquatic vegetation is utilized for cover and refugia.

The June sucker was federally listed as endangered on April 30, 1986. Critical habitat, which was designated on the same date, includes the lower 4.9 miles of the main channel of the Provo River, from the Tanner Race diversion downstream to Utah Lake. Decline in abundance of June suckers can be attributed to habitat alteration through dewatering, channelization of tributary streams, and degradation of water quality; competition with and predation by non-native species; commercial fishing; and killing of adults during the spawning run.

Borax Lake Chub
The permanent habitat of the Borax Lake chub (*Gila boraxobius*) is a 10.2-acre thermal lake located in the Borax Lake Basin of Oregon. This lake, which is shallow and fed by hot and cool springs, is perched about 30 feet above the desert floor in a “pedestal” of deposited salts. The saline lake bottom is inhospitable to rooted plants, although some of the precipitated minerals are finely divided and silt-like. Irrigation channels have been dug from the lake to supply water for hay fields, and the chub may also be found in these channels. The chub is found in Lower Borax Lake, an artificial pond, when it has water in it. This habitat is highly alkaline, with murky water and little vegetation. If enough overflow water is received, marshes and temporary pools may also provide habitat for the chub. All of the Borax Lake chub’s known habitats in southeastern Oregon comprise approximately 640 acres.

The Borax Lake chub is an opportunistic omnivore (Hudson et al. 2000). Spawning can occur year-round, but primarily occurs in the spring. Substantial spawning activity and larval chubs have been observed during autumn, following the cessation of unusually hot spring inflows during the preceding months.

The Borax Lake chub was federally listed as endangered on October 5, 1982. Critical Habitat has been designated in Harney County, Oregon, and includes all 640 acres of habitat in Township 37 South, Range 33 East, including Borax Lake, marsh areas to the south and southwest, Lower Borax Lake, and hot springs north of Borax Lake. Because the lake depends upon several subterranean springs for its water supply, lowering the rim of the lake or tapping and diverting the springs could have severe effects upon the species. Borax Lake is in a known geothermal resource area, and both diversion and geothermal exploration appear to constitute a threat to the species.

Hutton Tui Chub
The following information, taken from Moyle (1976), refers to tui chubs in general. Tui chubs occur in a wide variety of habitats, most commonly in the weedy shallows of lakes and quiet waters in sluggish rivers. They do well in a wide variety of water conditions from warm to cold, and clear to eutrophic. In the fall, they seek out deeper water and may spend winters in a semi-dormant state on the bottom of lakes. Tui chubs are opportunistic omnivores concentrating on invertebrates associated with bottom or aquatic plants (i.e., clams, insect larvae, insects, crayfish), as well as algae and plant material.

Tui chub usually spawn from late April to late June; eggs adhere to plants or the bottom and hatch in 9 days. In large deep lakes, they tend to form large schools in shallow water frequently associated with beds of aquatic vegetation. In shallow lakes, with heavy aquatic growth, schooling is less noticeable. Tui chubs tend to disperse amongst the vegetation, presumably as protection from predators. They also appear to be able to adapt to the severe long and short-term climatic fluctuations characteristic of the interior basins where they are most common. The minnow family in general has been successful because they have a well-developed sense of hearing, release a fear scent when injured (a warning signal to others), have a broad diet, and exhibit high fecundity. Despite these advantages, many native minnows are declining in numbers as their environment deteriorates beyond their ability to cope with the changes or they are displaced by more aggressive introduced species.

The Hutton tui chub (*Gila bicolor* ssp.) is endemic to Hutton Spring and a nearby unnamed spring in Lake County, south-central Oregon (NatureServe Explorer 2001). These springs are located in a grassy rangeland bordered to the north and west by shrubby rangeland and to the east and south by the lake bed of pluvial Alkali Lake.

The Hutton tui chub was federally listed as threatened on March 28, 1985. Critical habitat has not been designated. The current isolation of the Hutton tui chub was caused by the desiccation of pluvial Alkali Lake (Snyder 1908a, Hubbs and Miller 1942). Present status is in part a result of past access by cattle to the springs in which the Hutton tui chub occurs (Franzreb 1985). Threats include pumping of water from the springs, which occurred in the past but is no longer occurring (Bond 1974, Franzreb 1985), and contamination of groundwater by dispersal of chemicals from a nearby herbicide-manufacturing residue disposal site (Franzreb 1985). Modification of the springs by heavy equipment (causing siltation, erosion, vegetation cover loss, water diversion and drawdown) has also had detrimental effects on the chub population.

Cowhead Lake Tui Chub
The primary reference for this section is:
USFWS. 1998m. Proposed Endangered Status for the Cowhead Lake Tui Chub. Federal Register 63(60): 15152-15158.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the Sacramento USFWS Office, Sacramento, California.

The Cowhead Lake tui chub (*Gila bicolor vaccaceps*) is found in the vicinity of Cowhead Lake, a lake in the extreme northeastern corner of Modoc County, California, in an area known as the Modoc Plateau. The volcanic rock characteristic of this area is porous, causing most of the rainfall to percolate through into the groundwater, which surfaces as springs. Cowhead Slough and Cowhead Lake are fed mainly by snowmelt runoff and springs via Eightmile Creek and other smaller tributaries from the Warner Mountains. There may also be several faults at the upper end of the slough that provide subsurface flow (Sato 1992). The entire current estimated range of this species is approximately 3.4 miles of Cowhead Slough and connected ditches within the bed of Cowhead Lake.

The habitat type is sagebrush steppe, which is generally a treeless, shrub-dominated community characterized by sagebrush with perennial bunch grasses in the understory and some juniper (Young et al. 1988). Cold, harsh winters, dry summers, and low rainfall characterize the area.

Approximately one half of this subspecies’ range is on public land. The other half of the range is on land that has been under private ownership since the 1950s. The lakebed of Cowhead Lake is approximately 2,700 acres, with an elevation of 5,241 feet. Approximately 40% of the lakebed occurs on privately-owned land, and the remaining 60% has unknown title, based on a title search done in 1997 (Modoc County Title Co. 1997). The lake went dry sometime in the 1930s. Since the drought ended, the lake has been mechanically pumped dry so that the lakebed can be used to grow hay. There are a series of irrigation ditches, two reservoirs on nearby creeks, and a mechanical pumping system, which have modified the hydrology of the Cowhead Basin. There have been no formal studies on the life history or habitat of the Cowhead Lake tui chub.

The Cowhead Lake tui chub was proposed for federal listing as an endangered species on March 30, 1998. This subspecies is threatened throughout its range by a variety of human impacts, including the dewatering of Cowhead Lake, livestock grazing, agricultural activities, and by random naturally occurring events.

Owens Tui Chub

The Owens tui chub (*Gila bicolor snyderi*) is associated with streams in the Owens Valley of California that have slow current, mud bottoms, and abundant submerged vegetation. The Owens Basin consists of three valleys; Long Valley and Adobe Valley in the north drain into Owens Valley to the south. The Owens Basin contains a variety of springs, lakes and flowing water habitats, many of which were inhabited by the Owens tui chub at one time. However, much of the natural habitat has been modified for irrigation purposes, impounded to create reservoirs, or dewatered to provide for the needs of Los Angeles.

The Owens tui chub was federally listed as endangered on August 5, 1985. Critical habitat has been designated for this species in portions of Hot Creek (Section 35, Township 35 South, Range 28 East) and Owens River (Sections 19 through 25 and 36, Township 4 South, Range 3O East) in Mono County, California. Known constituent elements include high quality, cool water with adequate cover of rocks, undercut banks or aquatic vegetation, and a sufficient insect food base. Over the past 3 to 4 decades, habitats have been modified, streams have been diverted, and rivers have been dammed, as a result of increased water demands (Williams 1985). Introduction of exotic fishes has also been a major factor in the decline of the Owens tui chub. Predation by trout has impacted populations, and hybridization with the Lahontan tui chub has occurred extensively throughout the Owens Basin. Future threats include: 1) any substantial alterations of the habitats where the chubs still occur, 2) introduction of closely related fish species that may hybridize with the Owens tui chubs, and possibly 3) geothermal development.

Aquatic Snails of the Snake River

The primary reference for this section is:
USFWS. 1995c. Snake River Aquatic Species Recovery Plan. Snake River Basin Office, Ecological Services. Boise, Idaho.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Utah valvata snail (*Valvata utahensis*), Idaho springsnail (*Fontelicella idahoensis*), Snake River physa snail (*Physa natricina*), Bliss Rapids snail (*Taylorconcha serpenticola*), and Banbury Springs limpet (*Lanx* sp.) are part of the native mollusk fauna of the Snake River of Idaho, which characteristically require cold, fast water or lotic habitats. These five species are unique in that, unlike many other mollusk species, which are widely distributed and somewhat tolerant of pollution, they are primarily limited to the Snake River basin below American Falls Dam, and are generally intolerant of pollution. The locations and habitats of each of these species are described below. Very little is known about their life history.

Utah Valvata Snail

The Utah valvata snail inhabits areas between sand and silt/mud grains, in shallow, shoreline water and in pools adjacent to rapids or in perennial flowing waters associated with large spring complexes. The species avoids areas with heavy currents or rapids. It prefers well-oxygenated areas of limestone mud or mud-sand substrate among beds of submergent aquatic vegetation. It is absent from pure gravel-boulder substrate.

The Utah valvata snail is primarily a detritivore, grazing along the mud surface ingesting diatoms or small plant debris. In habitats with boulders on mud, the snail has been observed grazing aquatic plants, and diatoms and other sessile organisms that live attached to rocky surfaces.

Historically, the Utah valvata snail occurred in Utah Lake and the Snake River of southern Idaho (Taylor 1987a). However, recent surveys throughout Utah revealed no live snails, and the species is believed to be extirpated there (Clarke 1991a). At present, the Utah valvata snail occurs in a few springs and mainstem Snake River sites in the Hagerman Valley. Additional locations include a few sites immediately upstream and downstream of Minidoka Dam, near Eagle Rock dam site and below American Falls downstream to Burley.

Idaho Springsnail

The Idaho springsnail is a Lake Idaho endemic, and was historically found from Homedale to Bancroft Springs, Idaho. At present, the species is discontinuously distributed in the mainstem Snake River at a few sites near the C. J. Strike Reservoir upstream to Bancroft Springs, a reduction of nearly 80% from its historic range. This species has declined in numbers and the remaining populations are small and fragmented.

The Idaho Springsnail is found only in permanent flowing waters of the mainstem Snake River; the snail is not found in any of the Snake River tributaries or in marginal cold-water springs (Taylor 1982). The species is an interstitial dweller that occurs on mud or sand with gravel-to-boulder size substrate.

Snake River Physa Snail

The Snake River physa snail occurs on the undersides of gravel-to-boulder sized substrate in swift current in the mainstem Snake River. Living specimens have been found on boulders in the deepest accessible part of the river at the margins of rapids. The historic range of this species is believed to have extended from Grandview through the Hagerman Reach (Taylor 1988). As of 1995, two populations (or colonies) were believed to remain in the Hagerman and King Hill reaches, with possibly a third colony immediately downstream of Minidoka Dam.

Bliss Rapids Snail

The Bliss Rapids snail occurs on stable cobble-to-boulder sized substrate in flowing water of unimpounded reaches of the mainstem Snake River, and in a few spring habitats in the Hagerman Valley. The species does not burrow in sediments and normally avoids surfaces with attached plants. Known river populations of the Bliss Rapids snail occur only in areas associated with spring influences or rapids-edge environments, and tend to flank shorelines. They are found at varying depths if dissolved oxygen and temperature requirements persist, and are found in shallow (less than 0.5 inches deep), permanent cold springs (Frest and Johannes 1992). The species is considered moderately intolerant of light, and resides on the lateral sides and undersides of rocks during daylight (Bowler 1990). The species can be locally quite abundant, especially on smooth rock surfaces with common encrusting red algae.

The Bliss Rapids snail was known historically from the mainstem Snake River and associated springs between Indian Cove Bridge and Twin Falls (Hershler et al. 1994). Based on live collections, the species currently exists as discontinuous populations within its historic range. These colonies are primarily concentrated in the Hagerman Reach, in tailwaters of Bliss and Lower Salmon dams, and several unpolluted streams: Thousand Springs, Banbury Springs, Box Canyon Springs, and Niagara Springs.

Banbury Springs Limpet

The Banbury Springs limpet has been found only in spring-run habitats with well-oxygenated, clear, cold (59 to 61 °F) waters, on boulder or cobble-size substrate. All known locations have relatively swift currents. They are found most often on smooth basalt, and avoid surfaces with large aquatic macrophytes or filamentous green algae. The species may be found in water as shallow as 2 inches, but is most common at depths up to 6 inches (Frest and Johannes 1992). All limpets are particularly affected by dissolved oxygen fluctuations, since respiration is accomplished only through the mantle; lungs, gills, and other specialized respiratory structures are lacking. At present, the Banbury Springs limpet is known to occur only in the largest, least disturbed spring habitats at Banbury Springs, Box Canyon Springs, and Thousand Springs, Idaho.

The Snake River physa snail, Banbury Springs limpet, Utah valvata snail, and Idaho springsnail were federally listed as endangered on December 14, 1992. The Bliss Rapids snail was federally listed as threatened on the same date. Critical habitat has not been designated for any of these species. With the advent of exploration and development, the Snake River ecosystem has undergone a substantial transformation from a primarily free-flowing, cold-water system to a slower-moving, warmer system. The human-induced environmental stressors to the Snake River include numerous point and nonpoint pollution sources, diversion of water for irrigation or hydropower, and construction of several mainstem dams. Therefore, threats to these species include activities that deplete oxygen or reduce water quality, such as agricultural runoff; and activities that cause changes or fluctuations in water level, such as impoundments, pumping, or water diversion projects. Competition from introduced snail species is also a threat.

Bruneau Hot Springsnail

The Bruneau hot springsnail (*Pyrgulopsis bruneauensis*) is another aquatic snail that is restricted to the state of Idaho. It is a thermal species, restricted to the lower reaches of Hot Spring, a tributary of the Bruneau River in the southwestern portion of the state. This species inhabits thermal springs and seeps that range in temperature from 60 to 98.5 °F, and grazes on the algae and diatoms on the floor of the riverbed (Pacific Biodiversity Institute 2002). Its complete range includes a 5-mile portion of the Bruneau River and the lower third of Hot Spring. Most of the occupied springs are located along the Bruneau River at the confluence of and upstream of Hot Creek, on lands administered by the BLM. Some additional springsnail habitats located downstream of the Indian Bathtub and Hot Creek are on privately-owned land.

Bruneau hot springsnails are found on the exposed surfaces of various substrates, including rocks, gravel, sand, mud, and algal film, within geothermal habitats (Mladenka 1992). However, during the winter period of cold ambient temperatures and icing, the springsnails are most often located on the underside of flow substrates, habitats that are exposed least to cold temperatures. Reproduction occurs throughout the year, except when limited by high or low water temperatures. Sexual maturity occurs at approximately 2 months. Bruneau hot springsnails are dioecious (reproductive organs are located in separate male and female specimens), and lay single eggs on hard surfaces, such as rock substrates or the shells of other snails.

The Bruneau hot springsnail was listed as endangered on January 25, 1993. Critical habitat has not been designated. The species is threatened by the reduction and/or loss of geothermal habitats caused by the depletion of the regional geothermal aquifer underlying the Bruneau Valley area (Hudson et al. 2000). Within the past 30 years, discharge from many of the geothermal springs along Hot Creek and the Bruneau River has either ceased flowing or has exhibited a much reduced flow, thus restricting springsnail habitat (Young et al. 1979; Mladenka 1992; Berenbock 1993; Mladenka and Minshall 1996; Myler and Minshall 1998). Introduced predators, flash floods, and grazing also impact this species.

Lahontan Cutthroat Trout

The primary reference for this section is:

Hudson, B., J. Augsburger, M. Hillis, and P. Boehne. 2000. Draft Biological Assessment for the Interior Columbia River Basin Ecosystem Management Project Final Environmental Impact Statement. BLM and Forest Service. Boise, Idaho.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Lahontan cutthroat trout (*Oncorhyncus clarki henshawi*) in the only trout native to the Lahontan subbasin of the American Great Basin, west-central Nevada. Historically, the subspecies was found in the Carson, Humboldt, Truckee, and Walker rivers, and in their tributary lakes and streams. Since the late 19th century, fluvial (stream) and lacustrine (lake) populations of the Lahontan cutthroat trout have been reduced to approximately 10.7% and 0.4% of their original habitat, respectively.

Lahontan cutthroat trout occupy a great variety of habitats, from large rivers and lakes to small tributary streams. They are unusually tolerant of both high temperatures (> 81 °F) and large daily fluctuations in temperature (up to 68 °F). In addition, they are tolerant of high alkalinity (>3,000 ppm) and dissolved solids (>10,000 ppm). However, they are intolerant of competition or predation by non-native salmonids (LaRivers 1962, Trotter 1987, Behnke 1992).

Lahontan cutthroat trout are obligate but opportunistic stream spawners. Typically, they spawn from April through July, depending on water temperature and flow characteristics, though autumn spawning runs have also been reported for some populations. Fish may spawn more than once, although post-spawning mortality rates of 60 to 90% have been reported. Lake residents migrate into streams to spawn, typically on well-washed gravels in riffles. Adults court, pair, and deposit and fertilize eggs in a spawning bed dug by the female, which may then be defended for some period of time.

The Lahontan cutthroat trout was federally-listed as threatened on July 16, 1975. Critical habitat has not been designated. The observed major decline in this species has been attributed to habitat loss, introgression with introduced rainbow trout, and competition with other introduced species of trout, such as brown and brook trout. Habitat loss and the adverse impacts of non-native fishes continue to be the primary threats to the Lahontan cutthroat trout (Coffin and Cowan 1995, Gerstrung 1998).

Desert Dace

The desert dace (*Eremichthys acros*) occurs in warm springs and their outflows, including small irrigation ditches, within the Soldier Meadows area of Nevada, a small, roughly circular basin approximately 5 to 6 miles in diameter. Outflows from the numerous small springs either terminate in marshy areas or coalesce into Mud Meadow Wash, which eventually terminates in the Black Rock Desert. Occupied habitats include spring pools up to 8 feet in depth, with peripheral emergent vegetation and little or no current, as well as small flowing natural channels and irrigation ditches with dense emergent vegetation (Hubbs and Miller 1948b, Nyquist 1963). The species prefers water temperatures between 73 and 84 °F. Temperatures of 70 to 75 °F are required for spawning, which apparently occurs throughout early and mid summer (Sigler and Sigler 1987), and possibly year round (Matthews and Moseley 1990). The dace is apparently primarily herbivorous in its feeding habits.

The desert dace was federally listed as threatened on December 10, 1985. Critical habitat has been designated in the thermal springs and their surrounding riparian areas for a distance of 50 feet from these springs and ouflows in Sections 5, 8, 18, and 19, Township 40 North, Range 25 East; Sections 23 through 26, Township 40 North, Range 24 East, of Humboldt County, Nevada. Many of the spring outflows have been diverted from their natural channels into man-made ditches for irrigation, domestic use, and providing water for livestock (La Rivers 1962, Nyquist 1963, USFWS 1985j). These diversions have reduced habitat available to the desert dace, and are expected to continue in the foreseeable future. Potential threats to the species include geothermal development, groundwater depletion, and introduction of exotic fishes.

Cui-ui

Cui-ui (*Chasmistes cujus*) are obligate lacustrine suckers with a very restricted distribution, occurring only in Pyramid Lake, Nevada. In the spring, adults migrate from Pyramid Lake up the lower Truckee River to reproduce, and return to Pyramid Lake immediately after spawning. Larvae emigrate to Pyramid Lake shortly after hatching (LaRivers 1962; Scoppettone et al. 1983; Sigler et al. 1985; Scoppettone et al. 1986). Habitat of adults in Pyramid Lake is the inshore benthic region. Few cui-ui juveniles have been collected from Pyramid Lake, and most were collected at depths less than 66 feet. Adult cui-ui spawn in Truckee River over predominately gravel substrate, at water depths ranging from 8 to 43 inches, and stream velocities ranging from 0.9 to 4.6 feet per second. Cui-ui spawning has also been reported in Pyramid Lake at the entrance of freshwater streams on fine to coarse gravel (Koch 1973, 1982) and in the Marble Bluff fishway where the substrate is predominately compacted soil (Scoppettone et al. 1986). Upstream migrating prespawning adults require pool environments, typically log jam pools, as refugia during the day (Scoppettone et al. 1981). Koch (1982) recommended a safe maximum temperature for adult cui-ui of 68 to 71 °F. Adults feed primarily on zooplankton, filamentous algae and aquatic insects (Nevada Fish and Game Commission 1958, LaRivers 1962).

The cui-ui was federally listed as endangered on March 11, 1967. Critical habitat has not been designated. The reproductive cycle of this species was blocked by the construction of the Derby Dam on the Truckee River in 1905. In the past, this species has also been impacted by channelization projects that removed protective cover, deep pools, and shade required by the species. Livestock have removed riparian vegetation and increased the potential for erosion. Factors perceived as future threats to this species include upstream passage of migrating adults over Marble Bluff Dam, adequate Truckee River flows for migrating adults and larvae, proper river water temperatures for incubating embryos and out-migrating larvae, stream bank and channel erosion, increases in Pyramid Lake salinity and Truckee River water quality, and nutrient loading to Pyramid Lake (Galat 1983; Sigler et al. 1985; Coleman 1986).

White River Spinedace

The White River spinedace (*Lepidomeda albivallis*) occurs in cool springs that represent remnant segments of the ancient White River of eastern Nevada. The species is presently found only in a three-spring system within the Kirch Wildlife Management Area, where predation by largemouth bass restricts it to a relatively unsuitable portion of the spring system. Habitats occupied by the White River spinedace are characterized by relatively cool temperatures (64 to 72 °F) and clear water (Miller and Hubbs 1960). Spinedace occur in both deep water source pools and shallower effluent streams, and may prefer areas with moderate to swift flows over gravel substrates (Miller and Hubbs 1960, LaRivers 1962). Aquatic vegetation found in springs inhabited by White River spinedace include pondweed and watercress, while rushes and cattails are abundant near shoreline areas (Miller and Hubbs 1960). The range of this species is restricted to the source pool and short sections of effluent stream at Lund Town Spring and Flag Springs.

Little is known about the food habits, reproductive characteristics, and life history of the White River spinedace. However, based on information for other spinedace species, diet is likely to consist primarily of aquatic and terrestrial insects, with some plant material and detritus being consumed (Minckley and Carufel 1967, Rinne 1971). Plant material, primarily filamentous algae, becomes a larger component of the diet when insect abundance is low.

Spawning in the closely related Virgin River spinedace, occurs in shallow tailout areas of pools over a substrate of fine gravel (Rinne 1971). The White River spinedace spawns throughout the summer (Minckley and Carufel 1967, Minckley 1973). Sexual maturity is reached after one year (Rinne 1971), and spawning takes place in fish that are 1 year or older.

The White River spinedace was federally listed as endangered on September 12, 1985. Critical habitat has been designated in the following springs and outflows, as well as surrounding land areas for a distance of 50 feet from the springs and outflows, in White Pine County: Preston Big Spring (Section 2, Township 12 North, Range 61 East) and Lund Spring (in portions of Section 4, Township 11 North, Range 62 East and Section 33, Township 12 North, Range 62 East); and Nye County: Flagg Springs (in portions of Sections 32 and 33, Township 7 North, Range 62 East), Nevada. Habitat deterioration has been attributed to channelization and piping of spring outflows, diversion of water from spring sources, and use of copper sulfate to control aquatic vegetation (Hardy 1980; Courtney et al. 1985; Williams et al. 1985). Further alteration of spring habitats in such a manner would thus be detrimental to existing spinedace populations. Introduced species, including guppies, and mosquitofish, compete with and in some instances prey on spinedace, and are present in one of the two locations where the White River spinedace now exists (Courtney et al. 1985). Any further modifications of spring habitats (e.g., channelization, water diversion, and reductions of water quality) where the White River spinedace occurs would bring about further population declines and possible extinction of this species (Hardy 1980). Additional introductions, or increases in existing populations, of exotic species would have similar negative effects (Hardy 1980, USFWS 1985k, Williams et al. 1985).

Clover Valley Speckled Dace and Independence Valley Speckled Dace

The primary reference for this section is:
USFWS. 1998n. Recovery Plan for the Endangered Speckled Dace of Clover and Independence Valleys (*Rhinichthys osculus lethoporus* and *Rhinichthys osculus oligoporus*). Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Clover Valley speckled dace (*Rhinichthys osculus oligoporus*) is restricted to three spring systems in the Clover Valley, located in Elko County, Nevada: Bradish Spring, Clover Valley Warm Springs, and Wright Spring Ranch. The Independence Valley speckled dace (*Rhinichthys osculus lethoporus*) is found only in the marsh of the largest spring system in Independence Valley, which is also located in Elko County, Nevada. This spring system is known as the Independence Valley Warm Springs.

No other freshwater fish occupies a more widely distributed or variety of habitats than the speckled dace species (Moyle 1976). They are found throughout all major western drainage systems from the Colorado River south to Sonora, Mexico. Speckled dace primarily inhabit cool, flowing, permanent streams and rivers, but are also successful in a variety of other habitats. Throughout their range, they are found primarily among rocks in riffles in streams and on rocky or sandy bottoms stirred by wave action in lakes.

Clover Valley speckled dace are found primarily in reservoirs and outflows of the three spring systems identified above. There do not appear to be any marshes associated with these springs, only the outflows that have been heavily modified. Details of the subspecies' seasonal habitat requirements, population size, distribution over time, reproductive potential, and available habitat are unknown.

Independence Valley speckled dace are found in a temperate, permanent desert stream/marsh fed by numerous springs. The subspecies is found primarily in the shallow waters of the marsh of this spring system, among the sedges and grasses. It is believed that speckled dace also historically occupied the stream, but were forced out by predation by non-native species. Currently, the subspecies inhabits a large portion of the marsh, as well as two seep areas northeast of the marsh.

Generally, speckled dace are characterized as diurnal (active during the daytime), bottom browsers that feed primarily on small invertebrates (such as aquatic insects), plant material, and zooplankton. However, they will also feed on large, flying insects at the water's surface, and occasionally on the eggs and larvae of other minnows when available. Seasonal diet changes have been noted (Jhingram 1948, Miller 1951); dace most often eat algae and detritus in the fall, bottom-dwelling insects in the winter and spring, and flying insects in the summer. Based on the habitat they occupy, the Clover Valley and Independence Valley speckled dace probably have similar food preferences.

Specific reproductive patterns of the two dace subspecies have not been examined. Generally, speckled dace mature in their second summer. They are capable of spawning throughout the summer, but peak activity usually occurs in the months of June and July at water temperatures of 65 °F (Moyle 1976, Sigler and Sigler 1979). Males congregate in spawning areas from which they remove debris to expose a bare patch of rock or gravel. Males surround the female when entering a spawning area. Eggs are deposited underneath rocks, into spaces in the gravel, or close to the bottom, and fertilized. Eggs hatch in 6 days on average, and the larval fish remain in the gravel for 7 to 8 days. After emerging from the gravel, the young tend to concentrate in the warm shallows of streams.

The Clover Valley speckled dace and the Independence Valley speckled dace were federally listed as endangered on October 10, 1989. Critical habitat has not been designated for either species. The primary factors that threaten these subspecies include irrigation, or other activities that modify habitat, and competition with and predation by non-native sport fishes. In addition, their small population sizes and limited distribution make them vulnerable to random occurrences.

White River Springfish and Hiko White River Springfish
The primary reference for this section is:
USFWS. 1998o. Recovery Plan for the Aquatic and Riparian Species of Pahranagat Valley. Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The White River springfish (*Crenicthys baileyi baileyi*) and Hiko White River springfish (*Crenicthys baileyi grandis*) are endemic to the Pahranagat Valley, located in south-central Lincoln County, Nevada, approximately 92 miles north of Las Vegas. White River springfish are currently restricted to a spring pool at Ash Springs, where the population has ranged from approximately 1,200 to 9,800 in the past 10 years (as of 1998). The fish are found throughout the pool, with infrequent occurrences in the outflow stream (Tuttle et al. 1990). Hiko White River springfish occupy the pools of Hiko and Crystal springs, and have been introduced into Blue Link Spring in Mineral County, Nevada.

The plant community of the Pahranagat Valley is typical of the Mojave Desert, and is dominated by the creosote bush-burroweed vegetation association (Kanim 1986). Livestock grazing is a principle land use in Pahranagat Valley, and pastures with a variety of grasses and legumes have been established in the valley bottom. Very little information is available on the life history and habitat requirements of either subspecies of White River springfish. However, it is assumed that this subspecies has a similar life history and habitat needs that are comparable to other *Crenicthys* subspecies. Adults are found at varying depths, from 1.3 to 5.6 feet, but prefer deeper water (3.6 feet). Juveniles will also use all depths, but generally occur in shallower (2.1 feet) water and are more vertically dispersed. Larval springfish restrict their movement to the top of the water column (0 to 2 feet), and are found most frequently at 1.1 feet. All age classes are present in areas of calm water (Tuttle et al. 1990).

White River springfish are feeding generalists (Deacon and Minckley 1974, Williams and Williams 1982, Wilde 1989). Invertebrates, especially amphipods (small crustaceans), appear to be important items in their diet (Wilde 1989). Springfish may also be highly herbivorous, ingesting filamentous algae, vascular plants, and diatoms (Williams and Williams 1982). Differences in diet probably result from differences in habitat that dictate food item availability. Herbivory may be most common in the winter when invertebrates are not abundant (Wilde 1989). Springfish forage along the substrate and in plants, as evidenced by the ingestion of bottom-dwelling invertebrates, plant fragments, and detritus. They are active only in the daytime, with peaks occurring in the morning and afternoon.

Both White River springfish subspecies are uniquely adapted for surviving in environments of extreme temperatures and low dissolved oxygen content (Hubbs and Hettler 1964). The ability of springfish to actively thermoregulate by moving in and out of areas of extreme temperatures, which would be lethal under extended exposure, and to live in water with a broad range of temperatures, has enabled them to survive in areas deemed too hostile for other fish species.

Springfish are asynchronous, which means that individual females will spawn at different times of the year. Most females average two spawning periods a year, while the spawning season of the entire population extends over a long period of time each year. Another subspecies of White River springfish spawns year-round, with peak spawning activity from April through August (Scoppettone et al. 1987). The period of spawning activity may be regulated by the primary productivity in the spring system (Schoenherr 1981).

The White River and Hiko White River springfish were federally listed as endangered on September 27, 1985. Critical habitat was designated for both subspecies at the same time. Critical habitat for the White River springfish includes Ash Springs, its outflow, and surrounding land areas for a distance of 50 feet from these areas. Critical habitat for the Hiko White River springfish includes the two springs historically occupied by the subspecies, along with their outflows and surrounding land areas for a distance of 50 feet from these springs. Fort both subspecies, constituent elements include warm water springs and their outflows and surrounding land areas that provide vegetation for cover and habitat for insects and other invertebrates on which the subspecies feed. Populations of both subspecies of springfish continue to face threats to their existence from the continued presence of non-native species, diseases not previously found in native fish populations, habitat manipulation, and loss of genetic material exchange between populations.

Railroad Valley Springfish
The primary reference for this section is:
USFWS. 1996e. Railroad Valley Springfish (*Crenichthys nevadae*) Recovery Plan. Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Railroad Valley Springfish (*Crenichthys nevadae*) is the only fish species native to the thermal spring systems of Railroad Valley, Nye County, Nevada. The species is uniquely adapted to survive in an environment of high water temperature (86 to 100 °F at the spring source) and low dissolved oxygen content (1.5 to 6.0 ppm). This combination of metabolic stresses is well beyond the tolerance levels of most other fish species (Hubbs and Hettler 1964). In their natural environment, Railroad Valley springfish will occupy habitats with water temperatures at the extremes of their tolerance limits (57 °F or 104 °F) for limited amounts of time. They adjust their body temperatures by moving in and out of areas where the water temperature would be lethal under extended exposure (Williams 1986).

Railroad Valley springfish are opportunistic feeders, ingesting a wide variety of foods (Williams 1986). There is evidence that the species is predominantly herbivorous during the spring, consuming primarily filamentous algae. By summer, the species shifts to carnivory, when animal foods, primarily seed shrimp, constitute a majority of the diet. Railroad Valley springfish have been observed diving into algal mats, as if for specific food items, and also drift feeding (Deacon et al. 1980).

Spawning in this species has never been observed, but it may be similar to that of the White River springfish.

The Railroad Valley springfish was federally listed as threatened on March 31, 1986. Critical habitat was designated for the species on the same date. Critical habitat includes the six springs that were historically occupied by the species, along with their pools, portions of the outflow streams and marshes, and a 50-foot riparian zone around all such areas. Constituent elements for critical habitats include clear, unpolluted thermal spring waters ranging in temperature from 84 to 97 °F in pools, flowing channels, and marshy areas with aquatic plants, insects, and mollusks. The historical populations have been impacted to various degrees by habitat loss and modification resulting from water diversion, non-native fish introductions, and groundwater depletion. The primary threats to the species are exotic fish species and activities that modify habitat, such as channelization and diversion of water, groundwater pumping, and oil exploration.

Pahranagat Roundtail Chub
The Pahranagat roundtail chub (*Gila robusta jordani*) is found only in Ash Springs, located in the Pahranagat Valley, Lincoln County, Nevada, and about 7,400 feet of its outflow. Below that point the flow is confined to a concrete irrigation ditch from which water can be diverted for use on crops and pasture. The Pahranagat roundtail chub is usually quite rare in the upper 6,400 feet of the outflow stream, but maintains good numbers of adults in a

single microhabitat in the lower portion of the natural channel (Hardy 1982). The lower section of the natural channel (from about 6,400 to 7,400 feet below Ash Springs) is a generally broad, straight channel. There are scattered dense stands of willow and grape along the stream margin with some ash and cottonwood. Root projections, fallen branches (and logs), and overhanging branches provide aquatic cover. The substrate is sand, silt and mud. Runs and pools comprise about 92 and 8% of the available habitat, respectively. There are no riffles in this segment. Stream gradient is low, banks are not well-defined and the channel is about 20 feet in width.

The relative scarcity of deep, slow run/pool habitats with associated cover may impose some limitation on population size in this last remaining habitat available to the species. Temperature throughout much of the summer remains above 81 °F throughout the available habitat.

In the outflow of Ash Spring the breeding season may occur in February and March when adults leave their sheltered pool (Hardy 1982). This period coincides with annual thermal minimum temperatures. Juveniles have been observed in the outflow from March through September, disappearing rapidly from the population during October through January. Adults, therefore, appear to live through at least 2 winters prior to spawning.

The Pahranagat roundtail chub was federally listed as endangered on October 13, 1970. Critical habitat has not been designated for this subspecies. Its present endangered status is a result of habitat loss and predation and competition with introduced exotic species. The species was extirpated from Crystal Springs, possibly as a result of the introduction of largemouth bass into the system. The subspecies appears to be presently threatened by having lost most of its stream habitats, adverse consequences of interaction with exotic fishes and snails, and loss of young to downstream intermittent habitats.

Big Spring Spinedace
The primary reference for this section is:
USFWS. 1993h. Big Spring Spinedace, *Lepidomeda mollispinis pratensis,* Recovery Plan. Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Big Spring spinedace (*Lepidomeda mollispinis pratensis*) is one of three native fishes occupying the stream habitat of Meadow Valley Wash in Lincoln County, Nevada. Big Spring spinedace are restricted to a 5-mile section of stream, which flows through privately-owned and public lands in Condor Canyon, north of Panaca, Nevada.

Habitat and life history requirements of the subspecies are poorly understood. However, the primary known constituent elements of Big Spring spinedace critical habitat include: 1) clean, permanent, flowing, spring-fed stream habitat with deep pool areas and shallow marshy areas along the shore; and 2) the absence of non-native fishes.

Food preferences and feeding habitat are unknown, but closely-related spinedace are opportunistic drift feeders, feeding primarily on aquatic insect larvae, but consuming algae and other plant material when insects are scarce (Rinne 1971, Minckley 1973). It has been suggested that vegetation, especially watercress, is important in providing habitat for aquatic insect and invertebrate foods for the Big Spring spinedace (Allan 1985). Big Spring spinedace spawning behavior has never been observed (as of 1993), and spawning habitat requirements are unknown.

The Big Spring spinedace was federally listed as threatened on March 28, 1985. On the same date, critical habitat was designated for the species along 4 stream miles of the Meadow Valley Wash, and a 50 foot riparian zone on either side of the stream. Introduced, non-native species and the diversion of water and other hydrologic alterations of its habitat threaten the subspecies. Because of its limited distribution, it is vulnerable to events that may severely reduce or extirpate its extant population. Other activities that disturb riparian habitat or alter water quality can also impact the subspecies.

Subtropical Steppe/Subtropical Desert Ecoregions

Subtropical steppes are hot, semi-arid regions that border and grade into more arid subtropical deserts, located in the Great Basin and Southwestern areas of the U.S. Because of the dry climate, in which annual losses of water through evaporation exceed annual gains through precipitation, no permanent streams originate in these ecoregions. However rivers that originate in more northern states, such as the Colorado River and the Rio Grande and their tributaries, run through these regions, and support aquatic life. Other important aquatic habitats include springs and other desert wetlands, which are often fragile and isolated, and provide the only suitable habitat for rare aquatic species in an otherwise dry landscape.

Because of the similarity in aquatic habitats in subtropical steppes and subtropical deserts, and because many of the river systems in which TEP species are found run through both ecoregion divisions, it is difficult to separate species out from one region or the other. Although some species do occur solely in the subtropical desert ecoregion, there are only two species (Little Colorado spinedace and Kanab ambersnail) that occur solely in the subtropical steppe ecoregion. Numerous species are found in both.

Little Colorado Spinedace
The primary reference for this section is:
Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation: Wildland Urban Interface Fuel Treatment. U.S. Department of Agriculture Forest Service, Southwestern Region. Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Little Colorado spinedace (*Lepidomeda vittata*) is a minnow that is endemic to the Little Colorado River Basin, and native to most of the north-flowing tributaries and headwaters of the Little Colorado River (Miller 1963). The known historical distribution is similar to the current distribution, except that the species may have once also occurred in the Zuni River watershed south of Gallup, New Mexico (Sublette et al. 1990).

The most important habitat constituents for this species include clean, permanent flowing water with pools and a fine gravel or silt-mud substrate. The diet of the Little Colorado spinedace varies seasonally, and consists of mostly aquatic and terrestrial insects. Adult aquatic insects are eaten preferentially. This species forages opportunistically, and is able to switch diets based on food availability (Blinn and Runck 1990). Spinedace mature at around 2.3 inches and are prolific spawners; they may spawn more than once a year. Spawning primarily occurs in spring and early summer, but can also occur sporadically throughout the summer and fall months (Minckley and Carufel 1967, Minckley 1984, Blinn and Runck 1990). Spawning products are broadcast over the bottom, on aquatic vegetation, or on debris (Minckley 1973). Growth is rapid, with individuals reaching the size of sexual maturity within 3 months. The life span of spinedace is about 3 years.

The Little Colorado spinedace was federally listed as threatened on September 16, 1987. Critical habitat has been designated in the following areas in Arizona: 31 miles of East Clear Creek (Coconino County) from its confluence with Leonard Canyon upstream to Blue Ridge Reservoir, and from the upper end of Blue Ridge Reservoir to Potato Lake; 8 miles of Chevelon Creek (Navajo County), from the confluence with the Little Colorado River upstream to the confluence of Bell Cow Canyon; and 5 miles of Nutrioso Creek (Apache County), from the Apache-Sitgreaves National Forest’s boundary upstream to Nelson Reservoir Dam. The critical habitat designation includes only the stream course. Threats to this species include stream diversions, impoundments, use of ichthyotoxins, and introduction of non-native species.

Kanab Ambersnail
The primary reference for this section is:
USFWS. 1995d. Kanab Ambersnail (*Oxyloma haydeni kanabensis*) Recovery Plan. Denver, Colorado.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Kanab ambersnail (*Oxyloma haydeni kanabensis*) is a rare terrestrial snail that is endemic to permanently wet areas within small wetlands of the Colorado Plateau. It lives in marshes watered by springs and seeps at the base of sandstone or limestone cliffs (Clarke 1991b, Spamer and Bogan 1993). It is restricted to a permanently wet soil surface or shallow standing water. The snails are also frequently seen just within the mouths of vole burrows. However, none have been found in the drier micro-habitats commonly frequented by other land snails. Cattails, dense sedge and grass, and other vegetation may provide crucial vegetative cover (i.e., protection from predators) and food resources for the snails (Clarke 1991b).

The subspecies is currently (as of 1995) known from three populations, the largest of which is located at Three Lakes, about 6 miles north-northwest of Kanab in Kane County, Utah (Clarke 1991b). Smaller populations are located in Kanab Creek Canyon in Kane County, and at Vasey's Paradise along the Colorado River in the Grand Canyon in Coconino County, Arizona.

Great diversity in the size of individuals within the Utah populations early in the active growing season indicates that reproduction probably occurs throughout all warm, wet periods of the year, and that Kanab ambersnails overwinter as juveniles, sub-adults, and adults (Clarke 1991b). Observations at Vasey's Paradise suggest that reproductive activity is focused in summer months, with die-off of large individuals in late summer and autumn (Blinn et al. 1992). It is probable that the Kanab ambersnail has a life span of about 12 to 15 months (Clarke 1991b).

The Kanab ambersnail was emergency listed on August 8, 1991, and a final rule listing it as an endangered species was published on April 17, 1992. Critical habitat has not yet been determined or designated for this species. Threats to the subspecies stem primarily from loss and/or adverse modification of its wetland habitat. Flooding in particular can affect populations by altering habitat through siltation and scouring. Livestock grazing may also be a threat to the survival of the species.

Mojave Tui Chub
The primary references for this section are:
California Department of Fish and Game. 2000b. The Status of Rare, Threatened, and Endangered Animals and Plants of California, Mohave Tui Chub. California Department of Fish and Game Habitat Conservation and Planning Branch. Sacramento, California.

and

USFWS. 1983. Recovery Plan for the Mohave Tui Chub, *Gila bicolor mohavensis*. Portland, Oregon.

The Mohave tui chub occurred historically in the Mojave River from the confluence of the east and west forks at the base of the San Bernardino Mountains to its terminus at Soda Dry Lake, California. It is the only native fish in this river system. Formerly found in deep pools and slough-like areas of the Mojave River, this species now occurs only in highly modified refuge sites in San Bernardino County, California. The existing Mohave tui chub populations occur at four sites: Soda Springs, the California Department of Fish and Game’s Camp Cady Wildlife Area, China Lake Naval Air Weapons Center, and the Barstow Desert Information Center.

Mohave tui chub typically spawn from March/April to October. Females lay approximately 4,000 to 50,000 eggs over aquatic vegetation. Once hatched, the fry will school in the shallows, while medium-sized tui chub (1 to 3 inches) school in water 1 to 2 inches deep. Large chub are typically solitary and found in deeper water. Mohave tui chub feed on insect larvae and detritus.

The Mohave tui chub was federally listed as endangered on October 13, 1970. Critical habitat has not been designated. Habitat modifications, including damming of the headwaters and withdrawals of the river's underflow, and hybridization with an introduced species, the arroyo chub, contributed to the decline of the species. During the 1930s, arroyo chub were illegally introduced into the headwater reservoirs of the Mojave River as a baitfish. The arroyo chub quickly spread throughout the drainage. Mohave tui chub population numbers began to decrease as a result of competition and hybridization with the arroyo chubs. By 1979, species replacement was complete in their natural habitat. Current threats include genetic contamination, introduction of other exotic species, habitat alteration, water diversion, and pollution.

Virgin River Chub
The primary reference for this section is:
USFWS. 1994c. Virgin River Fishes Recovery Plan. Salt Lake City, Utah.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Virgin River chub (*Gila seminuda*) occurs in the Virgin River Basin, within the Moapa River in Nevada, and within the mainstem Virgin River from Pah Tempe Springs, Utah, downstream to the Mesquite Diversion, located near the Arizona-Nevada border. Virgin River chubs are most often associated with deep runs or pool habitats of slow to moderate velocities with large boulders or instream cover, such as root snags. Adults and juveniles are often associated together within these habitats; however, the larger adults are collected most often in the deeper pool habitats within the river. Chub are generally found in stream waters in velocities ranging up to 2.5 feet per second.

Virgin River chubs are omnivorous, showing considerable dietary shifts with age. In general, Virgin River chubs feed mainly on debris and chironomids in February; *Cladophora* and debris in June; debris and *Spyrogyra* and *Cladophora* in September; and unidentified drift animals, dragonfly larvae, debris, and *Cladophora* in December. Young fish feed almost entirely on macroinvertebrates, while adults feed almost exclusively on algae and debris (Greger and Deacon 1988).

Very little is known about the reproductive biology of the Virgin River chub. The exact time of spawning for this species is not known. However, it is known that Virgin River chubs successfully spawn in both artificial pond habitats and the mainstem Virgin River (Utah Division of Wildlife Resources and Dexter National Fish Hatchery and Technology Center, unpublished data).

The Virgin River chub was federally listed as endangered on August 24, 1989. Critical habitat was designated for the species on January 26, 2000, and includes the 87.5 miles on the mainstem Virgin River and its 100-year floodplain (only those portions that contain at least one of the primary constituent elements for critical habitat), extending from the confluence of La Verkin Creek to Halfway Wash. The critical habitat designation represents approximately 66% of the species' historical habitat within the Virgin River Basin. It also consists of the species' remaining occupied habitat, which flows through both public and private lands. The major limiting factors for the Virgin River chub are modification and loss of habitat, and the introduction and establishment of non-native fish, particularly red shiner. Building of dams and associated reservoirs, water diversion structures, canals, laterals, aqueducts, and the dewatering of streams causes loss or degradation of available habitat.

Woundfin
The primary reference for this section is:
USFWS. 2000g. Designation of Critical Habitat for the Woundfin and Virgin River Chub. Federal Register 65(17): 4140-4156.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Salt Lake City Field Office, Salt Lake City, Utah.

The original range of the woundfin (*Plagopterus argentissimus*) extended from near the junction of the Salt and Verde rivers at Tempe, Arizona, to the mouth of the Gila River at Yuma, Arizona (Gilbert and Scofield 1898, Minckley 1973). Woundfin were also found in the mainstem Colorado River from Yuma (Jordan and Evermann 1896, Meek 1904, Follett 1961) upstream to the Virgin River in Nevada, Arizona, and Utah and into La Verkin Creek, a tributary of the Virgin River in Utah (Gilbert and Scofield 1898, Snyder 1915, Miller and Hubbs 1960, Cross 1975). However, because no barriers or habitat considerations exist that would have precluded woundfin from existing further upstream in these rivers, it is believed that the woundfin likely occurred further upstream in the Verde, Salt, and Gila rivers in Arizona. With the exception of the mainstem of the Virgin River, woundfin are extirpated from most of their historical range. They presently range from Pah Tempe Springs (also called La Verkin Springs) on the mainstem of the Virgin River and the lower portion of La Verkin Creek in Utah, downstream to Lake Mead.

Adult and juvenile woundfin inhabit runs and quiet waters adjacent to riffles with sand and sand/gravel substrates. Adults are generally found in habitats with water depths between 0.5 and 1.4 feet, with velocities between 0.8 and 1.6 feet per second. Juveniles select areas with slower and deeper water, while larvae are found in backwaters and stream margins, which are often associated with growths of filamentous algae. Spawning takes place during the period of declining spring flows.

The woundfin was federally listed as endangered on October 13, 1970. Critical habitat was designated for the species on January 26, 2000. This designation includes the mainstem Virgin River and its 100-year floodplain (only those portions that contain at least one of the primary constituent elements for critical habitat), extending from the confluence of La Verkin Creek, Utah, to Halfway Wash, Nevada, and includes 37.3 miles in Utah, 31.6 miles in Arizona, and 18.6 miles in Nevada. This designation includes a total of 87.5 miles of the mainstem Virgin River, which represents approximately 13% of the woundfin's historical habitat. The area of the Virgin River designated as critical habitat consists of the remaining occupied habitat for the woundfin, which flows through both public and private lands. Threats to this species include dams, reservoirs, and water diversions that modify habitat, as well as non-native species.

Moapa Dace

The Moapa dace (*Moapa coriacea*) is endemic to the warm spring area at the headwaters of the Moapa (Muddy) River, in northern Clark County, southeastern Nevada. The species is restricted to warm springs, their outflows, and the warm waters of the upper mainstream Muddy River. Velocity flow is variable, but in many areas can be swift. Moapa dace are usually found in waters no cooler than 81 ^{o}F, although they have been taken below the low head dam in waters as cool as 67 ^{o}F (Deacon and Bradley 1972). Dissolved oxygen concentrations in Moapa dace habitat have been recorded between 1.6 and 8.9 ppm (Deacon and Wilson 1967), although concentrations below 2.4 ppm seem to be uncommon (Hubbs and Hettler 1964; Hubbs et al. 1967; USFWS 1984d). Streamside vegetation is dense throughout most of the Moapa dace habitat, frequently forming a complete canopy over the stream and filling the channel with snags and brush (Bradley and Deacon 1967, Deacon and Bradley 1972, USFWS 1984d). Streamside vegetation consists of ash, cottonwood, screwbean mesquite, willow, tamarisk, grape vines, and a variety of shrubs, grasses and, herbs.

The Moapa dace appears to be predominantly carnivorous, feeding on invertebrates, and lesser amounts of detritus and filamentous algae. Direct observation of feeding indicates that the species feeds relatively indiscriminately on drift. Fish tend to congregate at dawn and dusk in swift water near snags, and dash up into the current to pick off drift material passing by. Observations of feeding behavior in pool habitats indicate Moapa dace will consume benthic invertebrates directly off the bottom. Larvae, living in shallower, more slowly moving water, probably feed on the much smaller micro-crustacea.

Moapa dace can reproduce throughout the year in the nearly constant temperatures of their habitat. Peak reproduction probably occurs from February to April followed by peak emigration of the young in May (USFWS 1984d). This species has been observed spawning on sandy substrate in a water depth of 6 to 7.5 inches, and a near-bed velocity of 0.1 to 0.3 feet per second. Preliminary measurements of fecundity indicate a range of 97 to 386 eggs produced per adult female.

The Moapa dace was federally listed as endangered on March 11, 1967. Critical habitat has not been designated. A study by the USFWS (1984d) indicates that the Moapa dace occupies only 10% of its original range. The most important factor limiting the distribution and abundance of the Moapa dace within its former range may be turbidity caused by irrigation return flows into the formerly clear water. The feeding ability of the Moapa dace may have been severely curtailed by this increased turbidity. Other apparent reasons for the species' decline include competitive interactions with introduced exotic species (USFWS 1983a), parasites (commonly associated with aquarium fishes and introduced through these exotic fish; La Rivers 1962, Hubbs and Deacon 1964, Bradley and Deacon 1967, Minckley and Deacon 1968, USFWS 1983a), and declining water quality (chemical parameters and physical parameters) from channelization and irrigation for agricultural development (Cross 1976, USFWS 1983a). Future threats to the species include additional water development for irrigation or any activity that would increase the turbidity, reduce the low gene pool, channelize the stream course, or add exotic species to the stream in the headwaters of the Muddy River.

Ash Meadows Naucorid

The primary reference for this section is:
USFWS. 1990d. Recovery Plan for the Endangered and Threatened Species of Ash Meadows, Nevada. Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Ash Meadows naucorid (*Ambrysus amargosus*) is a small, flightless insect that occurs in aquatic habitats in Ash Meadows, Nevada, the sole habitat for 33 unique plants and animals. Ash Meadows is a large oasis in southwestern Nevada, situated at approximately 2,200 feet in elevation in the Mojave Desert, 40 miles east of Death Valley National Monument headquarters at Furnace Creek, California, and 90 miles northwest of Las Vegas, Nevada. The area includes approximately 50,000 acres of desert uplands and spring-fed oases that straddle the California-Nevada border. Its nearly 50 seeps and springs discharge about 17,000 acre-feet of water annually (Walker and Eakin 1963; Bateman et al. 1974). This water formerly flowed into an extensive marsh, which was drained in the mid-1960s. Thunderstorms occasionally caused floodwaters to discharge from Ash Meadows into the Amargosa River, which terminates in the floor of Death Valley.

A creosote bush vegetation community predominates in the surrounding region. On the nearly level terrain near the springs, vegetation is dominated by groves of velvet ash trees and screwbean mesquite in association with seep willow. Sand dunes in the area are dominated by western honey mesquite. Shadscale and alkali goldenbush dominate areas away from the direct influence of the spring waters. Large areas of seasonally wet, salt-encrusted soils are covered with saltgrass. Creosote bush dominates the better-drained soils on the surrounding slopes. Discharge from springs maintains soil moisture in the lowlands, while the uplands receive water only from rainfall that averages less than 3 inches annually.

Within Ash Meadows, the naucorid is known to occupy an extremely restricted habitat where flowing water passes over rock and pebble substrates at Point of Rocks Springs (La Rivers 1953). Although little is known about its life history or habitat requirements, food for closely related naucorids includes aquatic insect larvae that are preyed upon while the naucorid swims over and through the substrate (La Rivers 1951, Polhemus 1979). Reproduction occurs during early spring and summer. Female naucorids deposit eggs that adhere to the substrate during incubation (Usinger 1946).

The Ash meadows naucorid was listed as threatened on May 20, 1985. Approximately 10 acres at Point of Rocks Springs have been designated as critical habitat for this species. The small size and vulnerability of its habitats make the species highly susceptible to extirpation. The species is threatened by altered surface drainage patterns that reduce or eliminate surface water, lower the water table, or interfere with groundwater recharge.

Nevada Speckled Dace

The Nevada speckled dace (also commonly known as the Ash Meadows speckled dace; *Rhinichthys osculus nevadensis*) is endemic to spring systems and aquatic habitats formed by spring waters at Ash Meadows, in Nye County, Nevada. Although formerly more widespread in the area, the species is currently restricted to Jackrabbit Spring, Big Spring, the two westernmost springs of the Bradford Springs group, and the outflows of these springs (NatureServe 2001). This dace is known to occur in headwater spring pools, spring outflow creeks (including areas of the creek up to a mile or more from their spring sources), and marshes formed by spring flows (Soltz and Naiman 1978, Hardy 1980, Williams and Sada 1985). The subspecies also occurs in irrigation ditches and canals that utilize the spring flows for irrigation. The Nevada speckled dace appears to be rather general in its habitat requirements, utilizing areas of rather fast stream current, as well as quiet spring pools.

Speckled dace are typically omnivores (Moyle 1976). They often feed on bottom materials, including aquatic insect larvae, crustaceans, attached diatoms, snails, and algae. Some mid-water foods or even an occasional surface insect will be taken (Moyle 1976, Williams and Williams 1982). Terrestrial insects that fall in the water may also be consumed.

Speckled dace generally become mature in their second summer (Moyle 1976). The spawning season is often during the spring, but some spawning may occur all year, especially in spring habitats with a rather narrow range of temperatures. Speckled dace typically spawn on the gravel edge or riffles in stream habitats. No pair bonds are formed; rather, when a female enters a spawning area during the breeding season, several males may swim out to spawn with her. Eggs hatch in approximately 6 days (at water temperatures of 64 to 66 ^{o}F).

Human development in the area consists primarily of small, scattered residences with which subsistence gardens, small orchards, or agricultural fields may be associated. During the early 1970s, a large farm began operating in Ash Meadows. Development of the farm involved extensive removal of natural vegetation, land leveling, construction of irrigation wells, ditches, and fences, and other activities necessary for commercial farming (Worts 1963, Dudley and Larson 1976). The former major threats from dewatering and development were eliminated with the establishment of the Ash Meadows National Wildlife Refuge. However, some of the spring outflows that were diverted into ditches in the past remain today.

The Nevada speckled dace was federally listed as endangered on September 2, 1983. Critical habitat has been designated in Nye County, Nevada in Section 11 Township 18 South, Range 50 East and Sections 18 and 19 in Township 18 South, Range 51 East. The primary threats to the Nevada speckled dace consist of habitat destruction and the effects of exotic fish introductions. Because of the acquisition of many spring areas by the USFWS, the major threats in the future will most likely consist of additional exotic species introductions rather than physical habitat alteration.

Pahrump Poolfish

The Pahrump poolfish (also commonly referred to as the Pahrump killfish; *Empetrichthys latos*) was extirpated from its original range in the Pahrump Valley of Nevada. This species is now maintained in three refugia where it has been reintroduced: Corn Creek Springs, Spring Mountain Ranch State Park, and Shoshone Ponds. All three areas are bordered by natural desert vegetation and are protected from excessive public disturbance. Corn Creek Springs on the Desert National Wildlife Range is located about 40 miles north of Las Vegas, Nevada. Adult poolfish feed on debris, insect parts, sand, insect larvae, snails, eggs, and plants. Juveniles and young probably depend on zooplankton algae and debris as their primary food sources.

Isolated in desert springs, the Pahrump poolfish is a non-migratory species. Spawning of Pahrump killifish in constant-temperature natural springs probably can occur throughout the year, as is common with desert spring fishes (Deacon and Minckley 1974). Poolfish maintain their populations in desert springs containing no other fish species. They survive and reproduce well in these stable habitats with a diversity of plant and invertebrate species (Soltz and Naiman 1978; Deacon et al. 1980).

The Pahrump poolfish was federally listed as endangered on March 11, 1967. In 1993, the USFWS proposed a reclassification of its status to threatened. Excess pumping of groundwater for agricultural irrigation is seen as the primary cause of habitat destruction for this species (Minckley and Deacon 1968, Soltz and Naiman 1978, Deacon 1979). All native habitats of the Pahrump poolfish have been destroyed, and locations where transplanted populations occur are likely to require management intervention to maintain healthy populations. Current threats to the species include potential reintroductions of predatory fish, and possible mortality under extreme winter weather conditions.

Yaqui Topminnow

The primary reference for this section is:
USFWS. 1994d. Yaqui Fishes Recovery Plan. Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Yaqui topminnow (*Poeciliopsis occidentalis sonoriensis*) occurs in the Rio Yaqui Basin, a drainage that (in the United States) includes parts of Cochise County, Arizona, and Hidalgo County, New Mexico. This subspecies lives in shallow, warm, quiet waters, and occasionally in moderate to relatively swift currents (Galat and Robertson 1988, 1992). Preferred habitats usually include dense mats of algae and debris along stream margins or in eddies below riffles, typically over sandy substrates covered with organic muds and debris. The topminnows become most abundant in marshes, especially those fed by thermal springs or artesian outflows (Sims and Simms 1992). Topminnows eat detritus, living vegetative material, amphipod crustaceans, and aquatic insect larvae, including mosquitoes (Minckley 1973, Gerking and Plantz 1980).

Female Yaqui topminnows may have over 20 young per brood, with broods produced at intervals of approximately 20 days. Reproduction occurs year-round where winter temperatures are ameliorated by the inflow of springs, but under conditions of fluctuating temperature reproduction begins in early April and ends in October (Minckley 1973; Galat and Robertson 1988, 1992). Few individuals in nature live longer than a year.

The Yaqui topminnow was federally listed as endangered on March 11, 1967. Critical habitat has not been designated for this species. The Rio Yaqui watershed has been heavily utilized for cattle grazing and farming, changing the diversity of natural landscapes in the region. Severe grazing pressure has led to the incision of stream channels, desiccation of cienegas, and excessive exploitation of underground aquifers. The introduction of non-indigenous fish species into the system has also contributed to a further general decline in aquatic communities. These factors continue to threaten the Yaqui topminnow.

Beautiful Shiner

The beautiful shiner (*Cyprinella formosa*) occurs in several basins in southeastern Arizona and northwestern Mexico. It is most common in riffles of small streams, and presumably uses pools of intermittent streams for refugia (Minckley 1985). This species is not common in rivers but has been found in rapids with water velocities exceeding 3.3 feet per second, in addition to earthen tanks and ponds on the San Bernardino Ranch. The species has also successfully reproduced in earthen ponds at the Dexter National Fish Hatchery. Historic habitat for this species in the Mimbres River has been described as a lagoon-like system of deep pools with undercut banks (Antisell 1856). The beautiful shiner feeds on small aquatic and terrestrial macro-invertebrates. What little is known about the cover/shelter requirements for this species indicates that pools of intermittent streams are used as refugia. This species may live to 3 years and may spawn from spring through late summer (Pfleiger 1975, Becker 1983).

The beautiful shiner was federally listed as threatened on August 31, 1984. Critical habitat has been designated in aquatic habitats of the San Bernardino National Wildlife Refuge. Primary reasons for the decline of this species include arroyo cutting caused by overgrazing and the removal of riparian vegetation, pumping of groundwater, damming of watercourses, and the introduction of exotic species (USFWS 1984e). More specifically, within the U.S., capping of the artesian well leading to what is now Twin Ponds on the San Bernardino National Wildlife

Refuge in about 1970 destroyed a short spring-fed run and cienega that served as a breeding habitat and refuge for the beautiful shiner. Capping of the well forced the shiner into a pond inhabited by predatory bluegill, black crappie, and largemouth bass, causing the extinction of the minnow within the United States. The species has since been reintroduced to ponds in the San Bernardino National Wildlife Refuge.

Pecos Bluntnose Shiner
The primary references for this species are:

USFWS. 1992. Pecos Bluntnose Shiner Recovery Plan. Region 2. Albuquerque, New Mexico.

and

USFWS. 1987. Endangered and Threatened Wildlife and Plants; *Notropis simus pecosensis* (Pecos Bluntnose Shiner). Federal Register 52(34): 5295-5303

References cited in this section are internal to the about referenced documents. A complete list of these references is available from the USFWS, Albuquerque, New Mexico.

Pecos bluntnose shiner (*Notropis simus pecosensis*) are a moderate-sized shiner separable from co-occurring shiners by its robust body, blunt and rounded snout, and large, slightly subterminal mouth that usually extends even with the pupil. The Pecos bluntnose shiner occupies most major habitats within the Pecos River, but is most common in the main channel. It is typically found in low velocity water, 7 to 16 inches deep, over sand substrate (Hatch et al. 1985). Historically, the Pecos blunt shiner inhabited the mainstream Pecos River from Santa Rosa downstream to the vicinity of Carlsbad, New Mexico (Hatch et al., 1985). However, this species has decreased drastically in abundance and range. It is now restricted to two Pecos River segments: in the Pecos River below Lake Sumner downstream to the upper end of, and seasonally in, Brantley Reservoir, New Mexico, totaling approximately 100 miles of habitat.

Spawning is probably initiated in spring and continues through early autumn (Hatch et al. 1985). Pecos bluntnose shiner food and feeding habitats have not been investigated, but the species probably feeds on small aquatic macroinvertebrates, like that of many other shiners (Starrett 1951, Griswold 1963).

The Pecos bluntnose shiner was listed as threatened on February 20, 1987. On March 27, 1987, the USFWS designated two noncontiguous river reaches, totaling approximately 101 miles of the Pecos River, as critical habitat.

The primary reason for the decline of the Pecos bluntnose shiner is the modification of habitat and establishment of non-native fish species. Hatch et al. (1985) presented information that indicated stream desiccation, due to dam and reservoir operations, river pumping, and the proliferation of the non-native salt cedar (*Tamarix pentandra*), to be the main reason for decline in the Pecos River. Competition from the non-native Arkansas River shiner (*Notropis qirardi*) may also impact the Pecos bluntnose shiner throughout its occupied range (Bestgen et al. 1989). It is likely that other physical habitat modifications, pollution, and nonnative predators/competitors also have contributed to the decline of the species in the Pecos River drainage (Brooks et al. 1991).

Loach Minnow
The primary reference for this section is:
Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation: Wildland Urban Interface Fuel Treatment. Forest Service, Southwestern Region. Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The loach minnow (*Tiaroga* [= *Rhynichthys*] *cobitis*) is a small, slender fish with a life span of about 2 years. This species is endemic to the Gila River drainage of southwestern New Mexico, southeastern and east-central Arizona, and northeastern Sonora, Mexico (Miller and Winn 1951, Koster 1957, Minckley 1973). The loach minnow once occupied as much as 1,243 miles, but its range is now less that 124 miles (Propst et al. 1988), with present populations geographically isolated in the upstream ends of the species' historic range.

In Arizona, the species persists in Aravapai Creek and its tributaries, Turkey and Deer creeks, in the upper reaches of White River, and in limited reaches of the Black River, Blue River and tributaries, Campbell Blue and Little Blue creeks, Eagle Creek, and in the San Francisco River between Clifton and the New Mexico border (Propst et al. 1985). In New Mexico, the loach minnow still occurs in the upper Gila River, including the East, Middle, and West forks, and in the Cliff-Gila Valley reach of the Gila River, the San Francisco and Tularosa rivers and their tributaries, and Pace and Frieborn creeks. Loach minnows are generally absent downstream of the Cliff-Gila Valley (Propst et al. 1988; Propst and Bestgen 1991).

Loach minnows feed exclusively on aquatic insects (Abarca 1987). The are opportunistic, benthic insectivores, largely deriving their food supplies from among riffle-dwelling larval mayflies, blackflies, and midges. Loach minnows appear to actively seek their food among bottom substrates rather than pursuing animals entrained in the stream drift. Spawning typically occurs in the spring when water temperatures exceed 60 ^{o}F. The first spawn occurs in the minnow's second year of life, primarily during March through May (Britt 1982, Propst et al. 1988). However, under certain circumstances, loach minnows also spawn in the autumn (Vives and Minckley 1990). Eggs typically hatch in 5 to 6 days.

The loach minnow was federally listed as threatened on October 28, 1986. Designated critical habitat includes portions of 36 streams, which form seven complexes in the Verde, Black, Tonto, Gila, San Pedro, Blue, San Francisco, and Tularosa basins. Critical habitat includes the stream channel, an identified stream reach, and the 100-year floodplain (USFWS 2000h). Activities that can affect loach minnow habitat include removal of riparian cover, sedimentation, and control of water levels. Dams and reservoirs appear to eliminate loach minnows for many miles upstream and downstream. The spread of non-native predators, especially flathead catfish and channel catfish, can also directly reduce populations of the species.

Gila Trout

The primary reference for this section is:
Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation: Wildland Urban Interface Fuel Treatment. U.S. Department of Agriculture Forest Service, Southwestern Region, Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Gila trout (*Onchorhyncus gilae gilae*) is a moderate-sized salmonid that is native to the headwaters of the Gila River, New Mexico. Available information suggests that the Gila trout once ranged from the headwaters down to the Gila River's confluence with Mogollon Creek. Unique characteristics of Gila trout in Spruce Creek (New Mexico), and the possible historic occurrence of Gila trout in Eagle Creek (Arizona), suggest this species was also native to the San Francisco drainage (Minckley 1973). By 1950, the range of the species had been severely fragmented into small populations isolated in small headwater streams (Main Diamond, South Diamond, McKenna, Spruce, and Iron creeks; USFWS 1993i). Since 1975, Gila trout from each of the five relict populations have been translocated to other streams. These translocations have been largely successful.

Like many salmonids, Gila trout are opportunistic carnivores, consuming a large variety of aquatic and terrestrial insects entrained in the stream drift. Large Gila trout occasionally eat other fish (Van Eimeren 1988). Spawning occurs in the spring, when water temperatures reach about 46 ^{o}F and stream flows recede. Spawning begins in early April at the lowest elevations and continues through June at the highest elevations (Rinne 1980). Fish utilize substrates of fine gravel and coarse sand (0.07 to 1.5 inches) for spawning. Fry emerge from the spawning beds at about 8 to 10 weeks.

The Gila trout was federally listed as endangered on March 11, 1967. Critical habitat has not been designated. The continued decline in this species and its available habitat is attributable to a number of factors, including the introduction of non-native salmonids and land management practices (overgrazing, fires, lumbering, and mining) that have caused habitat loss and modification.

Gila Topminnow

The Gila topminnow (*Poeciliopsis occidentalis occidentalis*) is a small, live-bearing minnow that occurs in isolated springs in the Santa Cruz River system in New Mexico and Arizona, and on the San Carlos Apache Indian Reservation located in southeastern Arizona. Locations currently supporting Gila topminnows in New Mexico and Arizona include Redrock Canyon, Cottonwood Spring, Monkey Spring, upper Sonoita Creek, Fresno Canyon, Coal Mine Canyon, lower Sonoita Creek, Santa Cruz River north of Nogales, Cienega Creek, Sharp Spring, the upper Santa Cruz River, Bylas Spring, Middle Spring, and Salt Creek. Topminnows have fairly broad habitat requirements. They prefer shallow, warm, fairly quiet waters, but can adjust to a rather wide range of conditions, living in quiet to moderate currents. Topminnows live in a wide variety of water types: springs, cienegas, marshes, permanent streams, intermittent streams, and, formerly, along the edges of large rivers. Preferred habitat contains dense mats of algae and debris, usually along stream margins or below riffles, with sandy substrates, sometimes covered with organic muds and debris (Minckley 1973).

The diet of the Gila topminnow is fairly generalized, consisting mostly of bottom debris, vegetable material, and amphipod crustaceans. The topminnows feed voraciously upon aquatic insect larvae, such as mosquitoes, when available. The breeding season for this species lasts from January to August, but a few pregnant females are present throughout the year, and young are produced even in winter. Sexual maturity may occur in a few weeks to many months after birth, depending largely upon the time of year the individual is born. Topminnows are not thought to live longer than a year under natural conditions (Minckley 1973).

The Gila topminnow was federally listed as endangered on March 11, 1967. Critical habitat has not been designated. The decline of this species is attributable to several factors: the construction of dams; the introduction of non-native predatory and competitive fish; drainage of wetlands and cienegas; and the desiccation of streams, springs, and cienegas (Miller 1961). Today, because of the presence of barriers to movement, Gila topminnows can no longer re-distribute from their remaining isolated and widely separated populations to colonize formerly occupied habitats, even during years with above average rainfall.

Gila Chub

The primary reference for this section is:
USFWS. 2002f. Listing the Gila Chub as Endangered With Critical Habitat. Federal Register 67(29): 6459-6479.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Arizona Ecological Services Field Office, Phoenix, Arizona.

The Gila chub (*Gila intermedia*) is a fish in the minnow family that inhabits pools in smaller streams, springs, and cienegas, and can survive in small artificial impoundments (Miller 1946, Minckley 1973, Rinne 1975). Historically found throughout the Gila River basin in southern Arizona, southwestern New Mexico, and northeastern Sonora, Mexico, Gila chub have been extirpated or reduced in numbers and distribution throughout much of this range (Minckley 1973, Weedman et al. 1996). Numerous events, occurring nearly a century ago, led to long-term stream, cienega, and riparian habitat degradation throughout southern Arizona and northern Mexico, and the ecosystem has not fully recovered, and in some areas may never recover. Approximately 85 to 90% of the Gila chub's habitat has been degraded or destroyed, and much of it is unrecoverable.

Gila chub are highly secretive, preferring quiet deeper waters, especially pools, or remaining near cover including terrestrial vegetation, boulders, and fallen logs (Rinne and Minckley 1991). Undercut banks created by overhanging terrestrial vegetation with dense roots growing into pool edges provide ideal cover (Nelson 1993). Gila chub can survive in larger stream habitat such as the San Carlos River, and artificial habitats, like the Buckeye Canal (Stout et al. 1970, Rinne 1976). Gila chub interact with spring and small stream fishes regularly (Meffe

1985), but prefer deeper waters (Minckley 1973). Adults often are found in deep pools and eddies below areas with swift current, as in the Gila chub habitats found in Bass Canyon and Hot Springs in the Muleshoe Preserve area. Young-of-the-year inhabit shallow water among plants or eddies, while older juveniles use higher-velocity stream areas (Minckley 1973, 1991).

Gila chub are omnivorous (Griffith and Tiersch 1989), although adults appear to be principally carnivorous, feeding on large and small aquatic and terrestrial invertebrates and sometimes other small fishes (Rinne and Minckley 1991). Smaller individuals often feed on organic debris and aquatic plants, especially filamentous (threadlike) algae, and less intensely on diatoms (unicellular or colonial algae). Bottom feeding may also occur.

Spawning probably occurs over beds of submerged aquatic vegetation or root wads. Warmer water temperatures (68 to 75.2 °F) appear to contribute to a successful spawn (Nelson 1993). For the roundtail chub, a close relative of the Gila chub, spawning occurs when water temperatures are approximately 68 °F, and temperature appears to be the most important environmental factor triggering spawning (Bestgen et al. 1985).

The USFWS proposed listing the Gila chub as endangered on August 9, 2002. In addition, a total of 207.8 miles of stream reaches within seven river units (including 122.3 miles in federal land) were proposed for designation as critical habitat. These river units represent those areas that currently are within the geographical range occupied by the Gila chub, including small tributaries, springs, and cienegas. Where the species is still present, populations are often small, scattered, and at risk from known and potential threats and from random events. Threats include predation by and competition with non-native organisms, including fish in the sunfish and bass family, other fish species, bullfrogs, and crayfish; disease; and habitat alteration, destruction, and fragmentation resulting from water diversions, dredging, recreation, roads, livestock grazing, changes in the natural flow pattern, mining, degraded water quality (including contaminants from mining activities and excessive sedimentation), and groundwater pumping.

Spikedace

The primary reference for this section is:
Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation: Wildland Urban Interface Fuel Treatment. Forest Service, Southwestern Region, Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The spikedace (*Meda fulgida*) is a stream-dwelling member of the minnow family that was once common in large and moderate-sized rivers throughout the upper Gila River Basin of Arizona and New Mexico. The species is now restricted to less than 6% of its historic range. In Arizona, it occurs in Aravapai Creek, Eagle Creek, and the upper Verde River between Sullivan Lake and Tapco. In New Mexico, it occurs in the mainstem Gila River above redrock, and in the East, Middle, and West forks of the Gila River (Barber and Minckley 1966; Minckley 1973; Anderson 1978; Barrett et al. 1985; Marsh et al. 1989; Sublette et al. 1990).

Spikedace feed primarily on aquatic and terrestrial insects (Barber and Minckley 1983, Marsh et al. 1989), although the species will feed on fry of other fish during certain seasons. The aquatic insects consumed by the spikedace occur mainly in riffle habitats, which provide the clean and relatively stable conditions they require. The spikedace’s diet is highly dependent on the type of habitat and time of year (Minckley 1973). Spikedace feed by picking off food items entrained in stream drift. Spawning occurs from March through May, but there is some yearly and geographic variation (Barber et al. 1970; Anderson 1978; Propst et al. 1986). Breeding is initiated in response to a combination of declining stream discharge and increasing water temperatures. Young grow rapidly, attaining adult size by November of the year spawned. Spikedace live approximately 2 years, with reproduction typically occurring in 1-year-old fish.

The spikedace was federally listed as threatened on July 1, 1986. Critical habitat has been designated in portions of 24 streams, forming complexes: Verde, Tonto Creek, Gila, San Pedro, San Francisco, and Blue River basins.

Critical habitat includes the stream channel, the stream reach, and the 100-year floodplain. The primary causes of this species' continued decline are competition with and predation by introduced, non-native fish species (Miller 1961, Williams et al. 1985). Habitat destruction is also a threat.

Socorro Isopod

The Socorro isopod (*Thermosphaeroma thermophilus*) is endemic to central New Mexico (Pennak 1978). It is known to occur in only one location: two small pools fed by Sedillo Spring in Socorro County, New Mexico (NatureServe Explorer 2001). This species lives in thermal habitats, requiring warm springs that are less than a foot deep. Water temperatures throughout the system occupied by the Socorro isopod range from 77 to 91 °F (New Mexico Department of Game and Fish 1995), and the water surface is covered by algae (USFWS 1982c). The floor of the smaller pool is composed of 0.8 to 2.4 inches of sediment into which the isopods burrow.

The Socorro isopod is reported to feed on algae and detritus, and is also cannibalistic (Schuster 1977). Cannibalism involves feeding on wounded or otherwise not entirely intact isopods and/or attacking a healthy isopod (by several isopods). The species appears to be primarily nocturnal, avoiding direct sunlight. Activity increases toward late afternoon, reaching a peak about an hour before sunset, and remaining high until just before dawn.

Complete development of isopod embryos takes 30 to 40 days (New Mexico Department of Game and Fish 2002). Juveniles may molt up to eight times. In laboratory conditions, brood sizes ranged from 3 to 57 individuals, and gestation was about 30 days (USFWS 1982c).

The Socorro isopod was federally listed as endangered on March 27, 1978. Critical habitat has not been designated. In August 1988, the entire population died out at the spring, when the flow of water became occluded and the habitat dried out. However, a population of the isopod housed at the University of New Mexico saved the species from extinction, and a transplant has restored it to Sedillo Spring (New Mexico Department of Game and Fish 1988). Threats to the Socorro isopod include vandalism of its extremely limited habitat, any activity that alters the thermal spring or reduces its flow, and any activity that alters either the physical or chemical quality of the spring water.

Alamosa Springsnail and Socorro Springsnail

The primary reference for this section is:
USFWS. 1993j. Alamosa Springsnail (*Tyronia alamosae*) and Socorro Springsnail (*Pyrgulopsis neomexicana*) Draft Recovery Plan. Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Alamosa springsnail (or Caliente tryonia; *Tryonia alamosae*), and the Socorro springsnail (*Pyrgulopsis neomexicana*) are two species of aquatic snail that are endemic to thermal habitats in central New Mexico. Both species require a continued supply of free-flowing thermal spring water that is free of pollutants, bordered by a zone of organic detritus and vegetation that is sufficient to support their biological and habitat requirements.

The Alamosa springsnail is endemic to central New Mexico. The species is known only from a thermal spring complex in Socorro County, New Mexico, which consists of five individual springheads that flow together. The species also occurs in minor rivulets out of the main channel in the canyon where the springs arise (Taylor 1987b).

The Alamosa springsnail is found primarily where minor rivulets flow out of the main channel downstream of the springhead (Taylor 1987b). In these situations, there is a mat of watercress and filamentous green algae, over water 1 to 2 inches deep and flowing over fine gravel and sand among cobbles and rocks. The species is found in areas of slow-moving current, on gravel and among vegetation, and is most abundant where an organic film covers the pebbles and cobbles. As spring runs join and form a narrow, swifter, flowing brook, snails become less numerous. Water temperature at the springheads remains between 81 and 82.5 °F. It appears that seasonal fluctuations in water flow and temperature do not occur.

The Alamosa springsnail is herbivorous, feeding on algae and other materials that occur in the organic film on plants and debris. The species contains a series of embryos in various stages of development, and eggs hatch within the female parent or immediately after being forced from the parent. Because the Alamosa springsnail lives in a thermally constant environment, reproduction is probably not seasonal, and population size likely remains relatively stable (New Mexico Department of Game and Fish 1985).

The Socorro springsnail is known from only one spring on privately-owned property in Socorro County, New Mexico, where it was first found in 1979. The source of this spring has been impounded, reducing the flowing-water habitat to a very small pool. One tiny spring source having a small, improved pool, with a water temperature of 63 °F remains. The species is abundant on rootlets in this pool, but is not found in the ditches and ponds radiating from the spring into irrigation structures. Like the Alamosa springsnail, the Socorro springsnail occurs in slow-velocity water near spring sources, on stones and among aquatic plants. The species is also herbivorous, and feeds on algae and other materials in organic film. The Socorro springsnail produces eggs that develop and hatch after being laid, probably in the spring and summer.

The Alamosa and Socorro springsnails were listed as endangered on September 30, 1991. Critical habitat has not been designated for either species. The limited ranges of both springsnails make them vulnerable to habitat loss or alteration. Potential threats to the species include all activities that would substantially reduce spring flow or the food source that supports the springsnails. Alterations of the watersheds, springs, or associated runs could cause a reduction in water flow, a change in water temperature or water quality, or a modification in habitat or food sources, thus having a devastating impact on existing populations. It is believed that the greatest threat to these species is the potential loss of water flow. Excessive pumping from the aquifer that supplies water to the springs could destroy the springs and the species. Pollution of the springs could also negatively impact these species. Other threats include introduction of non-native fishes and other aquatic organisms, and collection.

Pecos Gambusia

The Pecos gambusia (*Gambusia nobilis*) is a fish that occurs abundantly in springs within a small range in the Pecos River Basin, New Mexico and Texas. However, the species also occurs in areas with little spring influence, but with abundant overhead cover, sedge-covered marshes, and gypsum sinkholes. The Pecos gambusia is found from the surface to depths of about 10 feet (USFWS 1983b). All populations, including those at historic, present, and introduction sites, occur in habitats between 2,700 feet and 3,900 feet in elevation. The species prefers water temperatures of 70 to 77 °F in the morning and 79 to 86 °F in the afternoon. In contrast, a potential competitor, western mosquitofish, is more tolerant of higher water temperatures. The Pecos gambusia is essentially restricted to stenothermal, clear water, lotic habitats. It lives in a variety of habitats in Bitter Lake National Wildlife Refuge (Bouma 1984). Lands surrounding the habitats are classified as Texas savanna and shrub/brush rangelands or mixed rangelands. Like other gambusia species, the Pecos gambusia is considered to be a carnivorous surface feeder (Bednarz 1979). It appears to be an opportunistic feeder, eating a variety of small invertebrates and filamentous algae. This species bears live young, primarily in shallow areas.

The Pecos gambusia was federally listed as endangered on October 13, 1970. Critical habitat has not been designated. The species faces two major threats—loss of habitat and the inability to interact successfully with non-native fish species, especially other gambusia species (USFWS 1983b).

Pecos Assiminea Snail, Roswell Springsnail, Koster's Tryonia, and Noel's Amphipod

The primary reference for this section is:
USFWS. 2002g. Listing Roswell Springsnail, Koster's Tryonia, Pecos Assiminea, and Noel's Amphipod as Endangered With Critical Habitat. Federal Register 67(29): 6459-6479.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS New Mexico Ecological Services Field Office, Albuquerque, New Mexico.

The Roswell springsnail (*Pyrgulopsis roswellensis*), Koster's tryonia (*Tryonia kosteri*), and Pecos assiminea snail (*Assiminea pecos*) occur at sinkholes, springs, and associated spring runs and wetland habitats in New Mexico and

Texas. These species are associated with the Roswell Basin, which has a surface area of around 12,000 square miles and is located in southeastern New Mexico. The Roswell Basin contains two major aquifers, which are the sources of springs inhabited by these rare snail species. The action of water on soluble rocks, such as limestone and dolomite, has formed abundant physical features, including sinkholes, caverns, springs, and underground streams (White et al. 1995).

The Roswell springsnail occurs at several locations on the Bitter Lake National Wildlife Refuge, and potentially at a site on private land east of Roswell, New Mexico. This species was formerly known from several other springs in the Roswell area, but these habitats have dried up, apparently as a result of groundwater pumping (Cole 1981; Taylor 1983, 1987). Koster’s tryonia also occurs on the Bitter Lake National Wildlife Refuge, and potentially at North Spring on privately-owned land east of Roswell. Like the Roswell springsnail, the species was formerly found at several other springs in the Roswell area that have since dried up as a result of groundwater pumping. The Pecos assiminea snail is currently known from two sites at Bitter Lake National Wildlife Refuge in Chaves County, New Mexico; from a large population at Diamond Y Spring and its associated drainage in Pecos County, Texas; and at East Sandia Spring in Reeves County, Texas.

The Roswell springsnail, Koster’s tryonia, and Pecos assiminae are all aquatic species. As with other snails in the family, the Roswell springsnail and Koster’s tryonia are completely aquatic but can survive in seepage areas, as long as flows are perennial and within the species’ physiological tolerance limit. These two snails occupy springs with variable water temperatures (50 to 68 °F) and slow to moderate water velocities, over compact substrate ranging from deep organic silts to gypsum sands and gravel and compact substrates (New Mexico Department of Game and Fish 1998). In contrast, the Pecos assiminea snail seldom occurs immersed in water, but prefers a humid microhabitat created by wet mud or the undersides of vegetation mats, typically within an inch or so of running water.

Like most snails, the Roswell springsnail. Koster’s tryonia, and Pecos assiminea snail feed on algae, bacteria, and decaying organic material (New Mexico Department of Game and Fish 1988). They will also accidentally ingest small invertebrates while grazing on algae and detritus.

These three snail species have lifespans of 9 to 15 months, and reproduce several times during the spring through fall breeding season (Taylor 1987, Pennak 1989, Brown 1991). All three species belong to a family of snails that are sexually dimorphic, with females characteristically larger and longer-lived than males.

Noel’s amphipod (*Gammarus desperatus*) is a small freshwater crustacean that commonly inhabits shallow, cool, well-oxygenated waters of streams, ponds, ditches, sloughs, and springs (Holsinger 1976, Pennak 1989). Noel’s amphipod is one of three species of endemic amphipods of the Pecos River Basin occurring from Roswell, New Mexico, south to Fort Stockton, Texas. Noel’s amphipod is currently known from only three sites at Bitter Lake National Wildlife Refuge. These sites include the Sago Springs Complex, Bitter Creek, and along a drainage canal near impoundment 6 on the Refuge.

Because they are light-sensitive, bottom-dwelling amphipods are active mostly at night and feed on algae, submergent vegetation, and decaying organic matter (Holsinger 1976, Pennak 1989). Young amphipods depend on microbial foods, such as algae and bacteria, associated with aquatic plants (Covich and Thorp 1991).

Most amphipods complete their life cycle in 1 year and breed from February to October, depending on water temperature (Pennak 1978). Amphipods form breeding pairs that remain attached for 1 to 7 days at or near the substrate while continuing to feed and swim (Bousfield 1989). They can produce from 15 to 50 offspring, forming a brood.

The Roswell springsnail, Koster’s tryonia, Pecos assiminea snail, and Noel’s amphipod were proposed for federal listing as endangered species on February 12, 2002. A total of approximately 1,524 acres of aquatic and adjacent habitat have been proposed as critical habitat for the four species. Areas included in this proposed designation include portions of the Bitter Lake National Wildlife Refuge, the Diamond Y Springs Complex in Pecos County, Texas, and East Sandis Spring in Reeves County, Texas. Primary constituent habitat elements for these species include permanent, flowing, unpolluted fresh to moderately saline water; slow to moderate velocities of water over substrates ranging from deep organic soils to limestone cobble and gypsum substrates; presence of algae, submergent vegetation, and detritus in the substrata; and water temperatures ranging from 50 to 68 °F, with natural diurnal and seasonal variation slightly above and below that range. All four species have an exceedingly limited distribution and are imperiled by local and regional groundwater depletion, surface and groundwater contamination, oil and gas extraction activities within the supporting aquifer and watershed, and direct loss of their habitat (e.g., through burning or removing marsh vegetation, cementing, or filling of habitat).

Pupfish

Pupfish (*Cyprindon* spp.) are small aggressive fish that inhabit freshwater habitats in the southwestern United States. There are six species of pupfish in the project area: Devil's hole pupfish (*C. diabolis*), desert pupfish (*C. macularis*), Ash Meadows Amargosa pupfish (*C. nevadensis mionectes*); Warm Springs pupfish (*C. nevadensis pectoralis*), and Owens pupfish (*C. radiosus*). Information about their locations and habitat are summarized in Table 5-1 below.

Pupfish are primarily substrate feeders, taking in and chewing food such as algae, and then expelling the remainder (Pister, No Date). Other types of food that pupfish may eat include aquatic insects, crustaceans, snails, and eggs. Pupfish spawn over a period of 7 or 8 months, reaching maturity 2 to 4 months after hatching. The rate at which juveniles reach maturity is dependent on habitat; in warm springs they mature in 2 to 4 months, and in habitats where fish remain dormant over the winter (e.g., marshes and ponds), fish can take up to 6 months to mature.

TABLE 5-1
Listed Pupfish in the Project Area

Species	Location	Habitat
Devil's hole pupfish	Devil's Hole, Ash Meadows National Wildlife Refuge, Nye County, Nevada (Death Valley National Park).	Deep, limestone pool and algae-covered ledge; 91 to 93 °F and 1.8 to 3.3 ppm dissolved oxygen.
Desert pupfish	Arizona: extirpated, but several reintroduced populations exist California: San Felipe Creek, San Sebastian Marsh, Salt Creek, and Salton Sea area.	Desert springs and outflow marshes, river-edge marshes, backwaters, saline pools, and streams.
Ash Meadows Amargosa pupfish	Ash Meadows National Wildlife Refuge, Amargosa Desert, Nye County, Nevada.	Pools and outflows of warm springs.
Warm Springs pupfish	Warm Springs, Ash Meadows, Nye County, Nevada.	Thermal springs and their outflows; 86 to 88 °F.
Owens pupfish	Seven locations in the Owens Valley, eastern California.	Well-vegetated shallow sloughs, or spring pools, near margins of bulrush marshes, good quality water, and silt- or sand-covered bottom.
Sources: Biological Resources Research Center (2001), NatureServe Explorer (2001).		

The Devil's Hole and Owens pupfish were listed as endangered on March 11, 1967. The Warm Springs pupfish was listed as endangered on October 13, 1970. The Ash Meadows pupfish was listed as endangered on September 2, 1983. The desert pupfish was listed as endangered on March 31, 1986. Critical habitat has been designated for the Ash Meadows Amargosa pupfish (the 10 spring areas within Ash Meadows in which this species occurs) and the desert pupfish. As species that occupy springs and pools, pupfish are very vulnerable to changes in habitat caused by water diversion, damming, and excessive groundwater pumping. Another major threat is introduced species that compete with or prey upon pupfish, such as largemouth bass, bullfrogs, and crayfish. Other potential threats to one or more species of pupfish include vandalism, stochastic fluctuations, and hybridization.

Temperate Steppe Ecoregion

The Temperate Steppe Ecoregion supports a semiarid climate with cold, dry winters and warm, hot summers. This region includes the Rocky Mountains, in which the headwaters of a number of major river systems (e.g., the Colorado, Green, Missouri, Snake, Platte, and Rio Grande) are located. Most of the TEP aquatic species that occur in this ecoregion are large river-dwelling fish species.

Kootenai River White Sturgeon
The primary reference for this section is:
Hudson, B., J. Augsburger, M. Hillis, and P. Boehne. 2000. Draft Biological Assessment for the Interior Columbia River Basin Ecosystem Management Project Final Environmental Impact Statement. BLM and Forest Service. Boise, Idaho

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The distribution of the Kootenai River population of white sturgeon (*Acipenser transmontanus*) extends from Kootenai Falls, Montana, located 31 river-miles below Libby Dam, downstream through Kootenai Lake to Corra Linn Dam on the lower West Arm of Kootenay Lake, British Columbia. The sturgeon population spawns within a 12 river-mile stretch of the Kootenai River from Bonners Ferry, Idaho, downstream to the lower end of Shorty's Island. Historically, spring runoff peaked during the first half of June in the Kootenai River upstream of the existing Libby Dam in Montana. Runoff from the lower elevations between Libby Dam and Bonner's Ferry, Idaho, was somewhat earlier, peaking in late May. Combined flows were often in excess of 60,000 cubic feet per second. During the remainder of the year, river flows declined to basal conditions of approximately 4,000 to 8,000 cubic feet per second. Annual flushing events re-sorted river sediments, providing a clean cobble substrate conducive to insect production and sturgeon egg incubation. Side channels and low-lying deltaic marsh lands were undiked at this time, providing productive, low velocity backwater areas. Nutrient delivery was unimpeded by dams, and occurred primarily during spring runoff. Floodplain ecosystems, such as the pre-development Kootenai River, are characterized by seasonal floods that promote exchange of nutrients and organisms in a mosaic of habitats, thus enhancing biological productivity.

White sturgeon are considered opportunistic feeders, and have been observed feeding on a variety of prey items, including clams, snails, aquatic insects, and fish. They are generally long-lived, with females living from 34 to 70 years (Pacific States Marine Fisheries Commission 1992). Only a portion of the adult white sturgeon are reproductive or spawn each year, and spawning frequency for females has been estimated at 2 to 11 years. Spawning occurs when the physical environment permits egg development and cues ovulation. White sturgeon are broadcast spawners, releasing their eggs and sperm in fast water. Kootenai River white sturgeon spawn during peak flows, from May through July (Apperson and Anders 1991), when high water velocities disperse and prevent clumping of the adhesive eggs. Following fertilization, eggs adhere to the river substrate and hatch after a relatively brief incubation period of 8 to 15 days, depending on the water temperature (Brannon et al. 1984). Recently hatched yolksac larvae swim or drift in the current for a period of several hours and then settle back into small spaces in the substrate.

The USFWS listed the Kootenai River population of white sturgeon as endangered on September 6, 1994. This population has been in general decline since the mid-1960s, when the Libby Dam began operation. Human activities have modified the natural flows of the Kootenai River, thereby altering the spawning, egg incubation, and rearing habitats of white sturgeon, and reducing overall biological productivity. These factors have contributed to the general lack of recruitment in the white sturgeon population since the mid-1960s. The change to the natural flows in the Kootenai River caused by flow regulation at Libby Dam is considered to be a primary reason for the continuing lack of recruitment and declining numbers in white sturgeon populations.

Pallid Sturgeon
The primary reference for this section is:
Duffy, W.G., C.R. Berry, and K.D. Keenlyne. 1996. Biology of the Pallid Sturgeon with an Annotated Bibliography through 1994. South Dakota Cooperative Research Unit Technical Bulletin Number 5. South Dakota State University. Brookings, South Dakota.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The pallid sturgeon (*Scaphirhyncus albus*) is found at the sandy or rocky bottoms of swift, large, turbid and free-flowing rivers in the Missouri and Mississippi river drainages of central North America. It is one of the largest freshwater fish in North America. Reduced from its historic range, the present distribution of this species includes the Missouri River to Fort Benton, Montana; the lower Mississippi River from New Orleans to its juncture with the Missouri River; the Atchafalaya River to its connection with the Mississippi River; and the lower Yellowstone River from the mouth of the Tongue River to its juncture with the Missouri River. At present, the complete range of the pallid sturgeon is approximately 3,500 miles.

Pallid sturgeon prefer turbid, flowing riverine habitat with rocky or sandy substrate and water depths of 13 to 16 feet (Erickson 1992). They inhabit floodplains, backwaters, chutes, sloughs, islands, sandbars, and main channel waters. In the Missouri River, sturgeon have been captured in the main channels along sandbars at the inside of river bends and behind wing dikes with deeply scoured trenches (Carlson et al. 1985). Fish collected in the Missouri River have been located primarily upstream of reservoirs, and show a preference for riverine-like conditions, if they exist (Kallemeyn 1983). The pallid sturgeon is primarily a fish eater, with large river minnows serving as the primary forage species (Coker 1930; Carlson et al 1985).

Males reach sexual maturity between 5 and 7 years of age, and females become sexually mature at 15 years of age (Keenlyne and Jenkins 1993). Pallid sturgeon may spawn as early as April in the lower portion of their range, or as late as early June in the extreme northern portion of their range. Reproduction coincides with natural high river flows. Under wild conditions, males do not spawn every year, and females may take up to 10 years between spawnings, depending on the quality and quantity of food available in their natural habitat. Therefore, fecundity of a female may vary considerably, with an individual female spawning only a few times during the normal life span.

The pallid sturgeon was federally listed as endangered on June 9, 1990. Critical habitat has not been designated. Over the years, the habitat of the pallid sturgeon has been dramatically altered. The most apparent change is the series of impoundments on the main stem of the upper Missouri River and channelization of the lower Missouri and Mississippi rivers. The upper Missouri River dams have created physical blockages that prohibit normal migration patterns, alter habitat characteristics, and restrict riverine fish to limited flowing river reaches (Hesse et al. 1989). Approximately 51% of the range of the pallid sturgeon has been channelized, 28% has been impounded, and the remaining 21% is affected by upstream impoundments that alter flow regimes and modify both turbidity and water temperatures (Kennlyne 1989). These forms of habitat alteration have changed river parameters such as current velocity, seasonal flows, turbidity, temperature, and nutrient supply paths within the food chain (Hesse 1987). These modifications adversely affect the pallid sturgeon by blocking movements to spawning and/or feeding areas, destroying spawning areas, altering conditions or flows of potential remaining spawning areas, and reducing food sources or the ability to obtain food (Keenlyne 1989). Pollution is becoming more of a threat to this bottom-feeding species throughout its range. In addition, hybridization with the shovelnose sturgeon seems to be increasing, probably as a result of environmental changes and reductions in habitat diversity.

Greenback Cutthroat Trout
The greenback cutthroat trout (*Oncorhyncus clarki stomias*) historically occurred in the sources of the South Platte River and Arkansas River in Colorado, from the headwaters to the foothills, and in a few headwater tributaries of the South Platte River in a small area of southeastern Wyoming (Behnke 1992). The present distribution of the subspecies includes areas in the South Platte Drainage: east slope drainages of Rocky Mountain National Park (Cow and Hidden Valley creeks; Pear Reservoir; West and Fern creeks; Fern, Bear, Caddis, and Odessa lakes; Big

Thompson River); Como Creek, South Boxelder Creek, South Fork of the Cache La Poudre River, and Black Hollow Creek. Greenback cutthroat trout are also found in the Arkansas River drainage: South Huerfano and Cascade creeks in San Isabel National Forest, Hourglass Creek, and Lake Fork above Turquoise Lake.

All of the present habitat where the subspecies occurs is essentially undisturbed headwaters of drainages from 7,000 to 11,000 feet elevation in the Rocky Mountain National Park, on Forest Service-administered lands (Roosevelt, San Isabel, and Pike National Forest), and in one spring-fed pond on Fort Carson. With the exception of the Fort Carson Pond, all habitats are associated with montane conifer forests and meadows. Some streams contain beaver dams and beaver ponds. There is nothing unique about greenback habitat, and the subspecies is able to live in any habitat and tolerate any water quality that supports other species of trout. However, greenback cutthroat trout cannot coexist with other species because of competition and/or hybridization (Behnke and Zarn 1976; Behnke 1976, 1979). Thus, any trout habitat can be greenback trout habitat if no other species of trout are present.

There is no evidence to suggest that greenback trout have feeding preferences that distinguish them from other trout species. Therefore, it can be assumed that a greenback trout of similar size and existing in similar habitat as other trout species will feed on similar food items—predominantly aquatic insects in streams, and zooplankton and benthic crustaceans and insects in lentic environments.

Cover and shelter requirements are similar to those of other trout species. Young and juvenile fish select shallower, more open habitats, and larger, older fish select deeper areas with more cover (boulders, log jams, particularly undercut streambanks). Present habitat of most greenback populations are very small streams, from 5 to 20 feet wide (Behnke and Zarn 1976, USFWS 1983c). Reproductive site requirements are similar to those of other trout species—suitable gravel substrate (0.25 to 2.0 inches in size) with adequate flow to maintain oxygen requirements of incubating eggs is necessary for successful reproduction (Behnke and Zarn 1976). No innate migration patterns exist, only movements during spawning to the nearest site with suitable spawning substrate.

Greenback trout attain sexual maturity at 2 or 3 years of age. Spawning in steams occurs annually after first maturation, in spring and early summer, peaking when daily water temperature exceeds 45 °F. The female constructs a spawning bed in gravel, and several males are usually in attendance, with the dominant male constantly driving away subdominant males. The dominant male fertilizes most of the eggs during the spawning act, but smaller, subdominant males may dart in, shedding sperm, and fertilize some eggs. The female may construct and spawn in two or three spawning beds over several days. On average, females lay 700 to 1,000 eggs per pound of body weight (Behnke and Zarn 1976). After the eggs are spawned and fertilized, the female covers them with gravel. After this, no additional parental care is given to eggs or offspring. Maximum life span in small streams is typically 4 or 5 years, although in lakes greenbacks may live 8 to 10 years.

The greenback cutthroat trout was federally listed as endangered on April 18, 1978. Critical habitat has not been designated. In the late 19th century the greenback cutthroat trout was greatly reduced in abundance by toxic mine pollution, and irrigation diversions for agriculture. Problems that have added to the decline of the trout include water drawdown, water temperature alteration, siltation, and erosion (linked to grazing and general agricultural practices). Other factors that have impacted the subspecies include timber removal; hydroelectric power diversions; man-made pollution caused by effluents from industrial, human sewage, and agricultural practices; and physical damage to watersheds caused by such construction activities as highways, ski areas, and housing developments (Behnke 1976, Behnke and Zarn 1976, USFWS 1977). Non-native trout (brook, rainbow, brown and other subspecies of cutthroat trout) have also been widely introduced throughout the range of the greenback. All pure populations of the greenback cutthroat trout occur in tiny headwater streams above barriers to upstream migration that protect the subspecies from non-native trout. Any impact on trout habitat, such as the loss of riparian vegetation, flow depletion, and accelerated erosion, would affect a greenback trout population in the same manner as it would other species of trout.

Kendall Warm Springs Dace

The Kendall Warm Springs dace (*Rhinichthys osculus thermalis*) is associated with the numerous seeps and springs of the Kendall Warm Springs area and its outflow stream located along the north face of a small limestone ridge. The Kendall Warm Springs, which are hydrologically linked to the Green River, are located within the Bridger-Teton National Forest in Sublette County, Wyoming. Vegetation near Kendall Warm Springs includes grasses, forbs, and small shrubs and trees such as willow, sagebrush, and aspen. Aquatic vegetation surrounds the stream and is often very thick within the pools. The most common aquatic species are monkeyflower, moss, sago pondweed, and stonewort. Plant growth is extremely important because it provides both escape cover and nursery areas for fry (Binns 1978).

The Kendall Warm Springs dace occurs in the pools and mainstream eddies of Kendall Warm Springs and the outflow stream. The stream flows 985 feet before dropping over a calcium-rich embankment into the Green River. Average streamflow is approximately 0.007 ft/s and the average gradient is 4%. Stream width averages 6 feet and depth is usually less than 1 foot. Water temperature is approximately 85 °F at the spring source, although the outfall temperature may drop to 78 °F during the winter. Water within the Kendall Warm Springs area is slightly alkaline, well-mineralized and fairly high in dissolved solids. Carbon dioxide is high (12 ppm) and dissolved oxygen is low (0.55 ppm) at the source of Kendall Warm Springs. However, concentrations are modified as the water flows over rocks and gravel within the stream to the point of supersaturation of dissolved oxygen (Binns 1978).

The numbers of Kendall Warm Springs dace seem to correlate with dissolved oxygen and carbon dioxide levels, with fewer fish upstream and none at the Kendall Warm Springs source. Adults inhabit fairly shallow pools and streams, and plant growth within the water is necessary for escape cover and protection from the main current. Fry also use the vegetation as nursery areas (USFWS 1982d). Spawning for this subspecies probably occurs several times each year, and possibly throughout the year. Speckled dace reach maturity at 2 years.

The Kendall Warm Springs dace was federally listed as endangered on October 13, 1970. Critical habitat has not been designated. In the past, the Kendall Warm Springs have been subject to many human activities that have affected the dace within the Bridger-Teton National Forest. Cattle were allowed to graze and trample plant life in and around the springs area. Several rock dams (passage barriers) were built to create small pools for bathing and clothes washing, and soaps and detergents in the water have damaged aquatic organisms. Twenty-five feet of habitat was replaced by culverts along a road built in 1934. These culverts (passage barriers) may prevent the Kendall Warm Springs dace from moving upstream and may also isolate the upper half of the population. The Kendall Warm Springs dace was also used as bait by fishermen for many years (USFWS 1982d, Baxter and Simon 1970). Activities by the Wyoming Game and Fish Department and the Forest Service have removed these threats to the subspecies. Although the Kendall Warm Springs dace remains limited to its extremely small habitat in the springs, it is believed that optimum population levels have been reached there, and that there are no immediate threats to the species (USFWS 1982d).

Mediterranean Ecoregion

Aquatic habitats in the Mediterranean Ecoregion, located along the Pacific Coast and including most of California and a portion of Oregon, are influenced by a number of factors. Rivers along the coast receive medium to high inputs from rainfall. Surface runoff in the region is rapid, water storage is relatively short, and rivers are prone to low flows during times of drought (Myers et al. 1998). The Sierra Nevada mountains receive predominantly winter rain, and contain the headwaters for the Rogue, Klamath, and Sacramento rivers. The hills in the rainshadow of the coastal mountains experience relatively low annual rainfall, and support tributary rivers to the Sacramento and San Joaquin rivers. The Sacramento and San Joaquin rivers run through California’s Central Valley, a region that is heavily influenced by agricultural practices. These rivers have peak flows in February and experience low flows in September and October after the summer drought. They are also the main migratory corridors for a number of anadromous salmon species, and empty into the Pacific Ocean via the San Francisco Bay. The Mediterranean Ecoregion also supports vernal pool habitats, which provide habitat for a number of rare mollusk species.

Modoc Sucker

The Modoc sucker (*Catostomus microps*) is known from only a few widely separated tributary systems to the upper Pit River in northeastern California—the Rush-Ash Creek system and the Washington-Turner-Hulbert system (Moyle 1976, Ford 1977). This species occurs primarily in sections of stream with low or intermittent flow, or pools of the meadowlands (Moyle and Mariochi 1975, Moyle 1976, Ford 1977). In general, sites where Modoc suckers have been found are characterized by the following: low flows (intermittent in some); largely shallow pools; muddy bottoms; partial shade trees, shrubs, boulders, or undercut banks; abundant cover from riparian vegetation and undercut banks; and moderately clear water (Moyle and Mariochi 1975). Water temperatures (summer and fall) in Modoc sucker habitat range from 46 ^{o}F (fall) to 74 ^{o}F (summer; Ford 1977). Modoc suckers are omnivorous, feeding on detritus, diatoms, filamentous algae, chironomid larvae, crustaceans, and aquatic insect larvae. Adult suckers usually remain on the bottom or close to it (Martin 1972).

Spawning usually occurs from mid-April to the last week in May or the first week in June (Boccone and Mills 1979). Spawning occurs over coarse fine gravel in the lower end of pools with abundant cover. Water temperatures range from 56 to 61 ^{o}F. There is some evidence from Johnson and Washington Creeks of upstream migration by Modoc suckers to small intermittent tributaries, such as Higgins and Rice flats, during spawning season. Also, a possible spawning migration of Modoc suckers has been observed from Moon (Lake) Reservoir upstream into Cedar Creek.

The Modoc sucker was federally listed as endangered on June 11, 1985. Critical habitat has been designated in Modoc County, California. Designated habitat includes intermittent and permanent water and adjacent land areas that provide vegetation for cover and protection from soil erosion of all or portions of: Turner Creek, Hulbert Creek, Cedar Creek, Washington Creek, Coffee Mill Gulch, Johnson Creek, Higgins and Rice flats, and Rush Creek, Modoc County, California. The Modoc sucker is endangered because of its very restricted distribution combined with destruction of habitat. A major portion of the Rush Creek Modoc sucker habitat is on privately-owned land used for grazing sheep and cattle, which trample streambanks, thereby causing destruction of habitat through increased erosion of streambanks, removal of aquatic and riparian vegetation needed as cover, and siltation (Moyle 1976; Cooper et al. 1978; Mills 1980; Cooper 1983; Chesney 1985). Destruction of natural barriers to the Sacramento sucker by flooding areas for the creation of pastures, and by channelization, has resulted in losses through hybridization and backcrossing in several of the Modoc sucker streams (Ford 1977; Cooper et al. 1978; Mills 1980; Cooper 1983; Chesney 1985). Diversions of water for irrigation reduce the number and sizes of pools available to the Modoc suckers (Ford 1977). In addition, introductions of brown trout have added to the predation pressure on the Modoc sucker (Cooper et al. 1978; Mills 1980; Cooper 1983). Destruction of habitat by overgrazing and limited distribution of pure populations of the Modoc sucker still threaten the species (Ford 1977, Chesney 1985).

Owens Pupfish

Background information on the Owens pupfish can be found on page 5-38 (Subtropical Steppe/Subtropical Desert), where southwestern pupfish species are discussed.

Shasta Crayfish

The primary reference for this section is:
USFWS. 1998p. Recovery Plan for the Shasta Crayfish (*Pacifastacus fortis*). USFWS, Portland, Oregon.

The Shasta crayfish (*Pacifastacus fortis*) is the only surviving species of crayfish endemic to California. Populations of this species are limited to the midsections of the Pit River drainage, primarily the Fall River and Hat Creek subdrainages in Shasta County. The greatest densities of Shasta crayfish are found in the pristine headwater springs of the Fall River, a few of which support locally abundant isolated populations. The distribution of this species is tied to the distribution of lava cobbles and boulders that originated in the volcanic geology of the Modoc Plateau.

Shasta crayfish are generally found in cold, clear, spring-fed headwaters. In general, suitable habitat is defined by the availability of cover, or refugia, provided by clean lava cobbles and boulders on gravel or sand. Although potential food resources, temperature, and water chemistry constituents (e.g., dissolved oxygen, calcium, pH) may also limit the distribution of the Shasta crayfish, the range of conditions under which the species is found is considerable.

Shasta crayfish are active only at night, remaining hidden during the day. In general, they come out from hiding after dark to browse on the periphyton (i.e., the community of plants, animals, and associated detritus, or debris) that adhere to and form a surface coating on the abundant lava rocks. Crayfish that are found in the open during daylight have generally either been disturbed from their refuge or appear ill.

The primary food of the Shasta crayfish appears to be the periphyton and invertebrates that are abundant in the species' native environment. Other potential food resources include trout, sucker, and sculpin eggs, which are seasonally abundant. Although some of the items the crayfish will consume are known, nothing is known about the species' actual nutritional requirements.

Shasta crayfish are long-lived and slow-growing, and take approximately 5 years to reach sexual maturity. Mating occurs in October or November, when the male deposits a capsule containing sperm, or spermatophore, on the underside of the female. Shortly afterwards, the female lays 10 to 70 eggs, which she fertilizes with sperm from the spermatophore and then attaches to the underside of her abdomen or tail. In the spring, the eggs hatch into immature larval forms, or first instars, that are attached to the undersides of the female's abdomen by threads to the inner egg membrane. These first instars then molt into second instars, miniatures of the adult that clasp the female with their tiny claws. After a second molt, the third instars grow in size and eventually become free-living.

The Shasta crayfish was federally listed as endangered without critical habitat on September 30, 1988. The limited distribution of the species, coupled with its apparent decline, led to its endangered status. Overall, Shasta crayfish have a low abundance and fragmented distribution, with migration and genetic exchange between populations limited by hydroelectric development, natural barriers, and loss of habitat. The primary threats to the species are the introduction and expansion of non-native species, and disturbances related to land use practices.

Unarmored Threespine Stickleback

The unarmored threespine stickleback (*Gasterosteus aculeatus williamsoni*) has been extirpated from most of its range in Southern California, and is now limited to a small, remnant range. This range includes a small tributary in the San Francisquito Canyon in the upper Santa Clara River drainage in Los Angeles County; the Santa Clara River at Soledad Canyon, and the Del Valle area further downstream (NatureServe Explorer 2001). Unarmored threespine sticklebacks occur in shallow (< 3.3 feet deep) coastal streams often flowing through riparian woodlands within dry mixed rangeland. The streams always have a very low gradient, and usually do not support rainbow trout or speckled dace, which often occur in higher gradient reaches of the same drainages, sometimes along with low plated sticklebacks. Observations of the Soledad Canyon population indicate that the species prefers areas of moderate flow with vegetation for cover. Riffles and ponds are the major habitats available, and sticklebacks tend to be most numerous in small ponds with moderate flow. Most breeding takes place in small, man-made pools. Natural cover includes stream banks, rocks, sunken logs and, most importantly, vegetation (vascular plants and filamentous algae; Baskin 1974). Fry generally are found in vegetation, and presumably depend on it for protection from predatory fishes and invertebrates.

Although seasonal migrations are well documented for some threespine stickleback populations, freshwater sticklebacks, including those in southern California are not known to undertake migrations. However, they actively disperse as the aquatic habitat expands in the late fall, and they apparently are washed downstream during flooding (Baskin 1974; Irwin 1982; Bell 1974-1979).

In general, the males tend to establish territories and build nests on the bottom in shallow, still water near cover. The nests normally are constructed of decaying aquatic plant fibers, but males appear to accept a wide range of vegetation types for nest construction. Nests, which are built in shallow pits dug in sandy, muddy substrate, are generally constructed in or near vegetation (Kynard 1979). The female is courted, deposits eggs in the nest, and is then driven out of the territory. The male then returns to fertilize the eggs.

The unarmored threespine stickleback was federally listed as endangered on October 13, 1970. Critical habitat has not been designated. The following factors have been identified as persistent threats to the species: channelization, groundwater and surface water use (drawdown); introductions of exotic aquatic organisms; industrial and residential (urban) construction; agricultural development; the development of recreational parks in Soledad Canyon; and excessive growth of aquatic vegetation, which may reduce dissolved oxygen through plant respiration and decomposition.

Vernal Pool Shrimp
The primary reference for this section is:
USFWS. 1994e. Endangered and Threatened Wildlife and Plants; Determination of Endangered Status for the Conservancy Fairy Shrimp, Longhorn Fairy Shrimp, and the Vernal Pool Tadpole Shrimp; and Threatened Status for the Vernal Pool Fairy Shrimp. Final Rule. Federal Register 59(180): 48136-48153.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

The Conservancy fairy shrimp (*Branchinecta conservatio*), longhorn fairy shrimp (*B. longiantenna*), vernal pool fairy shrimp (*B. lynchi*), and vernal pool tadpole shrimp (*Lepidurus packardi*) are aquatic crustaceans that are endemic to vernal pools in California. The vernal pools in which these species occur are found in the Central Valley, the coast ranges, and a limited number of sites in the Transverse Range and Santa Rosa Plateau. All four species are sporadic in their distribution, often inhabiting only one or a few pools in vernal pool complexes that are quite widespread (Eng 1990, King 1992, Simovich 1992; Brusca 1992). None are known to occur in riverine waters, marine waters, or other permanent bodies of water.

The three fairy shrimp and the vernal pool tadpole shrimp are ecologically dependent on seasonal fluctuations in their habitat, such as absence or presence of water during specific times of the year, duration of inundation, and other environmental factors that include specific salinity, conductivity, dissolved solids, and pH levels.

The Conservancy fairy shrimp inhabits vernal pools with highly turbid waters. It is known from six disjunct populations, occurring in large pools with low conductivity, total dissolved solids, and alkalinity (Barclay and Knight 1984; Eng et al. 1990). The Conservancy fairy shrimp is usually collected at cool temperatures and appears to be relatively long-lived (Patton 1984; Simovich et al. 1992). This species has been observed from November to early April.

The longhorn fairy shrimp inhabits clear to turbid, grass-bottomed vernal pools in grasslands, and clear-water pools in sandstone depressions. The water in grassland pools inhabited by this species has very low conductivity, total dissolved solids, and alkalinity (Eng et al. 1990). This species is only known from four disjunct populations along the eastern margin of the central coast range. All vernal pools inhabited by this species are filled by winter and spring rains, and may remain inundated until June. The longhorn fairy shrimp has been observed from late December until late April.

The vernal pool fairy shrimp, although it has a relatively wide range, primarily occurs in vernal pools with clear to tea-colored water, most commonly in grass- or mud-bottomed swales, or in basalt flow depression pools in unplowed grasslands. However, one population occurs in sandstone rock outcrops, and another population occurs in alkaline vernal pools. The water in pools inhabited by this species has low total dissolved solids, conductivity, alkalinity, and chloride (Collie and Lathrop 1976). Vernal pool fairy shrimp have been collected from early December to early May.

The vernal pool fairy shrimp has a sporadic distribution within vernal pool complexes (Patton 1984; County of Sacramento 1990; Jones and Stokes 1992, 1993; Stromberg 1993; Sugnet and Associates 1993b), wherein the majority of pools in a given complex are not inhabited by the species. The species is typically found at low population densities (Simovich et al. 1992), and only rarely does it co-occur with other fairy shrimp species.

Although the vernal pool fairy shrimp can mature quickly, allowing populations to persist in shorter-lived pools, it also persists later into the spring where pools are longer lasting.

The vernal pool tadpole shrimp inhabits vernal pools containing clear to highly turbid water, and ranging in size from 54 square feet to 89 acres. Pools have low conductivity, alkalinity, and total dissolved solids (Barclay and Knight 1984, Eng et al. 1990). These pools are located most commonly in grass-bottomed swales of grasslands in old alluvial soils underlain by hardpan, or in mud-bottomed pools containing highly turbid water. The vernal pool tadpole shrimp is known from 18 populations in the Central Valley, and from a single pool complex located on the San Francisco Bay National Wildlife Refuge in the city of Fremont, Alameda County, California.

The life history of the vernal pool tadpole shrimp is linked to the phenology of the vernal pool habitat. After winter rainwater fills the pools, the populations are re-established from eggs that have been dormant in the dry pool sediments (Ahl 1991, Lanway 1974). Eggs hatch shortly after inundation, with sexually reproductive adults appearing in about 3 to 4 weeks after hatching (Ahl 1991). A female surviving to large size may lay up to six clutches of eggs, which are sticky, and readily adhere to plant matter and sediment particles (Simovich et al. 1992). A portion of the eggs hatch immediately, and the rest become dormant and remain in the soil to hatch during later rainy seasons (Ahl 1991). The vernal pool tadpole shrimp matures slowly and is a long-lived species (Alexander 1976, Ahl 1991). Adults are often present and reproductive until the pools dry up in the spring (Ahl 1991, Simovich 1992).

Nearly all fairy shrimp feed on algae, bacteria, protozoa, rotifers, and bits of detritus (Pennak 1989). The females carry eggs in an oval or elongate ventral brood sac. The eggs are either dropped to the pool bottom or remain in the brood sac until the female dies and sinks. The “resting” or “summer” eggs are capable of withstanding heat, cold, and prolonged desiccation. When the pools refill in the same or subsequent seasons some, but not all, of the eggs may hatch. The egg bank in the soil may be comprised of the eggs from several years of breeding (Donald 1983). The eggs hatch when the vernal pools fill with rainwater. The early stages of the fairy shrimp develop rapidly into adults. These non-dormant populations often disappear early in the season long before the vernal pools dry up.

Tadpole shrimp are primarily benthic animals that swim with their legs down. They climb or scramble over objects, as well as plow along in bottom sediments, and their diet consists of organic detritus and living organisms, such as fairy shrimp and other invertebrates (Fryer 1987, Pennak 1989). Female tadpole shrimp deposit their eggs on vegetation and other objects on the bottom. Vernal pool tadpole shrimp populations pass the dry summer months as dormant eggs in pool sediments. Some of the eggs hatch as the vernal pools are filled with rainwater in the fall and winter of subsequent seasons.

The Conservancy fairy shrimp, longhorn fairy shrimp, and vernal pool tadpole shrimp were listed as endangered on September 19, 1994. The vernal pool fairy shrimp was listed as threatened on the same date. On August 6, 2003, the USFWS designated approximately 1,184,513 acres of vernal pool habitat as critical habitat for these and other vernal pool species. Urban, water, flood control, highway, and utility projects, as well as conversion to agricultural use, have eliminated vernal pools in southern California (Riverside and San Diego counties), the Central Valley, and the San Francisco Bay area (Jones and Stokes Associates 1987). Factors that threaten these species include changes in hydrologic patterns, overgrazing, OHV use, and any human activities that alter the watershed of the vernal pools. For some species, continued development could destroy existing habitat.

Marine Ecoregion

The Marine Ecoregion Division, which is located in western Oregon and Washington, includes such aquatic systems as the Puget Sound region, the western portion of the Columbia River Basin (including its confluence with the Pacific Ocean), and the Willamette River Basin of Western Oregon. Only three TEP aquatic species that could potentially be affected by vegetation treatments on public lands occur locally in this ecoregion. The ranges of these two of these species (the Lost River and shortnose suckers) extend into the Mediterranean Ecoregion, which begins in southern Oregon.

Although few species of concern to this project are permanent residents of this ecoregion, numerous ESUs of Pacific Northwest salmon migrate through the Marine Ecoregion on their way to and from the ocean phases of their life cycles. A discussion of these species begins on page 5-1.

Oregon Chub

The primary reference for this section is:
USFWS. 1998q. Oregon Chub (*Oregonichthys crameri*) Recovery Plan. Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Oregon chub (*Oregonichthys crameri*) is a small minnow that is endemic to the Willamette River Basin in western Oregon. The species was formerly distributed throughout the Willamette River Valley in off-channel habitats such as beaver ponds, oxbows, side channels, backwater sloughs, low gradient tributaries, and flooded marshes (Snyder 1908b). The current distribution of the Oregon chub is limited to 20 naturally occurring populations (in the Santiam River, Middle Fork Willamette River, Coast Fork Willamette River, and several tributaries to the Mainstem Willamette River) and four recently reintroduced populations (at Wicopee Pond, East Ferrin Pond, Fall Creek Spillway Pond, and Dunn Wetland).

Oregon chub are found in slack water off-channel habitats such as beaver ponds, oxbows, side channels, backwater sloughs, low gradient tributaries, and flooded marshes. These habitats usually have little or no water flow, silty and organic substrate, and considerable aquatic vegetation as cover for hiding and spawning (Pearsons 1989, Markle et al. 1991). The average depth of Oregon chub habitats is typically less than 6 feet, and the summer temperatures typically exceed 61 ^{o}F. Adult Oregon chub seek dense vegetation for cover and frequently travel in the mid-water column in beaver channels or along the margins of aquatic plant beds. Larval chub congregate in nearshore areas in the upper layers of the water column in shallow areas (Pearsons 1989). Juveniles venture farther from shore into deeper areas of the water column. In the winter months, chub can be found buried in the detritus or concealed in aquatic vegetation. Fish of similar size classes school and feed together. In the early spring, Oregon chub are most active in the warmer, shallow areas of the ponds.

Oregon chub are obligatory sight feeders. They feed throughout the day and stop feeding after dusk (Pearsons 1989). Chub feed mostly on water column fauna, primarily minute crustaceans such as copepods, cladocerans, and chironomid larvae (Markle et al. 1991).

Oregon chub spawn from April through September. Before and after spawning season, chub are social and non-aggressive. Spawning behavior begins with the male establishing a territory in or near dense aquatic vegetation (Pearsons 1989). Behaviors associated with reproduction and courtship include territorial behavior between males, head rubbing, directing of females by males, and twirling of both fish during the release of egg and sperm. Spawning activity has only been observed at temperatures exceeding 61 ^{o}F.

The Oregon chub was federally listed as endangered on October 18, 1993. Critical habitat has not been designated. The species evolved in a dynamic network of slack water habitats in the floodplain of the Willamette River. Major alteration of the Willamette River for flood control and navigation improvements has eliminated most of the river's historic floodplain. This alteration has also impaired or eliminated the environmental conditions in which the Oregon chub evolved. Remaining suitable habitats have been invaded by non-native fish predators and competitors. Current threats to the species include continued habitat alteration; the proliferation of non-native fish and amphibians; accidental chemical spills; runoff from herbicide or pesticide application on farms or along roadways, railways, and powerline ROW; desiccation of habitats; unauthorized water withdrawals, diversions, or fill and removal activities; and siltation resulting from timber harvesting in the watershed.

Lost River and Shortnose Suckers
The primary reference for this section is:
USFWS. 1993k. Lost River (*Deltistes luxatus*) and Shortnose (*Chasmistes brevirostris*) Sucker Recovery Plan. Portland, Oregon.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The Lost River sucker (*Deltistes luxatus*) and shortnose sucker (*Chasmistes brevirostris*) are large, long-lived suckers endemic to the upper Klamath Basin of Oregon and California. Historical records indicate that the two species were once widespread and abundant within their range. The present distribution of the Lost River sucker includes Upper Klamath Lake and its tributaries, Clear Lake Reservoir and its tributaries, Tule Lake and the Lost River up to Anderson-Rose Dam, and the Klamath River downstream to Copco Reservoir (Beak Consultants Incorporated 1987; Buettner and Scoppettone 1990, 1991). The present distribution of the shortnose sucker includes Upper Klamath Lake and its tributaries, Klamath River downstream to Iron Gate Reservoir, Clear Lake Reservoir and its tributaries, Gerber Reservoir and its tributaries, the Lost River, and Tule Lake.

Lost River and shortnose suckers are omnivores that feed primarily on zooplankton and insects. Both species generally spawn in rivers or streams and then return to the lake (Buettner and Scoppettone 1990). However, both species have separate populations that spawn near springs in upper Klamath Lake (Klamath Tribe 1993). Larval suckers usually spend relatively little time in tributary streams before they migrate back to the lake. Migration from spawning sites can begin in May or June. During the day, larvae typically move to shallow (depths of less than 20 inches) shoreline areas in the river, over substrates of sand, mud, and concrete (Buettner and Scoppettone 1990). Larvae are generally found in close proximity to rooted aquatic vegetation, and appear to avoid areas devoid of vegetation (Coleman and McGie 1988). It is believed that the suckers once used the extensive marsh system of the lower river as nursery habitat. Much of this habitat has been replaced by gently sloping, sandy, unvegetated shorelines.

Adult Lost River and shortnose suckers usually spend relatively little time in tributary streams and migrate back to the Lake after spawning. Adults appear to prefer areas with relatively low densities of algae and good water quality in terms of pH and dissolved oxygen, such as areas of the lake near inflows from streams or springs.

The Lost River and shortnose sucker were federally listed as endangered on July 18, 1988. The designation of critical habitat for both species was proposed in 1994, but has not occurred. The limited distribution of both sucker species, combined with the level of agricultural development and associated water and land use threats within the drainage, make these fishes susceptible to past and present habitat loss and degradation throughout their distribution. Cumulative impacts of land management on public and private lands has led to the endangered status of the Lost River sucker and shortnose sucker, and continues to hinder their recovery. Inputs of sediment and nutrients, and changes in timing and duration of stream flow as a result of road building have altered lake habitats. Habitat has also been lost through construction of dams, diversion of water from streams, reclamation of wetlands, and other changes.

Species in Multiple Ecoregions

Arkansas River Shiner
The primary reference for this section is:
USFWS. 1998r. Final Rule to List the Arkansas River Basin Population of the Arkansas River Shiner (*Notropis girardi*) as Threatened. Federal Register 63(225): 64771-64799;

and

USFWS. 2001g. Final Designation of Critical Habitat for the Arkansas River Basin Population of the Arkansas River Shiner. Final Rule. Federal Register 66(65): 18001-18034.

References cited in this section are internal to the above-referenced documents. A complete list of these references is available from the USFWS Oklahoma Ecological Services Field Office, Tulsa, Oklahoma.

The Arkansas River shiner (*Notropis girardi*) is a small fish found in the Canadian River in New Mexico, Oklahoma, and Texas, and the Cimarron River in Kansas and Oklahoma, which are both rivers in the Arkansas River Basin. This species utilizes a broad range of microhabitat features. However, adults are uncommon in quiet pools or backwaters, and almost never occur in tributaries having deep water and bottoms of mud or stone (Cross 1967).

Arkansas River shiners are generalist foragers, feeding on both items suspended in the water column and items lying on the substrate (Bonner et al. 1997). In the Pecos River, fly larvae, copepods, immature mayflies, insect eggs, and seeds were the dominant items in the species' diet (Gido 1997). The Arkansas River shiner spawns in July, usually coinciding with flood flows following heavy rains (Moore 1944). It appears to be in peak reproductive condition throughout the months of May, June and July (Polivka and Matthews 1997) and may actually spawn several times during this period (Wilde 1998). Arkansas River shiner eggs are non-adhesive and drift with the swift current during high flows. Hatching occurs within 24 to 48 hours after spawning. The larvae are capable of swimming within 3 to 4 days; they then seek out backwater pools and quiet water at the mouth of tributaries where food is more abundant (Moore 1944). Adult shiners attain a maximum length of 2 inches.

Historically, Arkansas River shiners inhabited the main channels of wide, shallow, sandy-bottomed rivers and larger streams of the Arkansas River Basin (Gilbert 1980), and were once widespread and abundant throughout the western portion of the Arkansas River Basin in Kansas, New Mexico, Oklahoma, and Texas. The species is now almost entirely restricted to about 508 miles of the Canadian River in Oklahoma, Texas, and New Mexico. An extremely small population may also still persist in the Cimarron River in Oklahoma and Kansas. In addition, a non-native population of the Arkansas River shiner has become established in the Pecos River of New Mexico within the last 20 years (Bestgen et al. 1989), but is not federally listed.

The Arkansas River shiner was federally listed as threatened on December 23, 1998. On April 4, 2001, the USFWS designated approximately 1,148 miles of rivers and 300 feet of their adjacent riparian zones as critical habitat for the Arkansas River shiner. This designation includes portions of the Arkansas River in Kansas, the Cimarron River in Kansas and Oklahoma, the Beaver/North Canadian River in Oklahoma, and the Canadian/South Canadian River in New Mexico, Texas, and Oklahoma.

The primary reason for the decline of this species is the inundation and modification of stream discharge by impoundments, channel desiccation by water diversion and excessive groundwater pumping, stream channelization, and introduction of non-native species. The Arkansas River basin population is threatened by habitat destruction and modification from stream dewatering or depletion due to diversion of surface water and groundwater pumping, construction of impoundments, and water quality degradation. Competition with the non-indigenous Red River shiner contributed to diminished distribution and abundance in the Cimarron River. Incidental capture of the species during pursuit of commercial bait fish species may also contribute to reduced population sizes. Drought and other natural factors also threaten the existence of the Arkansas River Shiner.

Humpback Chub

The humpback chub (*Gila cypha*) is restricted to the Colorado River system, where it once ranged from western Colorado and southwestern Wyoming to northern Arizona and possibly California (NatureServe 2001). In the lower basin, the largest remaining population occurs in the Little Colorado and Colorado rivers in the Grand Canyon (Douglas and Marsh 1996). In the upper basin, concentrations now occur at Black Rocks (west-central Colorado)/Westwater Canyon and Cataract Canyon of the Colorado River; Desolation and Gray canyons of the Green River; and Yampa and Whirlpool canyons in Dinosaur National Monument, Green and Yampa rivers. The habitats occupied by humpback chub subpopulations are disjunct, but very similar in appearance. This fish prefers deep, swift water in canyon habitats with boulder substrate (Valdez and Clemmer 1982). In the Little Colorado River, the fish is also found associated with calcium-rich dams (Kaeding and Zimmerman 1983).

The humpback chub was federally listed as endangered on March 11, 1967. On March 21, 1994, the USFWS designated 1,980 miles of river in the Colorado River basin in portions of Colorado, New Mexico, Utah, Arizona, Nevada, and California as critical habitat for the humpback chub and three other species (razorback sucker, Colorado pikeminnow, and bonytail chub; USFWS 1994f). The Colorado River has been changed by the construction of mainstream dams, which have changed the water quality from muddy and turbulent to clear and cold. Alteration of the flow and temperature regime of the Colorado River by development projects (i.e., dams, irrigation, dewatering and channelization projects) is cited as the primary reason for the decline of the humpback chub and for its precarious position today (Minckley 1973, Kaeding and Zimmerman 1983, USFWS 1984f). The proliferation of introduced species (Tyus et al. 1982) and the resultant competition and predation may have contributed to the decline of the species (Behnke and Benson 1983). Pollution (pesticides), eutrophication, and other factors such as parasitism (a parasitic crustacean-Lernaea), changes in the food base, and fishing pressure also may have attributed to the species' decline (USFWS 1984f). The fragmentation of the Colorado River system by dams has served to isolate subpopulations of the humpback chub, thus reducing gene flow and the ability of subpopulations to adapt to changing conditions.

Bonytail Chub

The bonytail chub (*Gila elegans*) is restricted to the Colorado River system, where it presently exists in very low numbers in its natural riverine and manmade reservoir habitat. Formerly abundant throughout the Colorado River and its larger tributaries, the species has recently only been found in the Yampa River (Dinosaur National Monument), the Green River (Gray and Desolation canyons), the Colorado River (Black Rocks and Cataract Canyon (Kaeding et al. 1986), Lake Mohave (Arizona-Nevada border), and Lake Havasu (Arizona-California) (Minckley and Deacon 1991). In riverine areas, the species is considered a "big-river" or mainstream fish since few have ever been captured in small tributaries (USFWS 1985l). However, in rivers bonytails tend to use pools and eddies instead of areas of faster current, and in reservoirs they are found more in lacustrine rather than riverine habitat (Vanicek 1967, Minckley 1973).

The bonytail chub is generally considered to be an insectivore (Valdez and Clemmer 1982); however, little information is available on specific food habits. Young chubs presumably eat chironomid larvae and mayfly nymphs in the Green River, where juveniles consume terrestrial and aquatic insects and the adults consume terrestrial insects, plant debris, and filamentous algae (Vanicek 1967). No other information is known on river feeding preferences, but the species is reported to eat plankton and algae in reservoir habitats (Minckley 1973). The breeding behavior of bonytail chubs has been observed in Lake Mojave (Jonez and Sumner 1954), where approximately 500 fish congregated over a gravel bar in 29.5 feet of water. Females were escorted by 3 to 5 males, and deposited eggs randomly with no indication of parental care.

The bonytail chub was federally listed as endangered April 23, 1980. On March 21, 1994, the USFWS designated 1,980 miles of river in the Colorado River basin in portions of Colorado, New Mexico, Utah, Arizona, Nevada, and California as critical habitat for the bonytail chub and three other species (razorback sucker, Colorado pikeminnow, and humpback chub; USFWS 1994f). The primary reasons for the decline of this species include flow depletions, loss of riverine habitat, dams, mining impacts and the resulting siltation, incidental capture, and the introduction of exotic fish. Presently, the fish is very rare, and its low population numbers impact the ability of the species to effectively reproduce. In addition, changes in the river flow regimes may be forcing the roundtail chub and the bonytail chub to reproduce in closer proximity (USFWS 1985l). Terrestrial habitats/areas within the range of the bonytail chub that may impact the riverine habitat include transportation/utility/communication corridors and facilities and shrub/brush and pinyon-juniper rangelands. Agricultural activities basin-wide include wheat, corn, beans (vegetables), pasture, and fruit (peaches and apples, especially in the Grand Junction, Colorado area).

Rio Grande Silvery Minnow

The primary reference for this section is:
Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation: Wildland Urban Interface Fuel Treatment. Forest Service, Southwestern Region. Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Rio Grande silvery minnow (*Hybognathus amarus*) is native to the Rio Grande Basin, historically occurring from Espanola, New Mexico, to the Gulf of Mexico (Bestgen and Platania 1991, USFWS 1994g). It also occurred in the Pecos River, a major tributary of the Rio Grande, from Santa Rosa, New Mexico, to its confluence with the Rio Grande in South Texas (USFWS 1994g). The species is now restricted to a 163-mile reach of the Rio Grande from around Cochiti Dam downstream to Elephant Butte Reservoir in New Mexico (USFWS 1999g). Within this reach, the silvery minnow is rare north of Albuquerque, uncommon between Albuquerque and Isleta, seasonally common between Isleta and San Acacia, and relatively common between San Acacia and the inlet of Elephant Butte Reservoir. Seventy percent of the remaining minnow population is reported to reside between San Acacia Diversion Dam and the headwaters of Elephant Butte.

The Rio Grande silvery minnow is herbiverous, with algae apparently an important food source. Spawning occurs during a brief period in late spring to early summer (May to June) when water temperatures are between 68 and 75 °F. Spawning coincides with spring runoff. The silvery minnow is a pelagic broadcast spawner, with semi-buoyant, non-adhesive eggs (Platania and Altenbach 1998, Propst 1999). Following fertilization, eggs drift with the current for up to 50 hours. Hatching time is temperature-dependent. Larvae drift for about a day after hatching and then move into low velocity habitats where food is abundant.

The Rio Grande silvery minnow was federally listed as endangered on July 20, 1994. A final rule designating critical habitat for this species was published on August 5, 1999. The only area designated as critical habitat is the area of the active channel of the mainstem Rio Grande in which this species is currently known to exist (USFWS 1999g). The decline of this species can be attributed to the modification of stream discharge patterns and channel desiccation by dams, water diversion, stream channelization (Bestgen and Platania 1991; Cook et al. 1992), competition and predation by introduced non-native species, and water quality degradation (USFWS 1999g). Because the range of this species has been so greatly restricted, it is also extremely vulnerable to a single, naturally occurring event. Because population numbers are highly variable both seasonally and annually (Propst 1999), a poor reproductive year could also prove devastating to this species.

Razorback Sucker

The primary reference for this section is:
Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation: Wildland Urban Interface Fuel Treatment. Forest Service, Southwestern Region. Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The razorback sucker (*Xyrauchen texanus*) was once abundant in the Colorado River and its major tributaries throughout the Colorado River Basin. However, this range has been much reduced. In the upper Basin, razorbacks are still widely distributed in the Green River Basin, with the largest concentrations in the upper Green River, and a small population in the lower Green River (Tyus 1987; McAda et al. 1994, 1996; Muth et al. 1998). In the Upper Colorado River, most documented occurrences have come from the Grand Valley area. A few suckers have been sampled in the mainstem of the Colorado River, downstream of the Green River confluence. Individuals have been captured in the San Juan arm of Lake Powell, and a few specimens have been confirmed in the river portion of the San Juan.

Present distribution in the lower basin includes extant populations in lakes Mohave and Mead, and small numbers in the Grand Canyon and downriver from Davis Dam to the Mexican border. No substantial recruitment to any population has been documented in recent years. Juveniles are most often collected from irrigation canals in Arizona and California. Hatchery-raised razorback suckers have been stocked into the mainstem and tributaries of the Salt, Verde, Gila, and lower Colorado rivers during the past decade.

The razorback sucker is a long-lived species that spawns in the late winter to early summer, depending on local water temperatures. In general, temperatures between 50 and 68 °F are appropriate for spawning (Bestgen 1990). Larvae and juveniles suffer very high mortality from predation, particularly from non-native species. For the first period of life, larval razorback suckers are nocturnal, hiding during the day. Their diet during this period consists mostly of plankton (Marsh and Langhorst 1988). Young fish grow fairly quickly, but growth slows once adult size is reached (McCarthy and Minckley 1987). The diet of adults consists of midge larvae, planktonic crustaceans, diatoms, filamentous algae, and detritus. For the most part, razorback suckers are bottom feeders, but they do contain mouthparts that are characteristic of planktonic and detrital feeding habits.

The razorback sucker was federally listed as endangered on October 23, 1991. In addition, critical habitat has been designated in 15 river reaches containing about 49% of the species' historic habitat (1,724 miles) within the Colorado River Basin and its 100-year floodplain. The decline of this species is primarily attributable to the impoundment of large portions of the Colorado River and its tributaries. These impoundments have altered habitat, substantially reducing flows in some reaches and modifying temperature regimes in others. In addition, recruitment of the species is limited by extreme predation pressure from introduced, fish-eating predators.

Colorado Pikeminnow
The primary reference for this section is:
Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation: Wildland Urban Interface Fuel Treatment. Forest Service, Southwestern Region, Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Colorado pikeminnow (formerly Colorado squawfish; *Ptychochelius lucius*) is the largest member of the minnow family native to North America (Miller 1961, Behnke and Benson 1983). This species formerly inhabited the Colorado River basin from its mouth in Baja, California, upstream to southern Wyoming (Propst 1999). Currently, the Colorado pikeminnow is primarily limited to three areas in the Upper Colorado River basin. In these areas, the fish is common only in the Green-Yampa River system of northwestern Colorado and northeastern Utah (Tyus 1990, 1991; Propst 1999). Reproducing populations still occur in the eastern part of Colorado in the Colorado and Gunnison rivers (Osmundson and Kaeding 1989, Osmundson and Burnham 1996), and in the San Juan River of New Mexico (Platania et al. 1991; Ryden and Ahlm 1996; Propst 1999). In the lower Colorado River Basin, pikeminnows have been reintroduced into the Salt and Verde river systems.

Juveniles of this species feed on insects and crustaceans, while individuals over 1.2 inches in length feed on fish. As adults, Colorado pikeminnows are almost exclusively fish-eaters (Vanicek and Kramer 1969). Spawning occurs between late June and about mid-August, depending on local hydrology and temperature regimes. Spawning coincides with rising water temperature and decreasing flow, with peak spawning activity occurring between 72 and 77 °F (Vacinek and Kramer 1969, Tyus 1990). Spawning areas are a complex of deep pools, eddies, and fast-moving water over cobble substrates (Miller 1995, Popst 1999). Eggs are broadcast over gravel and cobble substrates in riffles and rapids. After hatching, the larvae drift downstream to nursery areas (Tyus and Haines 1991). Nursery areas consist of shoreline, backwater, and embayment areas (Haynes et al. 1984; Haines and Tyus 1990). Migration is an important component in the reproductive cycle, and Colorado pikeminnows have been observed migrating over 186 miles to specific river reaches to spawn (Tyus 1985, 1986).

The Colorado pikeminnow was federally listed as endangered in 1967. In March, 1994, the USFWS designated 1,148 miles (29% of the species' historical range) as critical habitat. Six reaches of the upper Colorado River basin were included, five of which are located in Colorado and Utah and a sixth on the San Juan River in New Mexico. This species was nearly driven to extinction, primarily by water development programs, such as dams, that have altered stream morphology, flow patterns, temperatures, water chemistry, and silt loads of most major streams throughout the Colorado River Basin. Access to most spawning areas have also been blocked by dams. Interactions with non-native fishes may be an important factor in the continued survival or success of reintroduced populations of the Colorado pikeminnow. Predation by channel catfish, smallmouth bass, and flathead catfish are threats to this species.

Effects of Vegetation Treatments on Aquatic Species

This analysis considered TEP aquatic animal species and their critical habitat that are located within the project area. The potential effects of vegetation treatments on these species and their habitats are discussed below. Although the 98 aquatic species considered in this document have a variety of different habitat requirements and different abilities to tolerate changes to aquatic systems, there are some broad effects that treatments would be likely to have on all aquatic habitats and species. In general, any activity with the potential to alter aquatic habitats would also have the potential to affect the TEP species found in those habitats.

Given the programmatic nature of this document, the effects analysis that follows is necessarily general in nature, providing an overview of the sorts of effects that are likely to occur to aquatic species as a result of vegetation treatments. Local BLM offices, which have been monitoring many of these species and their habitats for years, have additional information about TEP aquatic species and their habitat requirements that will allow a more detailed analysis of effects than is feasible at the programmatic level.

Impacts from the proposed vegetation treatments may affect one or more specific life history requirements of the TEP species considered in this BA. For instance, the spawning, rearing, and feeding requirements of a particular TEP species may be very specific within that species' habitat. Therefore, effects must consider multiple life stages and thus multiple habitat needs of a particular species.

All else being equal, the potential impacts to TEP aquatic species that are narrowly endemic may be greater than the potential impacts to species that are more broadly ranging. However, local offices will often be able to provide more specific guidance for avoiding or minimizing impacts to narrow endemics. For this reason, it is likely that vegetation treatments could be better fine-tuned to ensure that these narrow endemics would not be adversely affected.

Effects Common to All Treatment Methods

The direct effects of vegetation treatments on aquatic species are discussed by specific treatment type in the appropriate sections that follow. Since all methods are similar in that they remove and/or manipulate vegetation, the primary indirect effects that are common to all treatment types are discussed here in order to avoid repetition in the sections that follow. In general, the vegetation treatments proposed by the BLM are expected to have short-term adverse and long-term beneficial effects on aquatic habitats. Combined with minimization measures and project design criteria, it is anticipated that adverse effects would be minimized. In addition, all projects would be implemented with the objective of creating long-term beneficial effects on TEP species and their habitats.

Indirect Effects

A general reduction in the plant biomass of riparian areas, which could occur by any of the treatment methods proposed for use on public lands, can have multiple consequences for aquatic species including an increase in water temperature and sedimentation, and a decrease in water storage capacity (USDA Forest Service 2000). Riparian cover provides shade to aquatic habitats, which cools water temperatures, and reduces the extent of water temperature fluctuation. In addition, riparian vegetation stabilizes the soil on banks, preventing erosion and sedimentation into streams and other aquatic habitats, and intercepts rainfall to reduce overland flow. Riparian vegetation also increases habitat quality by buffering streams from incoming sediments and other pollutants, building a sod of herbaceous plants to form undercut banks, increasing habitat complexity, and increasing terrestrial invertebrate prey for fish species (Platts 1991).

Increased sedimentation entering aquatic habitats as a result of destabilized streambanks and increased erosion can cover spawning/rearing areas, thereby reducing the survival of fish embryos and juveniles (USDA Forest Service 2000). Sedimentation can also fill pool habitats, making them unusable by fish and other aquatic organisms. A number of sublethal effects to aquatic species may also occur as a result of sedimentation, including avoidance behavior, reduced feeding and growth, and physiological stress (Waters 1995). Over the long-term, increased sediment loads reduce primary production in streams (USDA Forest Service 2000). Reduced instream plant growth, combined with the reductions in riparian vegetation, can limit populations of terrestrial and aquatic insects, which also serve as food sources for many TEP fish species.

The increased solar radiation that results from the loss of streamside (or poolside, etc.) vegetation causes temperatures, light levels, and autotrophic production (i.e., plants and algae) to increase. The resulting effects on some TEP species, and particularly salmonids, may be reduced growth efficiency, an increased likelihood of succumbing to disease, and an increase in food production.

By exposing more surface area of soil directly to rainfall, and increasing the overland flow of water into the aquatic habitat, removal of vegetation may result in decreased water storage capacity of the soil. Over the long-term, overland flow can erode the topsoil and cut rills and gullies or deepen existing gullies, concentrating runoff. As a result, sediment production is increased. Reduced infiltration and increased runoff may decrease the recharge of the saturated zone and increase peak flow discharge. Thus, the amount of water retained in the watershed to sustain base flows is reduced.

Increases in streamflow can lead to alterations in channel morphology. Doubling the speed of streamflow increases its erosive power by 4 times and its bedload and sediment carrying power by 64 times (USDA Forest Service 2000). Accelerated runoff can thus cause unstable stream channels to downcut or erode laterally, accelerating erosion and sediment production. Lateral erosion results in progressively wider and shallower stream channels, which can adversely affect fish populations. Pool/riffle and width/depth ratios, which are important habitat components for many TEP aquatic species, may also be altered.

The severity of the effects would vary by treatment method, location, the amount of plant material removed, and the distance from the aquatic habitat. Most of the effects would also be increased in severity if vegetation were removed prior to a period of heavy precipitation. Therefore, timing of the treatments is another important factor. The effects of vegetation removal would persist until riparian areas were revegetated.

Over the long term, all treatment methods that remove non-native and competing vegetation are likely to have a beneficial effect on the habitat of aquatic species, provided that native or other desirable plant species are returned to those habitats after the treatments. Noxious weeds can have substantial negative effects on stream/riparian areas by outcompeting more desirable riparian vegetation, reducing biodiversity, altering aquatic habitats (e.g., reducing streambank protection, undercut bank cover, overhanging vegetation cover, pool depth and volume, and detrital and nutrient inputs; and increasing erosion and fine sediment deposition, stream width, and thermal relationships), and altering natural ecosystem processes (National Fire Plan Technical Team 2002). Vegetation treatments that target plant communities adjacent to aquatic habitats should result in conditions that would be more suitable for supporting aquatic species. Therefore, vegetation treatments would eventually increase the amount of suitable habitat, potentially leading to an increase in TEP species populations.

A long-term benefit of the removal of fuels from riparian habitats is the decrease in the risk of a future high intensity wildfire. Because past fire suppression has radically altered vegetation structure and fuel loads, the risks for stand-replacing fires in areas that historically experienced lower intensity and lower frequency burns are now at all-time highs on some public lands where treatments are proposed to occur. In many cases, fuels reduction is the primary intent of these treatments. In the absence of such activities, a wildfire burning through watersheds that support TEP species could potentially have much worse effects on aquatic habitats and these species than any of the treatment methods themselves. A full discussion of the potential effects of fire on aquatic habitats and species is found in the following section on prescribed burning. Fire retardants that are commonly used to halt the spread of wildfires can be toxic to aquatic organisms if they reach surface waters, and may also alter primary and secondary production (Spence et al. 1996). When mixed with water and exposed to ultraviolet radiation, fire retardants break down into hydrogen cyanide, a substance that is extremely toxic to aquatic life (Fresques et al. 2002). In highly alkaline waters, high concentrations of ammonia, another lethal substance, can also be produced. Apart from direct mortality to TEP species, retardants can also kill their invertebrate food items, and the phosphates in the retardants can cause eutrophication of downstream reaches. Any treatment method that reduces ignitable fuels would minimize the chances that these harmful chemicals would need to be used in watersheds that support TEP species.

Prescribed Fire Treatments

Direct Effects

The direct effects of prescribed burning on aquatic species and their habitats include the heating of water and immediate chemical changes to aquatic habitats. Depending on its size and intensity, large quantities of heated slag and ash produced during a prescribed burn could enter the water, briefly raising water temperatures to lethal limits (Fresques et al. 2002). The accompanying changes in pH and increased levels phosphate as a result of phosphate leached from ash can also impair water quality. Ash created by wildfires or prescribed burning has been documented to have life-threatening effects on some species of fish (Agyagos et al. 2001), and could therefore directly affect TEP species. These effects would be short-term in duration.

Indirect Effects

Prescribed fire can substantially alter a streamside habitat through the removal of large amounts of vegetation. The indirect effects of biomass removal on aquatic species have already been discussed under Effects Common to All Treatment Methods. However, additional indirect effects would be possible. A fire capable of consuming a large amount of vegetation and exposing a large area of bare soil would likely result in a surge of nutrients into the aquatic system. This temporary increase in nutrients could temporarily benefit many TEP fish species by increasing food production.

The introduction of ash into an aquatic habitat, as discussed under direct effects above, would contribute to the degradation of water quality. In addition, if a foam line were used as a firebreak near an aquatic system, aqueous firefighting foam could potentially leach into the water. Other chemicals that could be released or leach into aquatic habitats include ignition fuels, or fuels used to power equipment (e.g., helicopters, vehicles, and mechanical equipment), which would further degrade the water quality.

Firelines created using manual or mechanical means would affect aquatic habitats in a manner similar to manual and mechanical vegetation treatments under the project.

Snags and other woody debris that falls into the aquatic habitat provide the principal structural features that shape the stream's morphology, linkages to the floodplain, habitat complexity, streambed materials, and other characteristics (National Fire Plan Technical Team 2002). Therefore, a prescribed fire intense enough to consume trees and snags would eventually have an adverse effect on habitat for numerous TEP species by eliminating future habitat resources.

Some activities associated with prescribed fire, such as creating wet lines and extinguishing hot spots after the majority of the fire has gone out, require the availability of a nearby water source. Water may be needed to fill portable pumps, pumps mounted to fire engines or water tenders, or 100- to 250-gallon buckets suspended by helicopters. Use of water from aquatic habitats that support TEP species could adversely affect those habitats, particularly in arid climates or during dry seasons, when water is limited. Taking water from aquatic habitats with TEP species could also result in inadvertent entrainment and/or harassment of those species.

Other potential indirect effects include setting up camps close to aquatic habitats or constructing roads to gain access to treatment sites, which would increase the potential for sedimentation into aquatic habitats. New roads would also increase the accessibility of the site, potentially resulting in increased human disturbance in the future, and increasing the spread of weeds onto the site.

Over the long term, a well-managed prescribed fire would have a beneficial effect on TEP aquatic species, as a result of improved and rejuvenated habitat, as well as increased productivity (Minshall and Brock 1991, Burton 2000). Over the long term, there could also be an increase in populations of TEP species as a result of a more healthy functioning ecosystem. This benefit would especially be true for riparian habitats that were historically subject to frequent, low intensity burns. Both the condition of the site prior to burning and the intensity of the burn would influence whether the end result of the fire was beneficial. Even a high intensity burn could eventually have a beneficial effect on riparian/aquatic habitats, especially if site restoration measures were followed post-burn.

A well-planned and managed prescribed burn would also reduce the risks of a future, high-intensity wildfire in riparian habitats, as well as the risks associated with suppressing such a fire, as discussed under Effects Common to All Treatment Methods. Because the BLM would follow guidance dictated by the *National Fire Plan* (USDI and USDA 1995), high intensity fires would not be set in sensitive habitats, and many of the adverse effects listed in this section would therefore be minimized. The proper fire management plan would involve fuels reduction and other measures designed to reduce the intensity of a prescribed fire in high wildfire-risk areas. Therefore, setting a controlled prescribed fire near an aquatic habitat would be likely to benefit TEP species by reducing the likelihood that the worst-case-scenario effects from fire, as discussed in this section, would occur.

Mechanical Treatment Methods

Direct Effects

Few direct effects to aquatic TEP species and their habitat would be likely as a result of mechanical treatment methods, unless these activities were conducted immediately adjacent to an aquatic habitat that supports TEP species or that is critical to their survival. Leaking of equipment fuel directly into the water would decrease water quality. In addition, the use of heavy equipment in riparian areas could lead to bank collapse, which would also degrade riparian habitat. If vehicles were allowed directly into aquatic habitats, additional effects would be likely.

Indirect Effects

Apart from the indirect effects to TEP aquatic species caused by the removal of large amounts of vegetation from riparian habitats (see Effects Common to All Treatment Methods), a number of additional effects could occur as a result of mechanical treatments. Mechanical treatments often disturb the soil during vegetation removal (e.g., chaining, tilling, and grubbing), increasing the potential for sediment transport into the stream. The closer these activities occur to the aquatic habitat, the greater their potential effect on the TEP species therein. Soil disturbance also increases the likelihood that weeds will recolonize the site (Sheley et al. 1995). Therefore, reseeding or some other form of site restoration would be crucial in order for mechanical treatment methods to benefit riparian habitats/aquatic species.

Mechanical treatments that uproot plants (e.g., chaining, tilling, grubbing, feller-bunching) decrease slope stability in riparian areas. The root strength of plants in riparian areas, particularly trees and shrubs, contributes to slope stability. Therefore, the removal of roots may lead to increased incidence of erosion and debris slides and flows (Sidle et al. 1985). Substantial impacts would be most likely if woody vegetation on slopes directly adjacent to aquatic habitats were removed. Further from the water, where the contribution of root strength to maintaining streambank integrity declines, effects would be proportionally less severe (National Fire Plan Technical Team 2002).

Because mechanical treatments can be used to remove trees and shrubs, some activities in riparian areas may remove plants and woody materials that would eventually become coarse woody debris, an important habitat element for many aquatic species. These effects to habitats would be greatest if woody vegetation within the distance of one tree height away from the channel were removed (National Fire Plan Technical Team 2002). Further from the water, the probability that a falling tree will enter the stream channel is much reduced, and the indirect effects of future coarse woody debris removal on aquatic habitats become less important.

Apart from the removal of noxious weed species, mechanical treatment methods in riparian areas can have a long-term beneficial effect on aquatic habitats by reducing woody overgrowth and other overabundant fuels. The removal of excess fuel that would not have been present under historical fire regimes can return riparian habitats to much healthier states. In addition, removal of these fuels would reduce the risk that a future stand-replacing or catastrophic fire would burn through riparian areas. It is for this reason that mechanical treatments are often used prior to prescribed burns to reduce fuels. With adequate buffers to ensure bank stability and coarse woody debris recruitment, and measures to reduce sedimentation into streams (see Conservation Measures section), mechanical treatments can help restore riparian areas to their historical states, without damaging aquatic habitats over the short term.

Manual Treatment Methods

Direct Effects

Direct effects to aquatic TEP species or their habitat are not anticipated to result from manual treatment methods.

Indirect Effects

The indirect effects associated with vegetation removal, as discussed under Effects Common to All Treatment Methods, could potentially occur with manual treatment methods. However, since manual treatment methods are only economically feasible for limited weed infestations, it is anticipated that these effects would not be as extreme as those resulting from more extensive biomass removal methods (e.g., fire or mechanical control). Manual treatment methods are typically associated with minimal environmental impacts, and as such are often appropriate for sensitive habitats, such as riparian areas. Some soil disturbance would occur during the removal of plants from the soil, but it would not be widespread and should not have a major effect on aquatic habitats. Provided manual methods are used appropriately (e.g., for small infestations and where native vegetation will replace the pulled weeds), effects of this treatment method should be beneficial.

Biological Control Treatments

Domestic Animals

Direct Effects

The potential direct effects of domestic animals on aquatic species and their habitats would minimal, provided the animals did not enter aquatic habitats. If animals were allowed to wallow and wade directly in the water, there could be some mortality or injury to TEP species, primarily eggs and pre-emergent fry, but also adults of smaller fishes. The input of domestic animal feces into aquatic habitats also degrades water quality.

Indirect Effects

In addition to the removal of vegetation, the disturbance to the soil caused by the movement of domestic animals in riparian and aquatic habitats can induce increased sedimentation. Grazing can also widen stream channels, promote incised channels, lower water tables, reduce pool frequency, and alter water quality (USFWS 1999h). The extent of these effects would vary depending on the number of animals used for the treatment, and the intensity and duration of the treatment. Under more intensive weed containment scenarios, mass erosion from trampling, sliding hooves, and streambank collapse could cause soils to move directly into the stream (USDA Forest Service 2002). Undercut banks, which often provide shelter to fish species, could be damaged or collapse in grazed areas, thus decreasing the amount of available fish habitat. In addition, heavy trampling could cause soil compaction, which reduces the infiltration of overback flows and precipitation into riparian soils.

Domestic animals could also degrade the quality of riparian and aquatic areas by facilitating the spread of non-native species in these habitats. These animals carry plant propagules on their hooves and in their fur, and can also release them in their feces.

Other Biological Control Agents

Direct Effects

Direct effects to aquatic TEP species or their habitat is not anticipated to result from the use of pathogens, insects, and similar organisms as biological control agents.

Indirect Effects

Use of biological control agents in aquatic and riparian habitats would result in the loss of some vegetation, so the general effects discussed under Effects Common to All Treatment Methods could potentially occur. Unlike under other treatment methods, however, the loss of vegetation resulting from biocontrol agents would be gradual, and therefore less likely to have a noticeable effect on aquatic systems. Some soil disturbance resulting from workers releasing agents in riparian areas could occur, but would be unlikely to have substantial effects on aquatic habitats.

Biological control agents would be thoroughly tested, and permitted by USDA Animal and Plant Health Inspection Service (APHIS) prior to release. Despite these safeguards, there is always a risk that the release of an organism into a habitat in which it does not normally occur can result in unforeseen ecological repercussions. These unanticipated effects of biological control agents would be impossible to predict, and it is believed that the appropriate precautions would be taken to prevent their occurrence.

Herbicide Treatments

Direct Effects

Aquatic TEP species could potentially come into contact with herbicides if sprayed formulations were to enter aquatic habitats during the application process, either through direct spray of the water by herbicides approved for use in aquatic habitats (i.e., diquat, fluridone, and certain formulations of 2,4-D, glyphosate, imazapyr, and triclopyr), accidental spray of the water by terrestrial herbicides, or off-site drift or surface runoff of herbicides sprayed in nearby upland habitats into aquatic habitats. Chemicals could also enter aquatic habitats during an accidental spill of herbicides before, during, or after the treatment. Aquatic species inhabiting water bodies exposed to herbicides would potentially come into contact with contaminated water. The potential risks to aquatic animals as a result of such direct contact with herbicides approved for use by the BLM were assessed in ERAs. New ERAs completed by the BLM in support of this BA address the risks to aquatic organisms associated with exposure to herbicides via each of the abovementioned exposure pathways, as summarized in Tables 5-2 through 5-5. The previously-completed Forest Service ERAs addressed three scenarios for aquatic organisms: an accidental spill, an acute exposure to a peak concentration of an herbicide in water as a result of a normal application (i.e., either through accidental direct spray or runoff from an adjacent application site), and a longer-term exposure to a contaminated aquatic habitat. The acute exposure scenario includes three levels of exposure: with the upper range representing an accidental direct spray, the lower range representing runoff in relatively arid regions, and the central range representing runoff in an area that is susceptible to runoff. The Forest Service risk assessments assume a universal aquatic habitat, which is representative of both a small pond and a small stream that were modeled in the BLM ERAs. Therefore, for Forest Service chemicals, results for ponds and streams (in Tables 5-2 through 5-5) are identical.

TABLE 5-2
Summary of Effects to TEP Fish in Ponds

Herbicide	Direct Spray	Off-site Drift	Spill	Surface Runoff
2,4-D	No effects	Not addressed in ERA	Adverse effects	No effects
Bromacil	Adverse effects	No effects	Adverse effects	Adverse effects
Chlorsulfuron	No effects	No effects	No effects	No effects
Clopyralid	No effects	Not addressed in ERA	Adverse effects	No effects
Dicamba	No effects	No effects	No effects	No effects
Diflufenzopyr	No effects	No effects	No effects	No effects
Diquat[1]	Adverse effects	NA	Adverse effects	NA
Diuron	Adverse effects	Adverse effects (maximum application rate)	Adverse effects	Adverse effects
Fluridone[1]	No effects	NA	Adverse effects	NA
Glyphosate	Adverse effects (maximum application rate; typical and maximum rates using more toxic formulation)	Not addressed in ERA	Adverse effects	No effects
Hexazinone	No effects	Not addressed in ERA	Not addressed in ERA	No effects
Imazapic	No effects	No effects	No effects	No effects
Imazapyr	No effects	Not addressed in ERA	Adverse effects	No effects
Metsulfuron methyl	No effects	Not addressed in ERA	Adverse effects (maximum application rate)	No effects
Overdrive®	No effects	No effects	No effects	No effects
Picloram	Adverse effects	Not addressed in ERA	Adverse effects	No effects
Sulfometuron methyl	No effects	No effects	No effects	No effects
Tebuthiuron	No effects	No effects	Adverse effects	No effects
Triclopyr acid	No effects[2]	Not addressed in ERA	Adverse effects	No effects
Triclopyr BEE	Adverse effects	Not addressed in ERA	Adverse effects	Adverse effects (maximum application rate)

[1] Diquat and fluridone are used to control aquatic weeds; direct application into a pond or stream is a typical use. Off-site drift and surface runoff scenarios do not apply, since these herbicides would not be applied in upland areas.
[2] For this herbicide, "direct spray" also considers a normal aquatic application directly into the water column.
Note: "Adverse effects" means ERAs predicted risks at both typical and maximum application rates, unless otherwise indicated.
NA = Not applicable.

The potential toxicological effects of herbicides on aquatic organisms, which were examined in ERAs, include mortality and sublethal effects. Examples of sublethal effects include altered behavior, stunted growth, reduced reproductive success, and physiological changes that make the organism more susceptible to environmental stresses (Spence et al. 1996). In this discussion, the term "adverse health effects" refers to the abovementioned or similar toxicological effects at the level of the organism. In addition, it is assumed that for TEP fish and aquatic invertebrates, these adverse health effects would potentially result in population-level effects for the species in question. Because many aquatic TEP species already have reduced, sensitive populations, mortality of individuals or reduced reproductive output could reduce the size of affected populations further, perhaps even leading to extirpation. Furthermore, if individuals were to become more physiologically predisposed to mortality from environmental stresses (such as predation, exposure to harsh environmental conditions), the risk for future population-level effects, including extirpations, would be increased.

TABLE 5-3
Summary of Effects to TEP Fish in Streams

Herbicide	Direct Spray	Off-site Drift	Spill[1]	Surface Runoff
2,4-D	No effects	Not addressed in ERA	Adverse effects	No effects
Bromacil	Adverse effects	No effects	Adverse effects	No effects
Chlorsulfuron	No effects	No effects	No effects	No effects
Clopyralid	No effects	Not addressed in ERA	Adverse effects	No effects
Dicamba	No effects	No effects	No effects	No effects
Diflufenzopyr	No effects	No effects	No effects	No effects
Diquat[2]	Adverse effects	NA	Adverse effects	NA
Diuron	Adverse effects	Adverse effects (maximum application rate)	Adverse effects	Adverse effects
Fluridone[2]	Adverse effects (maximum application rate)	NA	Adverse effects	NA
Glyphosate	Adverse effects (maximum application rate; typical and maximum rates using more toxic formulation)	Not addressed in ERA	Adverse effects	No effects
Hexazinone	No effects	Not addressed in ERA	Not addressed in ERA	No effects
Imazapic	No effects	No effects	No effects	No effects
Imazapyr	No effects	Not addressed in ERA	Adverse effects	No effects
Metsulfuron methyl	No effects	Not addressed in ERA	Adverse effects (maximum application rate)	No effects
Overdrive®	No effects	No effects	No effects	No effects
Picloram	Adverse effects	Not addressed in ERA	Adverse effects	No effects
Sulfometuron methyl	No effects	No effects	No effects	No effects
Tebuthiuron	No effects	No effects	Adverse effects	No effects
Triclopyr acid	No effects[3]	Not addressed in ERA	Adverse effects	No effects
Triclopyr BEE	Adverse effects	Not addressed in ERA	Adverse effects	Adverse effects (maximum application rate)

[1] Since the BLM ERAs did not assess the risks associated with spills into a stream, results for spills into a pond are presented here.
[2] Diquat and fluridone are used to control aquatic weeds; direct application into a pond or stream is a typical use. Off-site drift and surface runoff scenarios do not apply, since these herbicides would not be applied in upland areas.
[3] For this herbicide, "direct spray" also considers a normal aquatic application directly into the water column.
Note: "Adverse effects" means ERAs predicted risks at both typical and maximum application rates, unless otherwise indicated.
NA = Not applicable.

Direct Spray

Of the herbicides proposed for use, the following herbicides would potentially result in adverse health effects to fish if sprayed directly into aquatic habitats: bromacil, diquat, diuron, fluridone, glyphosate, picloram, and triclopyr BEE (Tables 5-2 and 5-3). Furthermore, the following herbicides would potentially result in adverse health effects to aquatic invertebrates if sprayed directly into aquatic habitats: bromacil, diquat, diuron, fluridone, glyphosate (the more toxic formulation), imazapic, tebuthiuron, and triclopyr BEE (Tables 5-4 and 5-5).

Since diquat, fluridone, 2,4-D, glyphosate, imazapyr, and triclopyr are all either strictly aquatic herbicides or are approved for use in aquatic habitats, direct spray into an aquatic habitat would be a normal treatment application for these herbicides. In all other scenarios (including upland scenarios with 2,4-D, glyphosate, imazapyr, or triclopyr), adverse health effects to aquatic TEP species predicted by ERAs would result from accidental spray of terrestrial herbicides into bodies of water.

Accidental Spill

Risk assessments predicted risks to fish and aquatic invertebrates as a result of an accidental spill of herbicide formulations, both terrestrial and aquatic, into a water body. As shown in Tables 5-2 and 5-3, such a spill of 2,4-D, bromacil, clopyralid, diquat, diuron, fluridone, glyphosate, imazapyr, metsulfuron methyl, picloram, tebuthiuron, or triclopyr could potentially result in adverse effects to fish. Adverse effects to fish were assumed for a spill of hexazinone as well. An accidental spill of one or more of these herbicides, with the exception of metsulfuron methyl, could also potentially result in adverse effects to aquatic invertebrates (Tables 5-4 and 5-5).

Off-site Drift

Of the terrestrial herbicides proposed for use addressed in BLM ERAs, only diuron would potentially result in adverse health effects to aquatic TEP species as a result of off-site drift into nearby aquatic habitats. Based on ERAs, stream- and pond-dwelling fish within 100 feet of a diuron application at the maximum application rate would be at risk. In addition, stream-dwelling aquatic invertebrates within 100 feet of a diuron application at the typical application rate, and pond-dwelling aquatic invertebrates within 900 feet of a diuron application at the maximum application rate would be at risk. Risk assessments prepared by the Forest Service did not consider an off-site drift scenario. Risks to fish and aquatic invertebrates from drift of these herbicides, with the exception of triclopyr BEE, seem unlikely, given the results of surface runoff scenarios. To be conservative, however, it is assumed that adverse effects to fish could potentially occur as a result of drift of glyphosate, picloram, and triclopyr BEE; and adverse effects to aquatic invertebrates could potentially occur as a result of drift of the more toxic formulation of glyphosate, or triclopyr BEE.

Surface Runoff

Herbicides used in vegetation treatments could indirectly affect aquatic TEP species if surface runoff from a contaminated upland area entered a water body. Of the terrestrial herbicides proposed for use, bromacil, diuron, tebuthiuron, and triclopyr BEE could result in adverse health effects to aquatic species under certain scenarios of surface runoff (Tables 5-2 through 5-5). Of these herbicides, diuron would likely pose the greatest risks to aquatic organisms via this exposure pathway, potentially resulting in adverse health effects to pond-dwelling fish and aquatic invertebrates in areas where precipitation is greater than 5 inches per year, and adverse effects to stream-dwelling fish and aquatic invertebrates in areas where precipitation is greater than 10 inches per year. Adverse health effects to stream-dwelling aquatic invertebrates could also occur where precipitation is greater than 5 inches per year if the maximum application rates were used. Runoff of bromacil would potentially result in adverse health effects to pond-dwelling fish under a variety of conditions. Runoff of tebuthiuron would potentially result in adverse health effects to pond-dwelling aquatic invertebrates, primarily in scenarios where the herbicide was applied to a nearby upland site at the maximum application rate. Runoff of triclopyr BEE would potentially result in adverse health effects to fish under certain site conditions.

TABLE 5-4
Summary of Effects to TEP Aquatic Invertebrates in Ponds

Herbicide	Direct Spray	Off-site Drift	Spill	Surface Runoff
2,4-D	No effects	Not addressed in ERA	Adverse effects	No effects
Bromacil	No effects	No effects	Adverse effects	No effects
Chlorsulfuron	No effects	No effects	No effects	No effects
Clopyralid	No effects	Not addressed in ERA	Adverse effects	No effects
Dicamba	No effects	No effects	No effects	No effects
Diflufenzopyr	No effects	No effects	No effects	No effects
Diquat[1]	Adverse effects	NA	Adverse effects	NA
Diuron	Adverse effects	Adverse effects (maximum application rates)	Adverse effects	Adverse effects
Fluridone[1]	Adverse effects (maximum application rate)	NA	Adverse effects	NA
Glyphosate	Adverse effects (more toxic formulation)	Not addressed in ERA	Adverse effects	No effects
Hexazinone	No effects	Not addressed in ERA	Not addressed in ERA	No effects
Imazapic	No effects	No effects	No effects	No effects
Imazapyr	No effects	Not addressed in ERA	Adverse effects (maximum application rate)	No effects
Metsulfuron methyl	No effects	Not addressed in ERA	No effects	No effects
Overdrive®	No effects	No effects	No effects	No effects
Picloram	No effects	Not addressed in ERA	Adverse effects	No effects
Sulfometuron methyl	No effects	No effects	No effects	No effects
Tebuthiuron	Adverse effects	No effects	Adverse effects (helicopter spill only)	Adverse effects (predominantly at maximum application rate)
Triclopyr acid	No effects[2]	Not addressed in ERA	Adverse effects	No effects
Triclopyr BEE	Adverse effects	Not addressed in ERA	Adverse effects	No effects

[1] Diquat and fluridone are used to control aquatic weeds; direct application into a pond or stream is a typical use. Off-site drift and surface runoff scenarios do not apply, since these herbicides would not be applied in upland areas.
[2] For this herbicide, "direct spray" also considers a normal aquatic application directly into the water column.
Note: "Adverse effects" means ERAs predicted risks at both typical and maximum application rates, unless otherwise indicated.
NA = Not applicable.

TABLE 5-5
Summary of Effects to TEP Aquatic Invertebrates in Streams

Herbicide	Direct Spray	Off-site Drift	Spill[1]	Surface Runoff
2,4-D	No effects	Not addressed in ERA	Adverse effects	No effects
Bromacil	Adverse effects (maximum application rate)	No effects	Adverse effects	No effects
Chlorsulfuron	No effects	No effects	No effects	No effects
Clopyralid	No effects	Not addressed in ERA	Adverse effects	No effects
Dicamba	No effects	No effects	No effects	No effects
Diflufenzopyr	No effects	No effects	No effects	No effects
Diquat[2]	Adverse effects	NA	Adverse effects	NA
Diuron	Adverse effects	Adverse effects	Adverse effects	Adverse effects
Fluridone[2]	Adverse effects	NA	Adverse effects	NA
Glyphosate	Adverse effects (more toxic formulation)	Not addressed in ERA	Adverse effects	No effects
Hexazinone	No effects	Not addressed in ERA	Not addressed in ERA	No effects
Imazapic	Adverse effects (maximum application rate)	No effects	No effects	No effects
Imazapyr	No effects	Not addressed in ERA	Adverse effects (maximum application rate)	No effects
Metsulfuron methyl	No effects	Not addressed in ERA	No effects	No effects
Overdrive®	No effects	No effects	No effects	No effects
Picloram	No effects	Not addressed in ERA	Adverse effects	No effects
Sulfometuron methyl	No effects	No effects	No effects	No effects
Tebuthiuron	Adverse effects	No effects	Adverse effects (helicopter spill only); airplane applications not evaluated.	No effects
Triclopyr acid	No effects[3]	Not addressed in ERA	Adverse effects	No effects
Triclopyr BEE	Adverse effects	Not addressed in ERA	Adverse effects	No effects

[1] Since the BLM ERAs did not assess the risks associated with spills into a stream, results for spills into a pond are presented here.
[2] Diquat and fluridone are used to control aquatic weeds; direct application into a pond or stream is a typical use. Off-site drift and surface runoff scenarios do not apply, since these herbicides would not be applied in upland areas.
[3] For this herbicide, "direct spray" also considers a normal aquatic application directly into the water column.
Note: "Adverse effects" means ERAs predicted risks at both typical and maximum application rates, unless otherwise indicated.
NA = Not applicable.

Indirect Effects

Herbicides that target aquatic and riparian vegetation may indirectly affect aquatic TEP species by removing plants in or adjacent to aquatic habitats. The potential short- and long-term consequences of removing target vegetation from these habitats has been discussed previously (under Effects Common to All Treatment Methods). However, herbicide applications often affect non-target vegetation in these habitats as well, some of which may provide necessary habitat components for aquatic TEP species, such as cover and food. Mortality of plants that provide key habitat for aquatic species would be expected to have short-term effects on TEP species, such as salmon, which feed on aquatic plants and rely on overhanging vegetation for cover. Chapter 4 of this BA provides more specific information on which herbicides would potentially affect non-target terrestrial and aquatic plants, either through direct or indirect means. Effects to aquatic species would typically last only until the next growing season, but would be expected to last longer if large riparian plants were lost as a result of herbicide spraying. In some cases, fish and invertebrates would be able to readily move to an area where appropriate habitat components were present.

Some TEP fish species could also be indirectly affected by herbicides through a potential reduction in prey items, primarily aquatic invertebrates and smaller fish. If herbicides were to cause a substantial reduction in food availability, TEP fish populations could decline. Such a scenario is unlikely, since buffers required to protect TEP fish species would also protect prey items in the habitat. Furthermore, any adverse effects that did occur would be temporary in nature.

Cumulative Effects

Private and tribal actions occurring on or near public lands could affect fish and other aquatic organisms discussed in this BA. Public activities, including recreation, OHV use, and fishing could impact listed species and species proposed for listing. Direct effects include removal of fish and other aquatic organisms by fishermen or other recreationists, and the mortality or injury to fish by OHVs, pack horses and mules, or recreationists and other public land users. TEP aquatic organisms could be harmed indirectly if OHV use, pack stock use, or other recreational activities impacted water flows or quality in or near the vicinity of the organisms. These adverse effects could result from spills of petroleum products from vehicles, or material found in animal feces, or other pollutants entering a nearby water bodies and impacting the organism or its habitat.

Livestock grazing on public lands could impact TEP fish and other aquatic organisms. Livestock could directly affect TEP organisms by trampling them. Indirect effects would include erosion and degradation of water quality, loss of forage and cover, and removal of water in areas of heavy livestock use that could affect aquatic organisms.

TEP fish and other aquatic organisms are at risk from private, industrial activities that occur on public lands, including mining, oil and gas and ROW development, and timber harvest activities that would potentially disturb large areas of habitat. Direct impacts would include loss of habitat and destruction or harm to TEP populations from clearing of land for construction of facilities, surface disturbance associated with timber harvest, and vegetation management at facilities. Water pollution, and introduction of noxious weeds into aquatic habitats could indirectly affect TEP fish and other aquatic organisms or their habitats. If herbicides were used to maintain vegetation along ROW or at facilities, herbicide drift could impact nearby TEP aquatic species.

Tribal actions that could harm TEP aquatic species include fishing and netting of animals for traditional lifeway uses. Indirect effects from tribal actions would be similar to those associated with recreation.

TEP aquatic species could be indirectly harmed by activities occurring on non-federal lands adjacent to public lands. For example, herbicide treatments on nearby agricultural lands or rangelands could drift onto public lands and harm TEP aquatic species. In addition, impacts to air and water quality, from the spread of weeds, or from wildfire would which result from activities that occur off public lands. Construction of dams and diversions, hydropower development, and habitat degradation associated with urbanization are activities that have adversely affected salmonids in most major rivers that these fish use to access public lands.

Conservation measures (see below) and SOPs identified in this BA and in the PEIS and PER would reduce the likelihood that TEP aquatic species would be impacted by vegetation treatments and non-federal activities on public lands. The BLM would conduct surveys for TEP species and other aquatic organisms, and an analysis of project impacts to these species would be done under NEPA as part of the permitting and siting process for land-disturbing activities conducted by private entities on public lands. The BLM would conduct local level consultation with the Services, as discussed in Chapter 3, for actions that have potential to affect TEP aquatic species. The BLM would also coordinate with tribes having an interest in TEP fish and other aquatic organisms, or potentially affecting these species, on public lands to minimize associated impacts.

Conservation Measures

Many local BLM offices already have management plans in place that ensure the protection of these species, and have completed formal or informal consultations on similar treatment activities. These consultations have identified protection zones alongside aquatic habitats that support these species. The conservation measures discussed below are probable steps required of the BLM to ensure that vegetation treatments would minimize impacts to TEP species. These conservation measures are intended as broad guidance at the programmatic level; further analysis of treatment programs and species habitats at the local level is required to better reduce potential impacts from proposed vegetation treatments. Completion of consultation at the local level will fine-tune conservation measures associated with treatment activities and ensure consistency of the treatments with ESA requirements.

The aquatic TEP species considered in this programmatic BA occur in varied habitats, over a large geographic area. The conservation measures guidance presented below is intended to apply broadly to aquatic species and habitats over the entire region covered by this BA, based on the common features found in nearly all aquatic and riparian habitats. Some species with alternate or unusual habitat requirements may require additional conservation measures to ensure a Not Likely to Adversely Affect determination at the local level. Such additional conservation measure are outside the scope of this BA, and will be completed at the local level.

Some local BLM plans have delineated protected riparian areas, or portions of watersheds where riparian-dependent resources receive primary emphasis, and management activities are subject to specific standards and guidelines (USDA Forest Service 1995). These protected riparian areas include traditional riparian corridors, wetlands, intermittent streams, and other areas that help maintain the integrity of aquatic ecosystems by 1) influencing the delivery of coarse sediment, organic matter, and woody debris to streams; 2) providing root strength for channel stability; 3) shading the stream; and 4) protecting water quality. Examples of protected riparian areas are the BLM’s Riparian Reserves of the Pacific Northwest and the Interior Columbia Basin, as described in the Aquatic Conservation Strategy (USDA Forest Service and USDI BLM 1994). The term “riparian areas,” as used in the conservation measures guidance below, refers to riparian protected areas, wherever such designations apply. However, since not all local BLM plans have made such designations, “riparian areas,” when the above-mentioned use is not applicable, generally refers to: 1) for streams, the stream channel and the extent of the 100-year floodplain; and 2) for wetlands, ponds, and lakes, and other aquatic habitats, the area extending to the edges of the riparian vegetation, provided it is no less than the minimum buffer distance for a given site established by local BLM biologists.

Conservation Measures for Site Access and Fueling/Equipment Maintenance

For treatments occurring in **watersheds** with TEP species or designated or undesignated critical habitat (i.e., unoccupied habitat critical to species recovery):

- Where feasible, access work site only on existing roads, and limit all travel on roads when damage to the road surface will result or is occurring.
- Do not engage in ground-disturbing activities during spawning and incubational periods.
- **Within riparian areas**, do not use vehicle equipment off of established roads.
- **Outside of riparian areas**, allow driving off of established roads only on slopes of 20% or less.

- Except in emergencies, land helicopters outside of riparian areas.
- **Within riparian areas**, do not fuel/refuel equipment, store fuel, or perform equipment maintenance (locate all fueling and fuel storage areas, as well as service landings outside of protected riparian areas).
- Prior to helicopter fueling operations prepare a transportation, storage, and emergency spill plan and obtain the appropriate approvals; for other heavy equipment fueling operations use a slip-tank not greater than 250 gallons; Prepare spill containment and cleanup provisions for maintenance operations.
- Do not conduct biomass removal (harvest) activities that will alter the timing, magnitude, duration, and spatial distribution of peak, high, and low flows.

Conservation Measures Related to Revegetation Treatments

- **Outside riparian areas**, avoid hydro-mulching within buffer zones established at the local level. This precaution will limit adding sediments and nutrients and increasing water turbidity.
- **Within riparian areas**, engage in consultation at the local level to ensure that revegetation activities incorporate knowledge of site-specific conditions and project design.

Conservation Measures Related to Prescribed Fire

Within riparian areas, in watersheds with TEP species or their habitats:

- Conduct prescribed burning only when long-term maintenance of the riparian area is the primary objective, and where low intensity fires can be maintained.
- Do not construct black lines, except by non-mechanized methods.
- Utilize/create only the following firelines: natural barriers; hand-built lines parallel to the stream channel and outside of buffer zones established at the local level; or hand built lines perpendicular to the stream channel with waterbars and the same distance requirement.
- Do not ignite fires using aerial methods.
- In forested riparian areas, keep fires to low severity levels to ensure that excessive vegetation removal does not occur.
- Do not camp, unless allowed by local consultation.
- Have a fisheries biologist determine whether pumping activity can occur in streams with TEP species.
- During water drafting/pumping, maintain a continuous surface flow of the stream that does not alter original wetted stream width.
- Do not alter dams or channels in order to pump in streams occupied by TEP species.
- Do not allow helicopter dipping from waters occupied by TEP species, except in lakes outside of the spawning period.
- Consult with a local fisheries biologist prior to helicopter dipping in order to avoid entrainment and harassment of TEP species.

Conservation Measures Related to Mechanical Treatments

Note: these measures apply only to treatments occurring in watersheds that support TEP species or in unoccupied habitat critical to species recovery (including but not limited to critical habitat, as designated by USFWS).

Outside riparian areas in watersheds with TEP species or designated or undesignated critical habitat (i.e., unoccupied habitat critical to species recovery):

- Conduct soil-disturbing treatments only on slopes of 20% or less, where feasible.
- Do not conduct log hauling activities on non-paved roads prone to erosion, where feasible.

Within riparian areas in these watersheds, more protective measures will be required to avoid adversely affecting TEP species or their habitat:

- Do not use vehicles or heavy equipment, except when crossing at established crossings.
- Do not remove large woody debris or snags during mechanical treatment activities.
- Do not conduct ground disturbing activities (e.g., disking, drilling, chaining, and plowing).
- Ensure that all mowing follows guidance to avoid adverse effects to streambanks and riparian vegetation and major effects to streamside shade.
- Do not use equipment in perennial channels or in intermittent channels with water, except at crossings that already exist.
- Leave suitable quantities (to be determined at the local level) of excess vegetation and slash on site.
- Do not apply fertilizers or seed mixtures that contain chemicals by aerial methods.
- Do not apply fertilizer within 25 feet of streams and supersaturated soils; apply fertilizer following labeling instructions.
- Do not completely remove trees and shrubs.

Conservation Measures Related to Biological Control Treatments using Livestock

For treatments occurring in watersheds that support TEP species or in critical habitat:

- Where terrain permits, locate stock handling facilities, camp facilities, and improvements at least 300 feet from lakes, streams, and springs.
- Educate stock handlers about at-risk fish species and how to minimize adverse effects to the species and their associated habitat.
- Locate watering troughs as far as is practical from the edges of riparian area, and equip each trough with a float valve.

Within riparian areas of these watersheds, more protective measures are required.

- Do not conduct weed treatments involving domestic animals, except where it is determined that these treatments will provide long-term benefits to riparian and adjacent aquatic habitats.
- Do not locate troughs, storage tanks, or guzzlers near streams with TEP species, where feasible.

Conservation Measures Related to Herbicide Treatments

The complexity of this action within riparian areas requires local consultation, which will be based on herbicide risk assessments.

Possible Conservation Measures:

- Maintain equipment used for transportation, storage, or application of chemicals in a leak proof condition.
- Do not store or mix herbicides, or conduct post-application cleaning within riparian areas.
- Ensure that trained personnel monitor weather conditions at spray times during application.
- Strictly enforce all herbicide labels.
- Do not broadcast spray within 100 feet of open water when wind velocity exceeds 5 mph.
- Do not broadcast spray when wind velocity exceeds 10 mph.
- Do not spray if precipitation is occurring or is imminent (within 24 hours).
- Do not spray if air turbulence is sufficient to affect the normal spray pattern.
- Do not broadcast spray herbicides in riparian areas that provide habitat for TEP aquatic species. Appropriate buffer distances should be determined at the local level to ensure that overhanging vegetation that provides habitat for TEP species is not removed from the site. Buffer distances provided as conservation measures in the assessment of effects to plants (Chapter 4 of this BA) and fish and aquatic invertebrates should be consulted as guidance (Table 5-6). (Note: the Forest Service did not determine appropriate buffer distances for TEP fish and aquatic invertebrates when evaluating herbicides in Forest Service ERAs; buffer distances were only determined for non-TEP species.)

- Do not use diquat, fluridone, terrestrial formulation of glyphosate, or triclopyr BEE, to treat aquatic vegetation in habitats where aquatic TEP species occur or may potentially occur.
- Follow all instructions and SOPs to avoid spill and direct spray scenarios into aquatic habitats. Special care should be followed when transporting and applying 2,4-D, bromacil, clopyralid, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, tebuthiuron, and triclopyr.
- Do not broadcast spray diuron, glyphosate, picloram, or triclopyr BEE in upland habitats adjacent to aquatic habitats that support (or may potentially support) aquatic TEP species under conditions that would likely result in off-site drift.
- In watersheds that support TEP species or their habitat, do not apply bromacil, diuron, tebuthiuron, or triclopyr BEE in upland habitats within ½ mile upslope of aquatic habitats that support aquatic TEP species under conditions that would likely result in surface runoff.

TABLE 5-6
Buffer Distances to Minimize Risks to TEP Fish and Aquatic Organisms from Off-site Drift to BLM-evaluated Herbicides from Broadcast and Aerial Treatments

Application Scenario	BROM[1]	CHLR	DICA	DIFLU	DIQT	DIUR	FLUR	IMAZ	OVER	SULF	TEBU
Minimum Buffer Distance (feet) from Fish and Aquatic Invertebrates											
Typical Application Rate											
Aerial	NA	0	NA	NA	NA	NA	NA	0	NA	0	NA
Low boom	0	0	0	0	NA	0	NA	0	0	0	0
High boom	0	0	0	0	NA	100	NA	0	0	0	0
Maximum Application Rate											
Aerial	NA	0	NA	NA	NA	NA	NA	0	NA	0	NA
Low boom	0	0	0	0	NA	100	NA	0	0	0	0
High boom	0	0	0	0	NA	900	NA	0	0	0	0

[1] BROM = Bromacil; CHLR = Chlorsulfuron; DICA = Dicamba; DIFLU = Diflufenzopyr; DIQT = Diquat; DIUR - Diuron; FLUR = Fluridone; IMAZ = Imazapic; OVER = Overdrive®; SULFM = Sulfometuron methyl; and TEBU = Tebuthiuron.
Boom height = The Tier I ground application model allows selection of a low (20 inches) or a high (50 inches) boom height.
NA = Not applicable.

Numerous conservation measures were developed from information provided in ERAs. The measures listed below would apply to TEP fish and other aquatic species at the programmatic level in all 17 western states. However, local BLM field offices could use interactive spreadsheets and other information contained in the ERAs to develop more site-specific conservation measures and management plans based on local conditions (soil type, rainfall, vegetation type, and herbicide treatment method). It is possible that conservation measures would be less restrictive than those listed below if local site conditions were evaluated using the ERAs when developing project-level conservation measures.

Local BLM offices should design conservation measures for treatment plans using the above conservation measures as guidance, but altering it as needed based on local conditions and the habitat needs of the particular TEP aquatic species that could be affected by the treatments. Locally-focused conservation measures would be necessary to reduce or avoid potential impacts such that a Not Likely to Adversely Affect determination would be reached during the local-level NEPA process. BLM offices that are responsible for the protection of Northwest salmonids are directed to the guidance document: *Criteria for At-Risk Salmonids: National Fire Plan Activities*, Version 2.1 (National Fire Plan Technical Team 2002), which contains detailed instructions for developing suitable conservation measures for these TEP species in conjunction with vegetation treatment programs, and from which many of the above-listed conservation measures were taken.

Determination of Effects

Given the assumption that any of the proposed vegetation treatments could occur anywhere on public lands, including riparian areas adjacent to aquatic habitats that support TEP species, the proposed treatment program **is likely to adversely affect** the aquatic species or their critical habitats discussed in this chapter, primarily through indirect consequences of the action. However, with the development of treatment programs that are consistent with locally and regionally agreed upon design criteria (similar to those that were developed through the National Fire Plan), most treatment effects could be reduced to a **Not Likely to Adversely Affect determination**. The previous section, Conservation Measures, lists the minimum steps required to ensure the protection of aquatic species and their critical habitats.

CHAPTER 6

TERRESTRIAL ANIMALS

This BA chapter considers a total of 67 terrestrial animal species that are listed as threatened or endangered, or that are proposed for listing. Background information is presented for each species by taxonomic grouping beginning with mollusks and ending with mammals. Within each grouping, species are further grouped, as appropriate, on the basis of habitat needs. Groupings, and species within these groupings, are ordered roughly by ecoregion.

Most of the information contained in this section was obtained directly from Federal Register documents, species recovery plans, biological assessments and evaluations, and other sources of information. Where primary reference(s) was/were used for species background and listing information, full citations are listed in the individual sections for each species. In some instances, citations were used from the primary reference(s), and the complete citations were not available from the primary reference(s) for inclusion in the Bibliography (Chapter 8). In the instances where complete citations were not available, information is listed in the individual sections on where there complete citations can be found (e.g., USFWS Sacramento Field Office, Sacramento, California). If information is not listed on the location of complete citations from the primary reference(s), then the complete citation can be found in the Bibliography.

Terrestrial Mollusks

Only one listed species of terrestrial mollusk occurs or could potentially occur within the project area: the Morro shoulderband snail. This species occurs in the Mediterranean Ecoregion division, in the same habitat as several listed plant species discussed elsewhere in this document.

Morro Shoulderband Snail

The primary reference for this section is:
USFWS. 1998s. Recovery Plan for the Morro Shoulderband Snail and Four Plants from Western San Luis Obispo County, California. USFWS. Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Morro shoulderband snail (*Helminthoglypta walkeriana*), also known as the banded dune snail, is a land snail that is endemic to the western portion of San Luis Obispo County, California. The species is found in coastal dune and scrub communities, on the south end of Morro Bay, where it is restricted to sandy soils (Holland 1986). Throughout most of the species' range, the dominant shrub associated with the snail's habitat is mock heather. Other prominent shrub and succulent species are buckwheat, giant woolly-star, chamisso bush lupine, dudleya, and in more inland locations, California sagebrush and black sage (Roth 1985). The Morro shoulderband snail has also been found under mats of non-native fig-marigold (also known as iceplant).

Away from the immediate coast, immature scrub in earlier successional stages may offer more favorable shelter sites than mature senescent stands of coastal dune scrub. The immature shrubs provide canopy cover for the snail, whereas the lower limbs of larger older shrubs may be too far off the ground to offer good shelter (Roth 1985). In addition, mature stands produce twiggy litter low in food value.

No studies or documented observations exist on the feeding behaviors of the Morro shoulderband snail, although it has been suggested that the species feeds on fungal material growing on decaying plant litter (Hill 1974).

The Morro shoulderband snail was federally listed as endangered on December 15, 1994. On February 7, 2001, the USFWS designated approximately 2,566 acres in San Luis Obispo County, California, as critical habitat for the species. Known threats to the Morro shoulderband snail include habitat destruction and degradation as a result of development, invasion by non-native plants (e.g., veldt grass), structural changes in the vegetation caused by plant senescence, and recreational use (e.g., OHV activity). Additional threats may include the small and isolated nature of the remaining populations, competition with the brown garden snail, pesticides (e.g., slug and snail baits), and the introduction of non-native predatory snails.

Effects of Vegetation Treatments on the Morro Shoulderband Snail

Effects Common to All Treatment Methods

Indirect Effects. Since the Morro shoulderband snail occurs in native coastal dune scrub communities, and its habitat is degraded by the invasion of non-native plant species, any vegetation treatment that successfully reduces the cover of non-native species in existing or potential snail habitats would be expected to benefit the species. In addition, removal of vegetation in senescent communities that are no longer suitable for the shoulderband snail could create suitable habitat for the species by returning coastal dune scrub communities to an earlier successional stage. Removal of fuels would also reduce the likelihood of a future uncontrolled wildfire, which would be capable of destroying a large portion of the snail's habitat.

Indiscriminate removal of vegetation, however, could adversely affect the Morro shoulderband snail, as the species requires some amount of plant cover for shelter and other biological needs.

Prescribed Fire Treatments

Direct Effects. Prescribed fire could result in mortality or injury to Morro shoulderbrand snails if animals were directly exposed to a burn. Snails are slow-moving animals that would be unlikely to escape from the path of a fire.

Indirect Effects. Coastal dune scrub communities evolved with fire, and are therefore adapted to this type of disturbance. Therefore, a prescribed burn that mimics the type of fire experienced by coastal dune scrub communities in the past would be expected to benefit the Morro shoulderband snail's habitat. Fire could create early successional habitat for the snail and aid in controlling non-native species. However, given the small size of remaining snail populations, a burn through existing habitat (compared to a burn that affected only a small portion of this habitat, or a nearby habitat where the snail does not occur) could adversely affect the species more than would be expected if the populations were secure and habitat was unfragmented.

Fire would also be expected to burn up the decaying litter that provides food for the shoulderband snail. This reduction in food would be temporary, and the severity of effects would be dependent on the presence of alternate food sources in close proximity of the treatment site.

Mechanical Treatment Methods

Direct Effects. Use of heavy equipment in habitat occupied by the shoulderband snail could crush snails. However, loose sandy soils may give, allowing some animals to be pushed safely into the soil.

Indirect Effects. Mechanical treatments could alter the structure of shoulderband snail habitat, making it less suitable for snails. Removal of large tracts of vegetation would be expected to increase the susceptibility of snails to predation, as their cover would be removed. Whether removal of cover would constitute an adverse effect would depend on whether alternative sources of cover were available in close proximity to the treatment site.

Manual Treatment Methods

Direct Effects. Few direct effects would be expected to result from manual treatment methods, although field crews could potentially crush some animals.

Indirect Effects. Hand removal of weeds and excess fuels should have few negative effects on the habitat of the Morro shoulderband snail. Utilization of manual treatment methods would allow the selective removal of the non-native species that threaten the snail, while avoiding the structural changes and loss of cover associated with larger-scale vegetation removal. Overall effects to the snail would likely be beneficial.

Biological Control Treatments

Domestic Animals
Direct Effects. There could be some direct effects associated with domestic animals crushing snails. However, these effects would likely be minor.

Indirect Effects. Temporary containment of domestic animals to control weeds in coastal dune scrub habitat could affect shoulderband snail habitat through the ingestion and trampling of vegetation, which could alter the structure of the area, making habitat less suitable for the snail. The primary effect would be loss of protective cover. However, as long as excessive removal of vegetation was not allowed, effects would likely be temporary and minimal, provided the grazing practices did not encourage the spread of non-native species.

Other Biological Control Agents
Direct Effects. Few direct effects would be expected from the release of biological control agents in shoulderband snail habitat. The presence of workers in snail habitat to release the agents or monitor their effects could cause some crushing of snails, but these effects would likely be minimal.

Indirect Effects. There is typically a small risk of unanticipated impacts to ecosystems associated with the use of biological control agents, despite the fact that agents are pre-tested under laboratory conditions. However, adverse effects to the shoulderband snail are not reasonably foreseeable as a result of using these agents.

Herbicides
Direct Effects. During herbicide treatments in areas inhabited by the Morro shoulderband snail, use of trucks/ATVs to apply herbicides, as well as walking or riding a horse through the area, could crush and injure or kill snails. However, some trampled snails would likely be pushed into the sandy soils rather than crushed.

If an herbicide application were to occur in or near shoulderbrand snail habitat, direct spray of snails could occur during the treatment. According to ERAs, direct spray of shoulderbrand snails by 2,4-D, bromacil, diquat, diuron, glyphosate, hexazinone, tebuthiuron, or triclopyr at the typical application rate, or clopyralid, imazapyr, or picloram at the maximum application rate, would potentially result in adverse health effects to snails. It is expected that adverse health effects would include mortality, reduced reproductive output, behavioral modification, and/or increased susceptibility to environmental stresses. Because the remaining populations of the Morro shoulderbrand snail are small and isolated, these toxicological effects could lead to a further decrease in the size and viability of the affected population, and possibly lead to extirpation of the population. Table 6-1 provides additional information on the application rates for which risks were predicted, as well as the relative level of risk.

Risk assessments also analyzed the risks to terrestrial invertebrates through dermal contact with vegetation after an herbicide treatment. This type of exposure scenario would entail much lower exposure levels than the direct spray scenario described in the previous paragraph, but would be a more likely exposure pathway. Based on the results of ERA, as summarized in Table 6-1, adverse health effects could potentially occur if Morro shoulderband snails were to come in contact with vegetation sprayed by 2,4-D, diquat, diuron, glyphosate, hexazinone, tebuthiuron, or triclopyr. As shown in Table 6-1, 2,4-D is the only herbicide that would potentially cause adverse effects to

TABLE 6-1
Summary of Effects to Terrestrial Invertebrates

Herbicide	Direct Spray	Level of Risk[1]	Dermal Contact with Sprayed Vegetation	Level of Risk[1]
2,4-D	Adverse effects	Typical rate: M Maximum rate terrestrial: M Maximum rate aquatic: H	Adverse effects	Typical rate: L Maximum rate terrestrial: L Maximum rate aquatic: M
Bromacil	Adverse effects	Typical rate: L Maximum rate: L	No effects	--
Chlorsulfuron	No effects	--	No effects	--
Clopyralid	Adverse effects	Typical rate: N/A Maximum rate: L	No effects	--
Dicamba	No effects	--	No effects	--
Diflufenzopyr	No effects	--	No effects	--
Diquat	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L
Diuron	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L
Fluridone	No effects	--	No effects	--
Glyphosate	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L
Hexazinone	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L
Imazapic	No effects	--	No effects	--
Imazapyr	Adverse effects	Typical rate: N/A Maximum rate: L	No effects	--
Metsulfuron methyl	No effects	--	No effects	--
Overdrive®	No effects	--	No effects	--
Picloram	Adverse effects	Typical rate: N/A Maximum rate: L	No effects	--
Sulfometuron methyl	No effects	--	No effects	--
Tebuthiuron	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L
Triclopyr acid	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L
Triclopyr BEE	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L

[1] L = low risk; M = medium risk; H = high risk; and N/A = ERAs did not predict risk at this application rate.
Note: Diquat and fluridone are aquatic herbicides that would not be used by the BLM in terrestrial applications. For 2,4-D, the maximum terrestrial application rate, rather than the maximum aquatic application rate, is the maximum rate that would be used in terrestrial applications.

invertebrates via this exposure pathway when sprayed at the typical application rate. Therefore, even manual spot treatments of this herbicide would have the potential to affect snails in the vicinity.

Indirect Effects. As discussed under Effects Common to All Treatment Methods, removal of non-native plant species would likely benefit the Morro shoulderbrand snail by increasing the quality of habitat. However, since the species relies on vegetation for food and cover, use of herbicides in habitat could adversely affect the snail by reducing the cover of native vegetation. Although vegetation losses would be short term in nature, snail populations could decline. Use of herbicides to treat vegetation in habitats that are not currently suitable for snails, especially those near to existing snail habitat, could benefit the species by increasing the amount of habitat and potentially allowing populations to expand in size.

Conservation Measures

The following conservation measures are the minimum steps required of the BLM to ensure that treatment methods would be unlikely to adversely affect the Morro shoulderband snail.

- Survey treatment sites within the range of the Morro shoulderband snail for the presence of the snail, prior to formulating treatment programs (should be conducted by a qualified biologist).
- Do not burn, conduct mechanical treatments, or use broad-spectrum herbicides in habitats occupied by snails.
- Do not perform herbicide treatments in habitats occupied by snails that will result in a substantial reduction of plant (and especially native plant) cover; where feasible, spot treat vegetation rather than spraying.
- Do not apply 2,4-D in Morro shoulderbrand snail habitat; do not broadcast spray 2,4-D within ¼ mile of Morro shoulderbrand snail habitat.
- When conducting herbicide treatments in or near Morro shoulderbrand snail habitat, avoid use of the following herbicides, where feasible: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, picloram, tebuthiuron, and triclopyr.
- Do not broadcast spray bromacil, diquat, diuron, glyphosate, hexazinone, tebuthiuron, or triclopyr in habitats occupied by Morro shoulderbrand snails; do not broadcast spray these herbicides in areas adjacent to Morro shoulderbrand snail habitat under conditions when spray drift onto the habitat is likely.
- If spraying clopyralid, imazapyr or picloram in habitats occupied by Morro shoulderbrand snails, use the typical, rather than the maximum, application rate.
- If conducting manual spot applications of diquat, diuron, glyphosate, hexazinone, tebuthiuron, or triclopyr to vegetation in Morro shoulderbrand snail habitat, utilize the typical, rather than the maximum, application rate.

Determination of Effects

Assuming that any of the proposed vegetation treatments could occur anywhere on public lands, these treatments, if applied directly to known shoulderband snail habitat, are **likely to adversely affect** the species. However, a small number of precautions taken at the local level during treatments would ensure that impacts to species were avoided. These include the general conservation measures listed in the previous section, as well as any additional measures deemed necessary by local BLM offices. Implementation of suitable conservation measures at the local level should result in a **not likely to adversely affect** determination.

Arthropods – Butterflies and Moths

There are a total of seven TEP butterfly and moth species occurring within the project area. These species occur in a number of different ecoregions throughout the west (apart from the subtropical ecoregions, or hot climates), but have similar general habitat requirements: open conditions and the presence of larval host plants and nectar sources.

Carson wandering skipper – Temperate Desert
Pawnee montane skipper – Temperate Steppe
Uncompahgre fritillary – Temperate Steppe
Quino checkerspot – Mediterranean
Kern primrose sphinx moth – Mediterranean
Oregon silverspot – Mediterranean/Marine
Fender's blue – Marine

Carson Wandering Skipper

The primary reference for this section is:
USFWS. 2002h. Determination of Endangered Status for the Carson Wandering Skipper. Federal Register 67(152): 51116-51129.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS, Nevada Fish and Wildlife Office, Reno, Nevada.

The Carson wandering skipper (*Pseudocopaeodes eunus obscurus*) is locally distributed in grassland habitats on alkaline substrates in eastern California and western Nevada. The subspecies is currently known from only two populations: one in Washoe County, Nevada, and one in Lassen County, California. Little is known about the specific habitat requirements of the Carson wandering skipper, beyond the similarities recognized among known locations of this subspecies. Based on these similarities, suitable habitat for the Carson wandering skipper has the following characteristics: elevation of less than 5,000 feet; location east of the Sierra Nevada; presence of saltgrass (the larval host plant); open areas near springs or water; and geothermal activity.

Based on observations, suitable larval habitat appears to be related to the presence of microtopographic variation, where areas in which saltgrass stands are above standing water allow for larval development. Since the few historic collections of the Carson wandering skipper have been near hot springs, it is possible that this subspecies may require the higher water table or ground temperature associated with these areas to provide the appropriate temperatures for successful larval development (Brussard et al. 1999).

Because adult Carson wandering skippers require nectar for food, suitable habitat areas must have an appropriate nectar source that is in bloom during the flight season. Plant species known to be used by the Carson wandering skipper for nectar include a mustard (crisped thelypody), racemose golden-weed, and slender birds-foot trefoil (Brussard et al. 1999). If alkaline-tolerant plant species are not present, but there is a fresh water source to support alkaline-intolerant nectar sources adjacent to the larval host plant, the area may provide suitable habitat.

Carson wandering skipper females lay their eggs on saltgrass (Hickman 1993), the larval host plant for the subspecies (Garth and Tilden 1986, Scott 1986). Saltgrass is a common plant species in the saltbush-greasewood community in the intermountain west. The plant usually occurs where the water table is high enough to keep its roots saturated for most of the year (West 1988 *cited in* Brussard et al. 1998). No other observations have been made of the early life stages of the Carson wandering skipper. However, the subspecies' life cycle is probably similar to other species in the grass skipper subfamily. Larvae live in silked-leaf nests, and some species make their nests partially underground. Pupae generally rest in the nest, and larvae generally hibernate (Scott 1986). Carson wandering skippers are thought to produce one brood per year during June to mid-July (Austin and Emmel 1998).

The Carson wandering skipper was federally listed as endangered on August 7, 2002. Critical habitat was found to be "not determinable" at the time of listing, and hence has not been designated. Because of the small, isolated nature of the known populations of this subspecies, extinction could occur from naturally occurring events or other threats. These threats include habitat destruction, degradation, and fragmentation resulting from urban and residential development; wetland habitat modification; agricultural practices (e.g., excessive livestock grazing); gas and geothermal development; and the invasion of non-native species. Other threats include collecting, livestock trampling, water exploration projects, road construction, recreation, and pesticide drift.

Pawnee Montane Skipper Butterfly

The primary reference for this section is:
USFWS. 1998t. Pawnee Montane Skipper Butterfly (*Hesperis leonardus montana*) Recovery Plan. Denver, Colorado.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Pawnee montane skipper butterfly (*Hesperis leonardus montana*) is a small, brownish-yellow butterfly that occurs only on the Pikes Peak Granite Formation in the South Platte River drainage system in Colorado. Its range, which is estimated at 23 miles long and 5 miles wide, includes portions of Jefferson, Douglas, Teller, and Park counties. The total known habitat within this range is estimated to be 37.9 square miles. The area occupied by the skipper is owned and/or administered by the Forest Service (Pike National Forest), Jefferson County, Colorado State Land Board, the BLM, the Denver Water Department, and private individuals.

Pawnee montane skippers occur in dry, open Ponderosa pine woodlands at an elevational range of 6,000 to 7,500 feet. The slopes are moderately steep, with soils derived from Pikes Peak granite. The understory is limited in the pine woodlands. Blue grama grass, the larval food plant, and the prairie gayfeather, the primary nectar plant, are two necessary components of the ground cover strata. Small clumps of blue grama occur throughout the warm, open slopes inhabited by skippers, and prairie gayfeather occurs throughout the ponderosa pine woodlands. Skippers are uncommon in pine woodlands that have a tall shrub understory (Keenan et al. 1986), or where young conifers dominate the understory (ERT Company 1986).

The vegetative community preferred by the skipper is a northernmost extension of the Ponderosa pine/blue grama grass habitat type documented from southern California and Northern New Mexico. However, prairie gayfeather does not occur in similar habitats to the south. The northeastern limit of the Ponderosa pine/blue grama grass community overlapping with the southwestern limit of the prairie gayfeather may contribute to the maintenance of the species in this limited area.

Pawnee montane skippers emerge from their pupae as adult butterflies in late July, which is apparently the same time that the prairie gayfeather flowers. Adults spend most of their short existence feeding and mating. Adult females deposit eggs singly and directly on the leaves of blue grama grass (Scott and Stanford 1982, McGuire 1982, Opler 1986). The species overwinters as young larvae, and little is known about the larval and pupal stages. Pupation is generally short (12 to 23 days) in most butterfly species. The skipper completes its life cycle (egg to larva to pupa to adult butterfly to egg) annually (Keenan et al. 1986). Adult skippers probably fly until a major killing frost occurs (ERT Company 1986).

The prairie gayfeather apparently requires openings from single event disturbances, such as logging or fire, but does not tolerate continuous disturbance. However, the skipper apparently does not colonize fire-created areas for at least several years after disturbance and regeneration. Besides the prairie gayfeather, other plants that have been used as nectar sources include musk thistle (which is classified as a noxious weed by Jefferson County), smooth blue aster, Canada thistle, beebalm, pineywoods geranium, sunflower, and broomlike ragwort.

The Pawnee montane skipper was federally listed as threatened on September 25, 1987. Critical habitat has not been designated. Encroachment of conifers and subsequent loss of grasses and prairie gayfeather reduces the quality and quantity of skipper habitat. In addition, there has been increased use of habitat by OHVs. Another current impact is the pine beetle control program on lodgepole pines, which entails road construction, stockpiled logs, and vehicles parked in meadow habitats. Because of the limited habitat and range of the Pawnee montane skipper, unexpected random events could have major deleterious effects on the population. Invasion by noxious weeds that may outcompete blue grama and prairie gayfeather, such as knapweed, is also a serious threat to the skipper.

Uncompahgre Fritillary Butterfly

The primary reference for this section is:
USFWS. 1994h. Uncompahgre Fritillary Butterfly Recovery Plan. Denver, Colorado.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Uncompahgre fritillary butterfly (*Boloria acrocnema*) has the smallest total range of any North American butterfly species. Its habitat is limited to two verified areas (inhabited by three colonies), and possibly an additional two small colonies in the San Juan Mountains and southern Sawatch Range in Gunnison, Hinsdale, and Chaffee counties in southwestern Colorado. All colonies known to the USFWS are associated with patches of snow willow, which provides larval food and cover, and are located above 12,500 feet. The species has been found only on northeast-facing slopes, which are the coolest and wettest microhabitat available in the San Juan Mountains (Scott 1982, Brussard and Britten 1989). Adults nectar on a range of flowering alpine plants (Seidl 1993).

The females usually lay their eggs on snow willow plants, or in litter within snow willow patches. It is believed that the species has a biennial life history, requiring 2 years to complete its life cycle (Scott 1982, Brussard and Britten 1989). Eggs laid in even years are caterpillars during the following odd year, and then mature into adults during the following even year. Although odd- and even-year broods may function as essentially separate populations, evidence of gene flow between the two (Brussard and Britten 1989) suggests that at times, larvae hatched early in the summer can develop into adults the following year.

The Uncompahgre fritillary was federally listed as endangered on June 24, 1991. Critical habitat has not been designated. Overcollection is considered the greatest human-caused threat to the species. Its sedentary nature, weak flying ability, and tendency to fly low to the ground make it easy to collect. Other actual or potential effects to the species include adverse climatic changes, small population size, and low genetic variability. There is also a minor potential threat from the trampling of larvae by livestock and humans.

Quino Checkerspot Butterfly

The primary reference for this section is:
USFWS. 2002i. Designation of Critical Habitat for the Quino Checkerspot Butterfly (*Euphydryas editha quino*). Federal Register 67(72): 18355-18395.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Carlsbad Fish and Wildlife Office, Carlsbad, California.

The Quino checkerspot butterfly (*Euphydryas editha quino*) is a subspecies of Edith's checkerspot that is locally distributed in sunny openings within chaparral and coastal sage shrublands in the interior foothills of Riverside and San Diego counties in California, and in adjacent Mexico. Like other subspecies of Edith's checkerspot, the Quino checkerspot shows a habitat preference for low-growing vegetation interspersed with barren spots (Osborne and Redak 2000). The thermodynamic requirements of the butterfly and its natural avoidance of shaded areas deter flight below the canopy of vegetation (Singer 2001). Male Quino checkerspot butterflies and, to a lesser extent, females, are frequently observed on hilltops and ridgelines (Osborne 2001).

The distribution of the Quino checkerspot is highly dependent on the availability of its primary larval host plant, dwarf plantain. Typically, butterflies occur where there are high densities of this plant, although other species of host plant are also used. Above the elevational limits of dwarf plantain (approximately 9,750 feet), woolly plantain and white snapdragon appear to be the primary host plants utilized by the butterfly (Pratt 2001). All host plant species occur in coastal sage scrub, open chaparral, grassland, and similar open-canopy plant communities. Dwarf plantain is often associated with soils with fine-textured clay or with cryptogamic crusts.

Edith's checkerspot butterflies use a much wider range of plant species for adult nectar feeding than for larval foliage feeding. The butterflies frequently take nectar from lomatium, goldenstar, yarrow, fiddleneck, goldfields, popcornflowers, gilia, California buckwheat, onion, and yerba santa (Murphy and Pratt 2000). Chia may also be used for nectar feeding (Orsak 1978, Osborne 2001), but is probably not preferred (Pratt and Murphy 2001). Quino checkerspot butterflies have been observed flying several hundred feet from the nearest larval habitat patch to nectar sources.

The life cycle of the Quino checkerspot butterfly includes four distinct life stages: egg, larva (caterpillar), pupa (chrysalis), and adult, with the larval stage divided into five to seven instars (periods between molts, or shedding skin). There is typically one generation of adults per year, with a 4- to 6-week flight period beginning between late February and May, depending on weather conditions (Emmel and Emmel 1973).

Quino checkerspot butterflies deposit eggs on plants located in full sun, preferably surrounded by bare ground or sparse, low-growing vegetation (Weiss et al. 1987, 1988; Osborne and Redak 2000). Eggs deposited by adults on host plants hatch in 10 to 14 days. Primary host plants must remain edible for approximately 8 weeks to support pre-diapause larvae if no secondary host plants are available (Singer 1972, Singer and Ehrlich 1979). Quino checkerspot butterfly larvae may undergo as many as seven molts prior to pupation. Newly hatched larvae spin a web and feed in clusters on the plant where their eggs were deposited. If larvae have accumulated sufficient energy reserves, they enter diapause as host plants age and become dry and inedible, and usually remain in diapause until December or January. Although the exact location of diapausing Quino checkerspot butterfly larvae is not known, clusters of post-diapause larvae found near dense grass and shrub cover indicate that they may diapause in these areas (Osborne and Redak 2000). Sufficient rainfall, usually during November or December, stimulates germination and growth of host plants, and apparently causes larvae to break diapause. Post-diapause larvae undergo from two to as many as four instars prior to pupating in webbed shelters near ground level. Adults emerge from pupae after approximately 10 days, depending on the weather (Mattoni et al. 1997).

Distributions of patches of Quino checkerspot habitat are defined by a matrix of adult resources (all larval resources are found within areas of adult movement), primarily nectar plants, oviposition plants, and basking sites. Habitat patch fragmentation occurs when land use changes compromise adult movement patterns and frequently results from habitat destruction that reduces resource availability. Such fragmentation may substantially reduce the ability of habitat patches to support local populations. Most Quino checkerspot butterfly populations are part of a larger metapopulation structure. Isolated habitat patches are not sufficient to ensure the long-term persistence of butterfly metapopulations (Hanski 1999). A local habitat patch population may be expected to persist on the time scale of years (Harrison 1989); however, persistence of metapopulations for longer terms results from the interaction among sets of local habitat patch populations at larger geographic scales. Maintenance of landscape connectivity (habitat patches linked by intervening dispersal areas) is essential in order to maintain metapopulation resilience. Land use changes that block dispersal between habitat patches and isolate local populations by compromising landscape connectivity can be just as detrimental to metapopulation survival as those that destroy or reduce the size of habitat patches.

The Quino checkerspot was listed as endangered on January 16, 1997. On April 15, 2002, approximately 171,605 acres of land in Riverside and San Diego counties were designated as critical habitat for the subspecies. The Quino checkerspot butterfly is threatened primarily by urban and agricultural development, non-native plant species invasion, OHV use, grazing, and fire management practices. These threats destroy and degrade the quality of habitat and result in the extirpation of local Quino checkerspot populations. Quino checkerspot butterfly population decline likely has been, and will continue to be, caused in part by enhanced nitrogen deposition, elevated atmospheric carbon dioxide concentrations, and climate change. Nonetheless, urban development poses the greatest threat and exacerbates all other threats. Activities resulting in habitat fragmentation or host or nectar plant removal reduce habitat quality and increase the probability of local Quino checkerspot butterfly population extirpation and species extinction. Other threats to the species include illegal trash dumping and predation.

Kern Primrose Sphinx Moth

The Kern primrose sphinx moth (*Euproserpinus euterpe*) is restricted to the northwest portion of the Walker Basin in southern Kern County, California, east of Bakersfield and south of Sequoia National Forest. The Basin is located at an elevation of approximately 4,820 feet, and is surrounded by mountains over 6,560 feet in elevation (USFWS 1983d). Currently, a large portion of the basin is devoted to agriculture (primarily barley cultivation and cattle pasture). The dominant vegetation in the sandy washes in which the colony occurs includes filaree, baby blue-eyes, and rabbitbrush, as well as California goldfields and Australian brome. The soil originates from decomposed granite and is largely alluvial in nature. Its texture is coarse to fine sand with very little silt.

The annual evening-primrose, on which the larvae of Kern primrose sphinx moths feed, occurs in dry, disturbed and sandy-gravelly areas below 9,850 feet in elevation in many plant communities, from Oregon to Baja California (USFWS 1983d). In the Walker Basin, the evening-primrose is frequently found along the edge of sandy washes adjoining fallow fields. Seeds begin to germinate in February and March, but the young seedlings are frequently difficult to locate and identify during the flight season of the moth. The plant community surrounding the basin floor is dominated by California juniper, blue oak, shrub live oak, interior live oak, rabbitbrush, sagebrush, and singleleaf pine. The distribution of the moth may be limited because the host plant does not occur in these plant communities. South of the Basin, the plant community is oak-grassland and appears unsuitable for the moth. Adult Kern primrose sphinx moths utilize nectar from filaree and baby blue-eyes.

The flight season lasts from the last week of February to the first week of April, peaking during the second or third week of March (Tuskes and Emmel 1981). The sphinx moth is a day flier, with adults flying during the warmer parts of the day, usually between 10:00 a.m. and 2:30 p.m. In the morning, males and females frequently bask on bare patches of soil, dirt roads, or rodent mounds. As the afternoon winds increase, adult basking locations change to areas protected from wind, such as washes, behind knolls, or on the ground among bushes.

The breeding period is coincident with the adult flight season: from the last week of February to the first week of April. Correct oviposition is on evening-primrose plants that occur in sandy-gravelly areas near washes in the Basin (Tuskes and Emmel 1981). However, female moths consistently deposit eggs on the filaree, a naturalized exotic plant. Larvae hatched from eggs deposited on filaree do not feed, and subsequently die of starvation within a few days. Such ovipositional errors may be an important factor in reproductive success and subsequently contribute to the scarcity of the moth (USFWS 1983d). At least 11 days are required for the eggs to hatch, and there are five larval instars before pupation occurs in May. Pupation occurs in the soil, and a pupation chamber is constructed near the surface, perhaps under rocks or other objects. The adults may emerge the following year, or may remain in the pupal stage for an undetermined number of years.

The Kern primrose sphinx moth was federally listed as threatened on April 8, 1980. Critical habitat has not been designated. Human activity probably has affected the population levels of the Kern primrose sphinx moth in at least three ways: 1) the introduction and establishment of non-native plants, particularly filaree, may have had a substantial impact on the ability of the moth to locate and oviposit on the correct host plant; 2) land use practices probably have directly influenced the survival of the moth and/or its host plant; and 3) flight characteristics of the moth result in higher mortality of females than males by collectors, which adversely affects the population's reproductive potential. Evening-primrose occurs in sandy soil along washes and in fallow fields in somewhat ruderal habitats. Much of the land in the Walker Basin that was appropriate habitat for the moth has been developed for agricultural purposes, and is used as cropland or pasture for cattle. Overexploitation of the species by collectors is also a concern.

Oregon Silverspot

The primary reference for this section is:
USFWS. 2001h. Oregon Silverspot Butterfly (*Speyeria zerene hippolyta*) Revised Recovery Plan. Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Oregon silverspot butterfly (*Speyeria zerene hippolyta*) occurs at disjunct sites near the Pacific coast, from Del Norte County, California, north to Long Beach Peninsula, Washington. The subspecies occupies three types of grassland habitat: marine terrace and coastal "salt spray" meadows, stabilized dunes, and montane grasslands. The first two habitats are strongly influenced by proximity to the ocean, with mild temperatures, high rainfall, and persistent fog. Of the two, the dune habitat tends to have lower relief, highly porous soils, and less exposure to winds. Conditions at the montane sites include colder temperatures, frequent cloud cover, substantial snow accumulations, less coastal fog, and no salt spray.

Oregon silverspot butterfly populations currently (as of 2001) are known to occur at only six sites. One is in Del Norte County (Lake Earl), two are in Lane County (Rock Creek-Big Creek and Bray Point), and two are in Tillamook County (Cascade Head and Mount Hebo). The population at a sixth site in Clatsop County (Clatsop Plains) has declined in recent surveys, with only one Oregon silverspot butterfly documented in 1998 (VanBuskirk 1993, 1998).

Each type of habitat must provide the Oregon silverspot with host plants, nectar sources, and other suitable environmental conditions. Caterpillars feed primarily on early blue violets. Stands of violets that are large enough to provide enough food for larval butterflies on the Oregon coast occur only in relatively open and low-growing grasslands, where violets may be an abundant component of the plant community (Hammond and McCorkle 1984). Apart from early blue violets, Oregon silverspot caterpillars are also known to feed on a few other violet species, such as yellow stream violets and Aleutian violets. Nectar plants most frequently used by Oregon silverspot butterflies are members of the aster family, including the following native species: Canada goldenrod, dune goldenrod, California aster, pearly everlasting, dune thistle, and yarrow. They are also known to nectar on two common introduced species: tansy ragwort and false dandelion. The flowering seasons of these species overlap, providing an array of nectar choices for adult butterflies throughout the flight season.

The Oregon silverspot butterfly goes through six larval instars and a pupal stage before metamorphosing into an adult. Newly hatched first-instar larvae immediately enter diapause after eating the lining of the eggshell. They remain in diapause until host plants send up new growth in spring, and feed until pupation in the summer. Very little is known about the biology of the caterpillar or pupae. Adult emergence starts in July and extends into September, with many males appearing several weeks before females appear. Mating usually takes place in relatively sheltered areas. Adults will often move long distances for nectar or to escape windy and foggy conditions.

The Oregon silverspot butterfly was federally listed as threatened on July 2, 1980, and critical habitat was designated at the same time. Lands included in the critical habitat designation are those that were known to be occupied by the butterfly at the time: portions of Section 15 and the south half of Section 10 that are west of a line parallel to and about 1,500 feet west of the eastern section boundaries of Sections 10 and 15, T16S, R12W, Lane County, Oregon. Invasion by exotic species, natural succession, fire suppression, and land development have resulted in the loss and modification of the species' habitat. Land use practices have altered disturbance regimes needed to maintain existing habitats and create new habitats for species expansion. Other threats to the subspecies include OHVs, grazing, erosion, road kill, and pesticides. The Oregon silverspot butterfly is also sought by collectors.

Fender's Blue Butterfly

The primary reference for this section is:
USFWS. 2000i. Endangered Status for *Erigeron decumbens* var. *decumbens* (Willamette Daisy) and *Icaricia icarioides fenderi* (Fender's Blue Butterfly) and Threatened Status for *Lupinus sulphureus* ssp. *kincaidii* (Kincaid's Lupine). Federal Register 65(16): 3875-3890.

References cited in this section are internal to the above referenced document. A complete list of these references is available from the USFWS Oregon State Office, Portland, Oregon.

The Fender's blue butterfly (*Icaricia icarioides fenderi*) is endemic to upland prairies of the Willamette Valley in Oregon. Although the precise historic distribution of this subspecies is unknown, recent surveys have indicated that the insect is confined to the Willamette Valley and currently occupies 32 sites in Yamhill, Polk, Benton, and Lane counties (Hammond and Wilson 1993, Schultz 1996). One population is found in wet, hairgrass-type prairie, while the remaining sites are found on drier upland prairies characterized by fescue. Fender's blue butterflies occupy sites located almost exclusively on the western side of the valley, within 21 miles of the Willamette River.

The primary habitat requirement for the fender's blue is its host plant, Kincaid's lupine, which is the larval food source. Of the 32 sites where Fender's blue butterfly occurs, Kincaid's lupine co-occurs as a larval host plant at 27. Spurred lupine and sickle keeled lupine may be secondary food plants used by the insect (Hammond and Wilson 1993).

It is thought that the life cycle of Fender's blue is similar those of related subspecies (Hammond and Wilson 1993, Mattoni 1997, Pratt 1997). Adult butterflies lay their eggs on the host plant, which serves as a food source for the caterpillars during May and June. Newly hatched larvae feed for a short time, reaching their second developmental stage in the early summer, at which point they enter an extended diapause (maintaining a state of suspended activity). Diapausing larvae remain in the leaf litter at or near the base of the host plant through the fall and winter, and may become active again in March or April of the following year. Some larvae may be able to extend diapause for more than one season depending upon the individual and environmental conditions (Mattoni 1997). Once diapause is broken, the larvae feed and grow through three to four additional developmental stages, enter their pupal stage, and then emerge as adult butterflies in April and May. A Fender's blue butterfly may complete its life cycle in 1 year.

The Fender's blue was federally listed as endangered on January 25, 2000. The designation of critical habitat for this species was deemed prudent, but has been deferred. The primary threats are habitat loss from agriculture and urban development, the invasion of non-native plant species into prairie habitat, and the small size of the remaining populations. Herbicide use and collecting are also factors that can impact this subspecies.

Effects of Vegetation Treatments on Butterflies and Moths

Effects Common to All Treatment Methods

Indirect Effects

The TEP butterfly and moth species that occur or potentially occur within the project area are found in open areas, which are typically either formed or maintained by some sort of a disturbance. Therefore, fire suppression activities have likely resulted in a reduction in available habitat for these species. All treatment activities that increase the amount of open habitat on the site would be expected to have long-term positive effects on these species, provided that the needed larval food plants and nectar plants are present on the site. Creation of open areas adjacent to known locations of TEP species would also have the potential to increase the size and range of existing populations in some instances.

Butterflies and moths are very susceptible to mortality caused by fire, as they are sedentary during the bulk of their life cycle. An unmonitored, uncontrolled wildfire could easily burn through an entire population of a rare butterfly or moth species, and many species are at risk of elimination from such an occurrence. The fuels reduction activities proposed by the BLM would likely provide a long-term positive benefit for these species by reducing the likelihood of a future devastating wildfire.

The removal of non-native plant species from habitats in which these species occur, or in nearby habitats, would also be expected to have positive effects. Non-native species can exclude larval food plants and nectar sources, which butterflies and moths are dependent upon for survival and the completion of their life cycle. Non-native species can also change the habitat structurally so that adults are unable to forage adequately.

Vegetation treatments could also have indirect negative effects on butterfly and moth populations by causing mortality to larval host plants or nectar plants. Without an adequate population of these habitat elements, the TEP species would be unable to persist.

Prescribed Fire Treatments

Direct Effects

A prescribed fire could negatively affect TEP butterflies and moths by killing adults, larvae, and eggs. Adults would be able to fly and some would likely escape a burn. However, the early life stages, which often inhabit the duff layer or soil surface, would be especially susceptible to direct mortality from fire (Nagel 1973, Martin and Mitchell 1981). Given the reduced numbers of many of these TEP species, even a small, well-controlled fire would be capable of eliminating an entire population.

Indirect Effects

A prescribed fire in habitat that is suitable for TEP species could negatively affect larval food plants and nectar sources, which could in turn severely affect butterfly and moth populations. Under most circumstances, these effects would be short-term in nature, but given the small, isolated habitats that remain, they could create conditions from which the TEP species are unable to recover. Other indirect effects of fire could include changes in microclimate, and a loss of cover, resulting in greater exposure to weather extremes and predators (Martin and Mitchell 1981, Warren et al. 1987).

Over the long term, habitat would be expected to benefit from prescribed fire, through the creation of open conditions that prevent trees and shrubs from shading out host and nectar plants, and a potential reduction in non-native species that compete with host plants and nectar sources. Prescribed fire occurring in historic habitat areas adjacent or close to current butterfly locations could have long-term positive effects for the species by increasing the amount of available habitat and potentially the sizes of species populations.

Mechanical Treatment Methods

Direct Effects

Like fire, mechanical methods are often used to control vegetation over a large area. Thus, they can have severe direct effects on small, isolated butterfly and moth populations. Equipment associated with mechanical control can crush or otherwise harm adults, larvae, and eggs throughout much of the year.

Indirect Effects

Removal of host plants and nectar plants on which the butterfly and moth species rely can have indirect effects on populations. Cutting aboveground portions of nectar plants during the growing season could cause some important species to fail to flower during a butterfly's flight season, reducing the availability of food (Pickering 1997). Over the long term, however, removal of invading woody vegetation and non-native plant species would be expected to have a positive effect on habitat. Techniques such as mowing would be especially likely to have a positive effect if carried out in areas adjacent to known butterfly or moth habitat. Over the long term, such habitat rehabilitation could increase the area of suitable habitat for these TEP species, potentially increasing the size of existing populations.

Manual Treatment Methods

Direct Effects

There would likely be some direct effects to butterflies and moths from trampling by field crews performing manual control. Even people that are trying to avoid butterflies or moths can easily step on larvae or damage eggs, which can be difficult to see.

Indirect Effects

Manual treatment methods are typically precise treatments that target certain undesirable species. Field crews would be able to avoid most damage to host plants or nectar plants. Therefore, the potential short-term effects to butterflies and moths would be much less severe than those caused by prescribed fire, biological control, mechanical control, or herbicides.

Biological Control Treatments

Domestic Animals

Direct Effects. Introduction of domestic animals into butterfly or moth habitat to contain weeds would directly affect TEP species populations, should large herbivores trample larvae and eggs. The extent of these effects would depend on the timing and intensity of the treatment, and the amount of area covered.

Indirect Effects. During weed containment, domestic animals might graze on or cause damage to host and nectar plants, indirectly affecting butterflies and moths by reducing the availability of food.

Long-term effects of moderate levels of grazing would likely be positive, as domestic animals can control the invasion of open areas by trees and shrubs. Containment of weeds adjacent to occupied habitat could have long-term positive effects by increasing the suitability of habitat for future inhabitance by TEP butterfly and moth species.

Other Biological Control Agents

Direct and Indirect Effects. There could be some trampling of larvae, eggs, and adults by workers releasing biological control agents into butterfly and moth habitats. This disturbance would be minimal, and of short duration. Over the long term, there is the potential for unforeseen impacts to butterflies and moths resulting from the release of biological control agents. The likelihood of such an occurrence is very slim and not anticipated.

Herbicides

Direct Effects

During herbicide treatments in areas where listed butterflies and moths occur, trucks and/or ATVs used to apply herbicides could crush larvae, eggs, and adults. Horses, or workers on foot with backback sprayers, could also trample butterflies and moths in the treatment area, resulting in injury or mortality.

Inadvertent exposure of TEP butterflies and moths to herbicides would be likely if treatments were to occur in areas where these species occur. Reasonable exposure pathways include direct spray (particularly during sedentary phases of the life cycle) and dermal contact with vegetation that has been treated with herbicides. According to ERAs, direct spray of butterflies and/or moths by 2,4-D, bromacil, diquat, diuron, glyphosate, hexazinone, tebuthiuron, or triclopyr at the typical application rate, or clopyralid, imazapyr, or picloram at the maximum application rate, would potentially result in adverse health effects (see Table 6-1). In addition, contact with vegetation treated by diquat, diuron, glyphosate, hexazinone, tebuthiuron, or triclopyr at the maximum application rate, or by 2,4-D at the typical application rate, could result in adverse health effects to TEP butterflies and moths. Adverse health effects could include mortality, reduced reproductive output, behavioral modification, and/or increased susceptibility to environmental stresses. These toxicological effects could lead to a further decrease in the size and viability of affected populations. Small, fragmented populations could potentially be extirpated or become more susceptible to future extirpation by environmental stresses and other factors. Table 6-1 provides additional information on the application rates for which risks to terrestrial invertebrates were predicted, as well as the relative level of risk for each herbicide.

Indirect Effects

Listed butterfly and moth species could suffer indirect effects from herbicide treatments if non-target host and nectar plants were sprayed by herbicides. Indirect effects to non-target plant species are predicted as a result of direct spray by all herbicides approved for use by the BLM. In addition, non-target plants could be impacted by off-site drift and surface runoff of several herbicides approved for use by the BLM (see Tables 4-2 through 4-4 for more information). Localized elimination or a reduction in numbers of host and/or nectar plants could result in adverse population-level effects to the listed butterfly or moth species that rely on these plants.

Conservation Measures

Many local BLM offices already have management plans in place that ensure the protection of these species during activities on public lands. The following conservation measures are the minimum steps required of the BLM to ensure that treatment methods would be unlikely to adversely affect TEP species.

Each local BLM office is required to draw up management plans related to treatment activities that identify any TEP butterfly or moth species or their critical habitat that are present in the proposed treatment areas, as well as the measures that will be taken to protect these species.

Management plans should, at a minimum, follow this general guidance:

- Use an integrated pest management approach when designing programs for managing pest outbreaks.
- Survey treatment areas for TEP butterflies/moths and their host/nectar plants (suitable habitat) at the appropriate times of year.
- Minimize the disturbance area with a pre-treatment survey to determine the best access routes. Areas with butterfly/moth host plants and/or nectar plants should be avoided.
- Minimize mechanical treatments and OHV activities on sites that support host and/or nectar plants.
- Carry out vegetation removal in small areas, creating openings of 5 acres or less in size.
- Avoid burning all of a species' habitat in any 1 year. Limit area burned in butterfly/moth habitat in such a manner that the unburned units are of sufficient size to provide a refuge for the population until the burned unit is suitable for recolonization. Burn only a small portion of the habitat at any one time, and stagger timing so that there is a minimum 2-year recovery period before an adjacent parcel is burned.
- Where feasible, mow or wet around patches of larval host plants within the burn unit to reduce impacts to larvae.
- In TEP butterfly/moth habitat, burn while butterflies and/or moths of concern are in the larval stage, when the organisms would receive some thermal protection.
- Wash equipment before it is brought into the treatment area.
- Use a seed mix that contains host and/or nectar plant seeds for road/site reclamation.

- To protect host and nectar plants from herbicide treatments, follow recommended buffer zones and other conservation measures for TEP plants species when conducting herbicide treatments in areas where populations of host and nectar plants occur.
- Do not broadcast spray herbicides in habitats occupied by TEP butterflies or moths; do not broadcast spray herbicides in areas adjacent to TEP butterfly/moth habitat under conditions when spray drift onto the habitat is likely.
- Do not use 2,4-D in TEP butterfly/moth habitat.
- When conducting herbicide treatments in or near habitat used by TEP butterflies or moths, avoid use of the following herbicides, where feasible: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, picloram, tebuthiuron, and triclopyr.
- If conducting manual spot applications of diquat, diuron, glyphosate, hexazinone, tebuthiuron, or triclopyr to vegetation in TEP butterfly or moth habitat, utilize the typical, rather than the maximum, application rate.

Determination of Effects

Under the assumption that any of the proposed vegetation treatments could occur anywhere on public lands, these treatments would be **likely to adversely affect** any and all of the TEP butterfly and moth species discussed in this section. However, if the proper precautions were taken at the local level during the formulation of treatment programs, impacts to these species could be avoided. The previous section, Conservation Measures, provides general guidance for the protection of these species. Local offices would be required to fine-tune and expand conservation measures, as necessary, in order to reduce effects to a **not likely to adversely affect** determination.

Arthropods – Terrestrial Insects

Only two listed terrestrial insects occur or could potentially occur within the project area: The valley elderberry longhorn beetle and the American burying beetle. The valley elderberry longhorn beetle occurs in the Mediterranean Ecoregion Division, whereas the American burying beetle is found in numerous ecoregions, all of which are in the eastern portion of the proposed project area. Because these two species occupy different habitats and have very different life history requirements, a separate effects analysis has been completed for each species.

Valley Elderberry Longhorn Beetle

The primary reference for this section is:
USFWS. 1984g. Valley Elderberry Longhorn Beetle Recovery Plan. Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The valley elderberry longhorn beetle (*Desmocerus californicus dimorphus*) is endemic to remnants of moist valley oak woodlands associated with riparian systems in the lower Sacramento and upper San Joaquin valleys of California, where its foodplant, elderberry, grows (Linsley and Chemsak 1972, Arnold 1983). These riparian forest remnants are difficult to characterize because they occur in many different forms throughout the valley. Under ideal conditions, they consisted of several canopy layers with a dense undergrowth (Katihah 1983). Fremont cottonwood, California sycamore, willow, and valley oak were common overstory species. The intermediate canopies consisted of California boxelder, Oregon ash, elder, and various species of willow. Vines were abundant in all canopy layers of the riparian forest. Undergrowth vegetation was quite diverse, and today includes a number of exotic weeds. As a result of urban and agricultural development within the beetle's range, elderberry today grows in a number of unnatural areas (e.g., urban parks, power-line corridors, agricultural land) that formerly were riverine floodplains, but which now represent lands reclaimed by man.

Eggs are laid in May on elderberry stems greater than 1 inch in diameter, on healthy plants. Larvae bore into stems and feed on the soft core of the plant, remaining in larval form inside excavated passages within the stem for as long as 2 years before emerging as adults. Adults feed on elderberry flowers and possibly foliage (Linsley and Chemsak 1972, Arnold 1983).

The valley elderberry longhorn beetle was federally listed as threatened on August 8, 1980. Critical habitat has been designated in the City of Sacramento, in the American River Parkway, and in Goethe Park, Sacramento County, California. Although the entire historical distribution of the species is unknown, the extensive destruction of riparian forests of the Central Valley of California strongly suggests that the beetle's range may have shrunk and become greatly fragmented. The primary threat to survival of the species has been, and continues to be, the loss and alteration of habitat by agricultural conversion, grazing, levee construction, stream and river channelization, removal of riparian vegetation, rip-rapping of shoreline, as well as recreational, industrial, and urban development (Arnold 1983). Insecticide and herbicide use in agricultural areas may be factors limiting the beetle's distribution. The age and quality of individual elderberry shrubs/trees and stands may also be a factor in the beetle's limited distribution.

Effects of Vegetation Treatments on the Valley Elderberry Longhorn Beetle

Effects Common to All Treatment Methods

Indirect Effects. All treatment activities that reduce the cover of non-native species in or near valley elderberry longhorn beetle habitat would be likely to have a positive effect on the species. The beetle is most abundant in dense native plant communities, and excessive weed growth has been identified as a threat to the beetle as well as elderberry, its host plant (USFWS 1984g).

Fire has also been identified as a threat to beetles and elderberries. A severe wildfire through longhorn beetle habitat could destroy host plants and other riparian vegetation. Therefore, all treatment methods that reduce fuels accumulations, thereby reducing the risk of a future catastrophic wildfire, would have a long-term positive effect on longhorn beetle habitat.

Removal of vegetation in riparian areas can negatively affect beetle habitat, depending on the extent of the removal. Apart from the removal of host plants, structural changes to riparian areas may also render them less suitable for supporting valley elderberry longhorn beetles.

Prescribed Fire Treatments

Direct Effects. As discussed above, fire has been identified as a threat to beetles and their host elderberry plants. A prescribed fire in elderberry habitat would likely kill beetles, which are small and would be unable to escape a burn.

Indirect Effects. A prescribed fire could destroy elderberry plants, on which the beetle depends for survival. A destruction of habitat crucial to the species' survival could be so devastating that populations of the beetle would be unable to recover. In addition to requiring elderberry plants for survival, the beetle is most abundant in plant communities with a mature overstory and a mixed understory. Thus, prescribed fire could alter the structure and composition of riparian habitats and reduce the suitability of habitat for the beetle. The removal of vegetation can also lead to erosion, which would further degrade the riparian habitat.

Mechanical Treatment Methods

Direct Effects. The use of mechanical treatment methods in longhorn beetle habitats could cause direct mortality to beetles and their eggs through crushing by heavy equipment.

Indirect Effects. Mechanical methods of removing vegetation could also kill or damage elderberry trees or seedlings. In addition, removal of large amounts of vegetation could alter the structure of riparian habitats, rendering them less suitable for supporting populations of longhorn beetles. Mechanical methods that remove weeds and fire hazards in nearby habitats, however, may have long-term positive effects on the species by helping to create habitats that may eventually be able to support longhorn beetles and by reducing the risk of future wildfires that could severely impact the species.

Manual Treatment Methods

Direct Effects. Manual treatment methods could have some direct effects on longhorn beetles. Field crews could crush beetles or their eggs, although the extent of these effects would likely be minimal.

Indirect Effects. Hand removal of weeds and other materials should have few negative effects on beetle habitat. This treatment method would allow workers to avoid harming elderberry trees and seedlings. Structural changes caused by vegetation removal would also likely be minor, and the long-term benefits should outweigh the short-term negative effects on habitat.

Biological Control Treatments

Domestic Animals
Direct Effects. There could be some direct effects associated with domestic animals crushing beetles or ingesting their eggs while grazing.

Indirect Effects. The most notable effects from the use of domestic animals to contain weeds would likely be the destruction of host plants and young elderberry plants that could serve as host plants in the future. In addition, domestic animals would be expected to thin the understory, altering the structure of the habitat and making it less suitable for longhorn beetles. Grazing can also lead to the degradation of riparian habitat. The severity of the effects of this treatment method would be dependent on its timing, intensity, and duration.

Other Biological Control Agents
Direct Effects. The release of biological control agents into longhorn beetle habitat would cause few direct effects to beetles. There could be some trampling/crushing associated with workers releasing the agents or doing monitoring, but these effects, should they occur, would be minimal.

Indirect Effects. There could be some unanticipated impacts associated with the use of biological control agents. Given that agents would be pre-tested under laboratory conditions and approved for use, these risks would be slim and adverse consequences are not reasonably foreseeable; however, given the host specificity of the beetle, consequences could be very severe.

Herbicides

Direct Effects. Injury or mortality of longhorn beetles could occur during herbicide applications, as a result of trucks/ATVs, horses, or people crushing beetles and eggs. In addition, longhorn beetles could be exposed to herbicides during treatments, either by being directly sprayed during the treatment or by coming into contact with treated vegetation after the spraying occurred. According to the ERAs, direct spray of beetles by 2,4-D, bromacil, diquat, diuron, glyphosate, hexazinone, tebuthiuron, or triclopyr at the typical application rate, or by clopyralid, imazapyr, or picloram at the maximum application rate, would potentially result in adverse health effects (see Table 6-1). In addition, if beetles were to come into contact with vegetation treated by diquat, diuron, glyphosate, hexazinone, tebuthiuron, or triclopyr at the maximum application rate, or by 2,4-D at the typical application rate, adverse health effects could potentially occur. Adverse health effects could include mortality, reduced reproductive output, behavioral modification, and/or increased susceptibility to environmental stresses. These toxicological effects could lead to a further decrease in the size and viability of affected populations. Small, fragmented populations could potentially be extirpated or become more susceptible to future extirpation by environmental stresses and other factors. Table 6-1 provides additional information on the application rates for which risks to terrestrial invertebrates were predicted, as well as the relative level of risk for each herbicide.

Indirect Effects. As discussed under Effects Common to All Treatment Methods, removal of non-native plant species would likely benefit the valley elderberry longhorn beetle, since it is most prevalent in native plant communities, and excessive weed growth is a known threat. However, since the species occurs in dense riparian habitats, herbicide treatments that would cause a large-scale removal of plant cover would likely have an adverse impact on the species. Such an herbicide treatment would be unlikely to occur in the riparian habitats where the beetle occurs, since the BLM would design its treatment programs to protect these habitats and adjacent streams. If host elderberry plants were to become injured or suffer mortality as a result of herbicide treatments, it is likely that populations of longhorn beetles would be adversely affected.

Conservation Measures

The following conservation measures are the minimum steps required of the BLM to ensure that treatment methods would be unlikely to adversely affect TEP species. These measures should be implemented in habitats where beetles are known to occur or are likely to occur.

- Survey proposed treatment sites within the range of the valley elderberry longhorn beetle for the presence of the beetle and its elderberry host plant (should be conducted by a qualified biologist).
- Establish a 100-foot buffer between suitable beetle habitat and mechanical treatments (except mowing of grasses/ground cover) and treatments using domestic animals. Suitable beetle habitat is defined as any area containing elderberry stems measuring 1 inch or more in diameter at ground level.
- Mow grasses/ground cover only between July and April.
- Do not mow within 5 feet of elderberry plant stems, and do not mow in a manner that damages plants.
- Protect all elderberry shrubs with evidence of beetle exit holes from prescribed fire using water or by removing fuels surrounding the plants.
- To protect host elderberry plants from herbicide treatments, follow recommended buffer zones and other conservation measures for TEP plants species, as listed on pages 4-127 through 4-131, when conducting herbicide treatments in areas where populations of elderberry occur.
- Do not broadcast spray herbicides in suitable beetle habitat; do not broadcast spray herbicides in areas adjacent to suitable beetle habitat under conditions when spray drift onto the habitat is likely.
- Do not use 2,4-D in valley elderberry longhorn beetle habitat.
- When conducting herbicide treatments in or near habitat used by TEP butterflies or moths, avoid use of the following herbicides, where feasible: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, picloram, tebuthiuron, and triclopyr.
- If conducting manual spot applications of diquat, diuron, glyphosate, hexazinone, tebuthiuron, or triclopyr to vegetation in valley elderberry longhorn beetle habitat, utilize the typical, rather than the maximum, application rate.

Summary of Effects

Assuming that any of the proposed vegetation treatments could occur anywhere on public lands, these treatments would be **likely to adversely affect** the valley elderberry longhorn beetle. However, if the proper precautions were taken at the local level during the formulation of treatment programs, impacts to these species could be avoided. The previous section, Conservation Measures, lists the minimum steps required to protect the valley elderberry longhorn beetle from vegetation treatments. Additional steps may be required at the local level to ensure that vegetation treatments would be **not likely to adversely affect** the species.

American Burying Beetle

The primary reference for this section is:
USFWS. 1991b. American Burying Beetle (*Nicrophorus americanus*) Recovery Plan. Newton Corner, Massachusetts.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The American burying beetle was historically widespread throughout the eastern U.S. and Canada, but the species has experienced a rapid decline, and now occupies less than 10% of its original distribution. At present, it occurs in a few eastern locales, and in Oklahoma, Nebraska, and South Dakota.

Little is known about the habitats associated with most historical collections of the American burying beetle. Beetles have been found in a wide range of habitat types, including riparian deciduous forests, deciduous forests, scrub forests, maritime scrub, coniferous forests, grasslands, and pasturelands. Although certain situations and soil types are not suitable for carcass burial (e.g., very xeric, saturated, or loose sandy soils), it is thought that carrion availability in any given area is more important for American burying beetles than vegetation or soil structure. However, both of these parameters do affect the occurrence and density of vertebrates, and of invertebrates that compete with the American burying beetle for limited carrion resources.

Rangewide, American burying beetles are generally active from late April through September. Adults are fully nocturnal, and are usually active only when nighttime temperatures exceed 60 °F. When not engaged in brood-rearing, adults feed on a broad range of available carrion, and may also capture and consume live insects (Scott and Traniello 1989).

Reproduction in the species depends on the availability of vertebrate carrion of an appropriate size and weight. The optimal weight of carrion selected by American burying beetles is between 3.5 and 7 ounces. Although the species can successfully produce a brood with smaller carcasses, there appears to be a positive relationship between carcass weight and fecundity (Kozol et al. 1988).

Using antennal chemoreceptors, most burying beetles are attracted to carrion at night, generally soon after dark. Upon discovery of a suitable carcass, males may broadcast pheromones to attract potential mates (Bartlett 1987, Eggert and Miller 1989). Males and females compete among themselves and with other species of burying beetle until one pair remains on the carcass. Typically, size is the prime determinant of success in claiming this resource. The victorious pair buries the carcass, usually before dawn. Eventually, a burial chamber is formed by the movements of the beetles, and the carcass is cleaned of feathers or fur and coated with anal and oral secretions, which retard decay and contamination.

Eggs are laid in an escape tunnel adjacent to the carrion, and at least one parent—usually the female—appears to be critical for survival of the young (Wilson and Fudge 1984). Adult beetles not only guard their offspring, but tend to feed them also (Fetherston et al. 1990). Larvae pupate in soil near the brood chamber, emerging as adults in about 48 to 60 days. For the most part there is one generation per year, although individuals are occasionally successful in rearing two broods of young in a single summer (Kozol 1990). Brood sizes vary from 3 to 31 individuals (Kozol 1990).

The reasons for the decline of this species are not understood. Theories include such factors as past spraying of DDT and other insecticides, the presence of a non-native and species-specific pathogen, the loss of primary forest habitat (Anderson 1982b), and other forms of habitat fragmentation. However none of these theories adequately explains why the American burying beetle declined, when similar members of the same genus are still relatively common rangewide.

The American burying beetle was federally listed as endangered in July of 1989. Critical habitat has not been designated. Potential threats to the species include activities that destroy or fragment habitat, such as development, agricultural practices and grazing, and interspecific competition. In addition, any activity that reduces the availability of carrion species can affect populations of the American burying beetle.

Effects of Vegetation Treatments on the American Burying Beetle

The project area falls mostly outside of both the historic and present range of the American burying beetle. In the western states, the species is known only from a few locales in Oklahoma, Nebraska, and South Dakota, and the only known extant population is in eastern Oklahoma. Because the American burying beetle is known from a variety of habitat types, and because the reasons for its decline and current threats are largely unknown, it is difficult to accurately assess the effects treatment methods would have on habitat or potential habitat for the species. It is believed that availability of suitable prey (primarily small mammals and birds) are more important that particular features of the habitat, beyond general soil type.

Effects Common to All Treatment Methods

Indirect Effects. Any treatment method that reduces the fuels buildup in and around known populations of the burying beetle could benefit the species by reducing the likelihood of a catastrophic wildfire that could wipe out an existing population or suitable habitat. Treatments that reduce the coverage of weeds would be expected to have fewer effects (beyond those associated with fuels reduction), although they could have some effect on the diversity of prey species present at a site.

Prescribed Fire Treatments

Direct Effects. A prescribed fire on a site known to support American burying beetles could have substantial effects by destroying insects, eggs, and larvae. Given the fragmentation of existing populations, high mortality to one of these populations would be a great loss for the species as a whole.

Indirect Effects. Prescribed fire could potentially have some benefits for the species. The reintroduction of fire to areas in which it has been suppressed would create disturbances that create forest openings and edge habitats, which could increase the diversity of prey items available to the species. If conducted in areas adjacent to known populations, these treatments could increase the success of the species.

Mechanical Treatment Methods

Direct Effects. The use of mechanical equipment to carry out vegetation treatments could have some direct effects on burying beetles. Heavy equipment could cause mortality to larvae, overwintering or inactive adults, and eggs in the soil. Carcasses and feeding larvae could also be physically disturbed by heavy equipment.

Indirect Effects. The small mammals on which burying beetles feed could be harmed or killed by mechanical treatments. For a brief period of time, the presence of suitable carcasses could increase.

Manual Treatment Methods

Direct and Indirect Effects. Using hand methods to remove vegetation from burying beetle habitat would be unlikely to affect beetles or their habitat, provided workers took measures to avoid injuring beetles, larvae, or eggs.

Biological Control Treatments

Domestic Animals
Direct Effects. Domestic animals could cause some direct mortality to beetles through trampling, although beetles are nocturnal and would therefore not be active during treatments.

Indirect Effects. Some prey species, such as the deer mouse, respond positively to grazing, while others respond negatively (USFWS 1984g). Therefore, it is difficult to say what the overall effect of using domestic animals to contain weeds would be on the beetle and its habitat. It is likely that at low to moderate levels, this form of treatment would have no overall effect on habitat.

Other Biological Control Agents
Direct and Indirect Effects. Biological control agents would be unlikely to affect beetles or their habitat. These agents would target particular weed species. There could be some trampling/crushing associated with workers releasing the agents or doing monitoring, but these effects would be minimal. Finally, there could be unanticipated impacts associated with the use of these agents. However, given that agents would be pre-tested under laboratory conditions and approved for use, these risks would be slim.

Herbicides

Direct Effects. If herbicide treatments were to occur in areas inhabited by the American burying beetle, use of trucks/ATVs to spray herbicides could crush and injure or kill beetles. Risks of crushing would be much less during treatments on foot or horseback.

Since they are completely nocturnal, it is unlikely that American burying beetles would be directly sprayed during herbicide treatments. It is more plausible that beetles would come into contact with sprayed vegetation after a treatment. If a direct spray scenario were to occur, adverse effects would be possible if 2,4-D, bromacil, diquat, diuron, glyphosate, hexazinone, tebuthiuron, or triclopyr were sprayed at the typical application rate, or if clopyralid, imazapyr, or picloram were sprayed at the maximum application rate (see Table 6-1). In the more plausible scenario of dermal exposure to treated vegetation, adverse health effects could occur if beetles contacted vegetation treated by diquat, diuron, glyphosate, hexazinone, tebuthiuron, or triclopyr at the maximum application rate, or by 2,4-D at the typical application rate. Adverse health effects could include mortality, reduced reproductive output, behavioral modification, and/or increased susceptibility to environmental stresses. Although little is known about the remaining populations of this species, it is possible that any reduction in the size and viability of a population would increase its risk of future extirpation. Table 6-1 provides additional information on the application rates for which risks to terrestrial invertebrates were predicted, as well as the relative level of risk for each herbicide.

Indirect Effects. Because the American burying beetle occurs in various habitats, the effects of herbicide treatments are not known. However, since availability of prey appears to be the most important habitat feature for the beetle, short-term removal of vegetation and alteration of plant communities to favor native species should not adversely affect beetle habitat. Indirect effects to American burying beetles could occur if herbicide treatments were to reduce the availability of prey. Sickness and mortality of birds and mammals could benefit the burying beetle by providing additional prey items. However the effects of burying, laying eggs in, and eating a carcass contaminated by herbicides are unknown. Therefore, it is assumed that adverse effects to beetle populations could occur under such a scenario.

Conservation Measures

Given the unlikelihood that the American burying beetle occurs on public lands, and therefore the unlikelihood that vegetation treatments would occur in burying beetle habitats, specific conservation measures are not proposed in this programmatic BA. At the local level, biologists should determine whether burying beetles are likely to occur in areas where treatments are scheduled to occur, and to develop conservation measures to avoid adverse effects to burying beetles at that time. Performing treatments outside of the active season would eliminate most risks for adverse effects to this species.

Summary of Effects

Given the lack of knowledge about the habitat requirements of this species, the overall effect of vegetation treatments on this species is hard to determine. However, given the nocturnal nature of the beetle, and the unlikelihood that it occurs on public lands, the proposed treatments would be **not likely to adversely affect** the American burying beetle.

Amphibians and Reptiles

Listed (and proposed) amphibian and reptile species predominantly occur in the Subtropical Desert habitats of the southwest, and in California, in the Mediterranean Ecoregion. A notable exception is the Wyoming toad, which occurs in the Temperate Steppe Ecoregion of Wyoming.

Amphibians:

Desert slender salamander – Subtropical Desert
Sonora tiger salamander – Subtropical Desert
Chiricahua leopard frog – Subtropical Desert/Subtropical Steppe
Wyoming toad – Temperate Steppe
California tiger salamander – Mediterranean
Arroyo toad – Mediterranean
California red-legged frog – Mediterranean

Reptiles:

Coachella Valley fringe-toed lizard – Subtropical Desert
Desert tortoise – Subtropical Desert
New Mexican ridge-nosed rattlesnake – Subtropical Desert
Giant garter snake – Mediterranean
Blunt-nosed leopard lizard – Mediterranean

Desert Slender Salamander

The desert slender salamander (*Batrachoseps aridus*) is found in crevices between limestone sheets and under limestone slabs and other rocks along the base of cliffs where continuous water seepage occurs (California Department of Fish and Game 2000c). The only confirmed known location for this species is Hidden Palm Canyon, a tributary of Deep Canyon, a large gorge draining desert slopes of the Santa Rosa Mountains (USDI BLM 2001b) located 10 miles south of Palm Desert in Riverside County, California. This site, which is owned by the California Department of Fish and Game, lies at the box end of a side canyon. Water seeping from the shaded north and northeast-facing walls of the box-end canyon provides moisture necessary for the survival of the salamander population. The population is estimated at fewer than 600 individuals and occupies a habitat of less than 0.5 acre. There is a second potential location for this species approximately 4.5 miles southeast of the Hidden Palm Canyon site, in Guadalupe Canyon, on land administered by the BLM (Giuliani 1981). Although no conclusive taxonomic information on specimens collected from this second site has been reported, the BLM considers this site habitat for the desert slender salamander (USDI BLM 2001b).

The most important structural component of the desert slender salamander's habitat is believed to be the limestone sheeting that covers portions of the canyon wall at both sides. The material has built up over a period of years as a result of seepage and precipitation of the solutes. By possessing a moist interior environment when other nearby retreats dry out, the sheeting may be a refuge of last resort for the salamander. Decay of plant roots and developmental patterns of the sheeting may account for the tunnels and pockets that provide retreats for salamanders. Erosion of this sheeting down to bedrock during severe tropical storms of 1976 resulted in a loss of approximately one-third of the available salamander habitat at the site. Desert slender salamanders likely feed on arthropods, although it is unknown whether arthropod populations affect the activity or limit the size of the salamander population.

This species is a terrestrial breeder, presumably laying eggs in an underground chamber, in a crevice, or under a rock. However, little is known about the breeding habits or courtship of any species of slender salamander, and the eggs of the desert slender salamander have never been observed.

The desert slender salamander was federally listed as endangered on June 4, 1973. Critical habitat has not been designated for the species. The continued existence of the species is threatened by a variety of factors, including its extremely restricted distribution. Since salamanders require moist conditions, desiccation of habitat during a prolonged drought could result in extirpation of the species. Maintenance of the habitat at Hidden Palm Canyon is dependent on seepage from groundwater originating in the watershed above the box canyon. Therefore, future groundwater pumping or diversion projects in this watershed could indirectly impact salamander habitat. The Guadalupe Canyon site, however, is more secure, since it is more remote, and the watershed is administered by the BLM and Forest Service (San Bernardino National Forest). Another potential threat is erosion caused by severe storms.

Sonora Tiger Salamander

The primary reference for this section is:
USFWS. 1997h. Sonora Tiger Salamander Recovery Plan (Draft). USFWS. Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS, Arizona Ecological Service Field Office, Phoenix, Arizona.

The Sonora tiger salamander (*Ambystoma tigrinum stebbinsi*) occurs in a limited number of wetland habitats in the San Rafael Valley of Arizona and Mexico, and the adjacent foothills of the Patagonia and Huachuca mountains (Arizona Game and Fish Department 1996). Suitable habitat, primarily in the form of cattle tanks (i.e., small earthen ponds), ponds, or impounded cienegas, is found in the Santa Cruz and San Pedro River drainages in Cochise and Santa Cruz counties. Historically, the subspecies probably inhabited springs, cienegas, and possibly backwater pools where permanent or nearly permanent water allowed survival of bronchiate adults. The historic and extant range of this species is within 19 miles of Lochiel, Arizona.

Cienegas in southern Arizona and northern Sonora, Mexico, are typically mid-elevation wetland communities often surrounded by relatively arid environments. These communities are usually associated with perennial springs and stream headwaters, have permanently or seasonally saturated highly organic soils, and have a low probability of flooding or scouring (Hendrickson and Minckley 1984). Cienegas, perennial streams, and rivers in the desert southwest are extremely rare, comprising less than 1% of the total land area of Arizona (Arizona Game and Fish Department 1993).

Sonora tiger salamanders may begin breeding as early as January, and eggs are laid until early May (USFWS 1997). Eggs are attached to aquatic vegetation, rocks, or other substrate in clumps of up to 50, and hatch within a few days. Larvae that hatch in permanent water often develop into branchiate adults and spend their entire lives in the water; all larvae that hatch in ephemeral waters metamorphose into the terrestrial form and return to aquatic habitats only to breed. Sexual maturity is reached in 5 to 6 weeks. Populations of the Sonora tiger salamander are dynamic. In particular, drought and disease periodically extirpate or greatly reduce populations.

The Sonora tiger salamander was listed as endangered on January 6, 1997, but critical habitat has not been designated. A variety of factors threaten the subspecies. Disease and predation by introduced non-native fishes and bullfrogs are probably the most serious and immediate threats, both of which have been implicated in the elimination of aquatic populations (Collins and Jones 1987, Collins 1996). Tiger salamanders also are widely used in Arizona as fishing bait, a use that poses additional threats. Other subspecies of tiger salamander introduced into habitats of the Sonora tiger salamander for bait propagation or by anglers could, through interbreeding, genetically swamp distinct populations of this subspecies. Additional threats include habitat destruction, reduced fitness resulting from low genetic diversity, and increased probability of chance extirpation, which is characteristic of small populations.

Chiricahua Leopard Frog

The primary reference for this section is:
USFWS. 2002j. Listing of the Chiricahua Leopard Frog (*Rana chiricahuensis*). Federal Register 67(114): 40790-40811.

References cited in this section are internal to the above-referenced documents. A complete list of these references is available from the USFWS Arizona Ecological Services Field Office, Phoenix, Arizona.

The Chiricahua leopard frog (*Rana chiricahuensis*) is known from cienegas, pools, livestock tanks, lakes, reservoirs, streams, and rivers at elevations of 3,281 to 8,890 feet. The species occurs in central and southeastern Arizona, west-central and southwestern New Mexico, and in Mexico (Platz and Mecham 1979, 1984; McCranie and Wilson 1987; Degenhardt et al. 1996; Sredl et al. 1997). The range of the species is divided into two parts: a southern group of populations (the majority of the species' range) located in mountains and valleys south of the Gila River in Arizona, New Mexico, and Mexico; and a group of northern montane populations in west central New Mexico and along the Mogollan Rim in central and eastern Arizona (Platz and Mecham 1979).

The Chiricahua leopard frog occurs in permanent aquatic habitats (which is required for reproduction), typically with abundant aquatic vegetation, at elevations from approximately 3,300 to 8,500 feet. The species feeds on a wide range of invertebrates. Leopard frogs nest in densely vegetated areas, with high canopy cover and dense foliage from ground level to about 13 feet. Chiricahua leopard frogs breed from spring to late summer, depending on elevation. Females produce egg masses that adhere, suspended just above the water surface, to vegetation growing in water 6 to 14-inches deep, near the shore of ponds and streams. Tadpoles occur approximately 2 to 9 months after hatching, and reach reproductive maturity about 2 to 3 years later.

The Chiricahua leopard frog was federally listed as threatened on July 15, 2002. Critical habitat has not been designated. The species is now absent from more than 75% of its known historical sites, and from numerous mountain ranges, valleys, and drainages within its former range. In areas where the Chiricahua leopard frog is still present, populations are often small, widely scattered, and occupy marginal and dynamic habitats. Known threats to the species include habitat alteration, destruction, and fragmentation; predation by non-native organisms; and disease.

Wyoming Toad

The Wyoming toad (*Bufo baxteri* [= *hemiophrys*]) is restricted to a very small range in the Laramie Basin of southeastern Wyoming. The Laramie Basin is a semi-arid, intermountain basin characterized by a predominant vegetation of short grasses and sagebrush, located at an elevation of between 7,200 and 7,500 feet. Since settlement and development of agriculture, the central lower portions of the basin have been irrigated using water diverted from the two major rivers, the Big and Little Laramie (Baxter 1952). The species occurs in floodplains and the short grass edges of ponds and lakes. The habitats once utilized by the Wyoming toad were floodplains ponds, small ponds and lakes produced by irrigation runoff, and the many small seepage lakes in the basin.

The Wyoming toad needs vegetative cover such as sedges and grasses, in a moist situation, throughout the summer to protect against the high evaporative power of the air in the relatively arid climate of the Laramie Basin. It probably utilizes any soft earth, such as pocket gopher burrows and sand dunes, to burrow to below the frost line for winter dormancy (Baxter 1952). The Wyoming toad requires warm (over 60 °F), shallow ponds or lake margins for reproduction; these ponds must remain filled during the period from late May until at least mid-August for completion of the tadpole stage. Adult toads are insectivorous and opportunistic in selection of food. It is unlikely that availability of food for either adults or larvae has ever been limiting for this toad.

During daylight hours in June and early July, adults and sub-adults are abundant and active in the sedges and grasses on the floodplain during June and early July. During late July, adults disappear, probably becoming largely nocturnal during the dry part of the summer and remaining beneath the surface of the ground during the day (Baxter 1952). In the period during which this toad was common in the Laramie Basin, adult toads emerged from winter dormancy in late May or early June, after daily air temperatures approach 80 °F. Breeding congresses developed in the warm, shallow floodplain ponds, and eggs were laid there. Tadpoles normally completed their transformation to adults by early August. Drying up of the floodplain ponds was a noticeable cause of mortality to the tadpoles (Baxter 1952).

The Wyoming toad was federally listed as endangered on January 17, 1984. Critical habitat has not been designated. The species was determined to be extinct in 1994, and captive-raised larvae and toadlets were released at Lake George, Rush Lake, and Mortenson Lake, Wyoming, in 1995. Reasons for the population's decline are not entirely clear, although aerial applications of pesticides for mosquito control, predation, and agricultural practices related to irrigated hay meadows have been implicated as possible causes. Current threats to this species are poorly known as well.

California Tiger Salamander

The primary reference for this section is:
USFWS. 2000j. Final Rule To List the Santa Barbara County Distinct Population of the California Tiger Salamander as Endangered. Federal Register 65(184): 57241-57264.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS, Ventura Fish and Wildlife Office, Ventura, California.

The Santa Barbara County population of the California tiger salamander (*Ambystoma californiense*) inhabits low elevation vernal pools and seasonal ponds and the associated grassland, oak savannah, and coastal scrub plant communities of the Santa Maria, Los Alamos, and Santa Rita Valleys in western Santa Barbara County (Shaffer et al. 1993; Sweet 1993, 1998a, 2000a). The population on the Santa Rosa Plain in Sonoma County is found in similar habitats. Although California tiger salamanders are adapted to natural vernal pools, manmade or modified ephemeral and permanent pools are now frequently used (Fisher and Shaffer 1996). California tiger salamanders prefer open grassland to areas of continuous woody vegetation.

Subadult and adult California tiger salamanders spend much of their lives in small mammal burrows found in the upland component of their habitat, particularly those of ground squirrels and pocket gophers at depths ranging from 8 inches to 3.3 feet beneath the ground surface (Loredo and Van Vuren 1996). California tiger salamanders use both occupied and unoccupied small mammal burrows, but an active population of burrowing mammals is necessary to sustain sufficient underground refugia for the species (Loredo et al. 1996). California tiger salamanders may remain active underground into summer, moving small distances within burrow systems. During aestivation (a state of dormancy or inactivity in response to hot, dry weather), California tiger salamanders eat very little (Shaffer et al. 1993). Once fall and winter rains begin, they emerge from these retreats on nights of high relative humidity and during rains to feed and to migrate to the breeding ponds (Stebbins 1985, 1989; Shaffer et al. 1993). Adults may migrate long distances between summering and breeding sites. The distance from breeding sites may depend on local topography and vegetation, the distribution of ground squirrel or other rodent burrows, and climatic conditions (Stebbins 1989, Hunt 1998). In Santa Barbara County, juvenile California tiger salamanders have been trapped more than 1,200 feet away from their birth pond (Mullen 1998), and adults have been found along roads more than a mile from breeding ponds (Sweet 1998a).

Once established in underground burrows, California tiger salamanders may move short distances within burrows or overland to other burrows, generally during wet weather. Dispersal distance is closely tied to precipitation; California tiger salamanders travel further in years with more precipitation. As with migration distances, the number of ponds used by an individual over its lifetime is dependent on landscape features. Migration to breeding ponds is concentrated during a few rainy nights early in the winter, with males migrating before females (Twitty 1941; Shaffer et al. 1993; Loredo and Van Vuren 1996; Trenham 1998; Trenham et al. 2000).

Female California tiger salamanders mate and lay their eggs singly or in small groups (Twitty 1941; Shaffer et al. 1993). The number of eggs laid by a single female ranges from approximately 400 to 1,300 per breeding season (Trenham 1998). The eggs typically are attached to vegetation near the edge of the breeding pond (Storer 1925, Twitty 1941), but in ponds with no or limited vegetation, they may be attached to objects (e.g., rocks, boards) on the bottom (Jennings and Hayes 1994). After breeding, adults leave the pond and typically return to small mammal burrows (Loredo et al. 1996), although they may continue to come out nightly for approximately the next 2 weeks to feed (Shaffer et al. 1993). Eggs hatch in 10 to 14 days. Larvae feed on algae, small crustaceans, and mosquito larvae for about 6 weeks after hatching, when they switch to larger prey (P. Anderson 1968). Larger larvae will consume smaller tadpoles of Pacific treefrogs, California red-legged frogs, western toads, and spadefoot toads, as well as many aquatic insects and other aquatic invertebrates (J. Anderson 1968, P. Anderson 1968). The larvae also will eat each other under certain conditions (Shaffer and Sweet cited in Collins 2000a).

Amphibian larvae must grow to a critical minimum body size before they can metamorphose to the terrestrial stage (Wilbur and Collins 1973). In general, the longer the duration of ponding, the larger the larvae and metamorphosed juveniles are able to grow. The larger juvenile amphibians grow, the more likely they are to survive and reproduce (Semlitsch et al. 1988, Morey 1998). In the late spring or early summer, before the ponds dry completely, metamorphosed juveniles leave the ponds and enter small mammal burrows after spending up to a few days in mud cracks or tunnels in moist soil near the water (Zeiner et al. 1988; Shaffer et al. 1993; Loredo et al. 1996). Like the adults, juveniles may emerge from these retreats to feed during nights of high relative humidity (Storer 1925; Shaffer et al. 1993) before settling in their selected aestivation sites for the dry summer months. Many of the pools in which California tiger salamanders lay eggs do not hold water long enough for successful metamorphosis, as larvae dry out and perish (P. Anderson 1968; Feaver 1971).

The Santa Barbara population segment of the California tiger salamander was federally listed as endangered on September 21, 2000. On July 22, 2002, the Sonoma County population segment was emergency listed as endangered. Other populations of the species are candidates for listing. Critical habitat has not been designated. Although California tiger salamanders still exist across most of their historic range, the habitat available to them has been greatly reduced. The breeding ponds and the associated upland habitats inhabited by salamanders have been degraded and reduced in number and area through changes in agriculture practices, urbanization, building of roads and highways, chemical applications, and overgrazing (Sweet 1993, 1998a,b; Gira et al. 1999; Santa Barbara County Planning and Development 2000). The primary threats to this species are destruction and modification of habitat, predation and competition by introduced or non-native species, habitat fragmentation, contamination of aquatic habitats, and overgrazing.

Arroyo Toad

The primary references for this section are:
USFWS. 1994i. Determination of Endangered Status for the Arroyo Southwestern Toad. Federal Register 59(241): 64859-64867;

and

USFWS. 2001i. Final Designation of Critical Habitat for the Arroyo Toad. Federal Register 66(26): 13656-13671.

References cited in this section are internal to the above-referenced documents. A complete list of these references is available from the USFWS Carlsbad Fish and Wildlife Office, Carlsbad, California.

The Arroyo toad (also commonly called the Arroyo southwestern toad; *Bufo californicus* [= *microscaphus*]) is a small toad that is restricted to rivers that have shallow, gravelly pools adjacent to sandy terraces. Historically, this species was found along the length of drainages in southern California from San Luis Obispo to San Diego County, and south into Mexico. However, urbanization and dam construction throughout the 20th century has destroyed and degraded habitat, limiting the occurrence of this species in the U.S. to small, isolated populations in the headwater areas of streams in Santa Barbara, Ventura, Los Angeles, Riverside, and San Diego counties (Sweet 1992). Populations may also occur in Orange, San Bernardino, and southern Imperial counties. Most of these populations occur on privately-owned lands, primarily within or adjacent to the Cleveland National Forest.

The Arroyo toad exhibits a breeding habitat specialization that favors shallow pools and open sand and gravel channels along low-gradient reaches of medium to large-sized streams (USFWS 1999). These streams can have either intermittent or perennial streamflow, and typically experience periodic flooding that scours vegetation and replenishes fine sediments. In at least some portions of its range, the species also breeds in smaller streams and canyons where low-gradient breeding sites are more sporadically distributed. Populations in smaller drainages are likely to be smaller and at greater risk of extirpation than those in larger streams and in larger habitat patches.

Arroyo toads also require, and spend most of their adult life in, upland habitats. Individual toads have been observed as far as 1.2 miles from the streams where they breed, but are most commonly found within 0.3 miles of those streams (Griffin et al. 1999; USFWS 1999; D. Holland, Camp Pendleton Amphibian and Reptile Survey, Fallbrook, California, unpublished data; Holland and Sisk 2000). Arroyo toads typically burrow underground during periods of inactivity, and thus tend to utilize upland habitats that have sandy, friable (readily crumbled) soils. Although the upland habitat use patterns of this species are poorly understood, activity probably is concentrated in the alluvial flats (areas created when sediments from the stream are deposited) and sandy terraces found in valley bottoms of currently active drainages (USFWS 1999; Griffin et al. 1999; Sweet 1999; Ramirez 2000; Holland and Sisk 2000).

Arroyo toads breed from late March until mid-June, in large streams with persistent water (Sweet 1989). Eggs are deposited and larvae develop in shallow pools with minimal current and little or no emergent vegetation. The substrate is sand or pea gravel overlain with flocculent silt. Larvae metamorphose in June or July, and juvenile toads remain on the bordering gravel bars until the pool no longer persists, typically 3 to 8 weeks (Sweet 1992). Sandy terraces with cottonwoods, oaks, and willows, and almost no grass and herbaceous cover at ground level provide optimal foraging habitat for juveniles and adults.

The Arroyo toad was listed as endangered on December 16, 1994. On February 7, 2001, a total of approximately 182,360 acres in Monterey, Santa Barbara, Ventura, Los Angeles, San Bernardino, Riverside, Orange, and San Diego counties, California, were designated as critical habitat. Critical habitat includes rivers or streams that support the appropriate habitat requirements for breeding activities and all life phases of the toad, and upland habitats of sufficient width and quality to provide foraging or living areas for subadult and adult arroyo toads. Threats to this species include habitat degradation by human factors (urbanization, agriculture, overgrazing, recreation, OHV use, and mining activities) and natural factors (drought, wildfires); predation by introduced fish and bullfrogs; and the small population size of the species

California Red-legged Frog

The primary references for this section are:
USFWS. 1996f. Determination of Threatened Status for the California Red-legged Frog. Federal Register 61(101): 25813-25833;

and

USFWS. 2001j. Final Determinations of Critical Habitat for the California Red-legged Frog. Federal Register 66(49): 14625-15674.

References cited in this section are internal to the above-referenced documents. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

The California red-legged frog (*Rana aurora draytonii*) is the largest native frog in the western U.S. (Wright and Wright 1949). Its historical range extended south from Marin County along the coast, and from Shasta County inland into Mexico (Jennings and Hayes 1985, Hayes and Krempels 1986). This subspecies has since undergone a 70% reduction in geographic range as a result of habitat loss and alteration, overexploitation, and the introduction of exotic predators. The most secure aggregations of California red-legged frogs are found in aquatic sites that support substantial riparian and aquatic vegetation and lack exotic predators (e.g., bullfrogs, bass, and sunfish).

California red-legged frogs use a variety of habitat types, including various aquatic, riparian, and upland habitats. They include, but are not limited to, ephemeral ponds, intermittent streams, seasonal wetlands, springs, seeps, permanent ponds, perennial creeks, man-made aquatic features, marshes, dune ponds, lagoons, riparian corridors, blackberry thickets, non-native annual grasslands, and oak savannas. Among the variety of habitats where California red-legged frogs have been found, the only common factor is association with a permanent water source. Apparently, California red-legged frogs can use virtually any aquatic system, provided a permanent water source, ideally free of non-native predators, is nearby. Permanent water sources can include, but are not limited to, ponds, perennial creeks (or permanent plunge pools within intermittent creeks), seeps, and natural and artificial springs. California red-legged frogs may complete their entire life cycle in a particular area (i.e., a pond that is suitable for all life stages) or utilize multiple habitat types. These variable life-history characteristics enable California red-legged frogs to change habitat use in response to varying conditions. During a period of abundant rainfall, the entire landscape may become suitable habitat. Conversely, habitat use may be drastically confined during periods of prolonged drought.

Breeding sites have been documented in a variety of aquatic habitats. Furthermore, breeding has been documented in these habitat types irrespective of vegetation cover. Frogs successfully breed in artificial ponds with little or no emergent vegetation (Bobzien 2000), and have been observed to successfully breed and inhabit stream reaches that are not cloaked in riparian vegetation (Bobzien et al. 2000). The importance of riparian vegetation for this subspecies is not well understood. It is believed that riparian communities offer good foraging habitats because of the moisture and camouflage that they provide. They also serve as dispersal areas, and support pools and backwater aquatic areas for breeding. However, other factors are more likely to influence the suitability of aquatic breeding sites, such as the general lack of introduced aquatic predators.

California red-legged frogs generally breed from November through March, and lay their eggs during or shortly after large rainfall events in late winter and early spring (Hayes and Miyamoto 1984). Females attach their eggs in masses to vertical emergent aquatic vegetation, such as bulrushes or cattails (Jennings et al. 1992), so that the egg mass floats on the surface of the water (Hayes and Miyamoto 1984). Eggs hatch in 6 to 14 days (Jennings 1988b), and about 3 to 7 months after hatching, larvae metamorphose into adults.

The diet of California red-legged frogs is highly variable. Larvae probably eat algae (Jennings et al. 1992), while the most common food item of adults is invertebrates (Hayes and Tennant 1985). Individuals disperse upstream and downstream of their breeding habitat to forage and seek aestivation (summer dormancy) habitat, which is essential for the survival of California red-legged frogs within a watershed during the dry season. Aestivation habitat, and the ability to reach it, can be limiting factors in population numbers and survival. Aestivation habitat potentially includes all aquatic and riparian areas within the range of the species, and includes any landscape features that provide cover and moisture during the dry season within 300 feet of a riparian area. Landscape features that red-legged frogs utilize for cover and moisture include small mammal burrows and moist leaf litter (Jennings and Hayes 1994b). Frogs may also use boulders or rocks and organic debris such as downed trees or logs; industrial debris; and agricultural features, such as drains, watering troughs, spring boxes, abandoned sheds, or hay-ricks.

During dry periods, the California red-legged frog is rarely encountered far from water. However, during periods of wet weather, starting with the first rains of fall, some individuals may make overland excursions through upland habitats. Most of these overland movements occur at night. Frogs have been observed to make long-distance movements that are straight-line, point-to-point migrations, rather than using corridors for moving between habitats.

The California red-legged frog was listed as threatened on May 23, 1996. On March 13, 2001, a total of approximately 4,140,440 acres of critical habitat were designated in the following counties in California: Alameda, Butte, Contra Costa, El Dorado, Fresno, Kern, Los Angeles, Marin, Mariposa, Merced, Monterey, Napa, Plumas, Riverside, San Benito, San Diego, San Joaquin, San Luis Obispo, San Mateo, Santa Barbara, Santa Clara, Santa Cruz, Solano, Sonoma, Stanislaus, Tehama, Tuolumne, and Ventura. The primary constituent elements for California red-legged frogs are aquatic and upland areas where suitable breeding and nonbreeding habitat is interspersed throughout the landscape, and that are interconnected by continuous dispersal habitat.

In most streams, California red-legged frogs are threatened by more than one factor. Factors associated with declining populations of the frog include degradation and loss of habitat through agriculture, urbanization, mining, overgrazing, recreation, timber harvesting, the introduction of non-native plants, impoundments, water diversions, degraded water quality, and introduced predators.

Coachella Valley Fringe-toed Lizard

The Coachella Valley fringe-toed lizard (*Uma inornata*) is a narrow endemic, restricted to areas of fine, windblown sand deposits in the sandy plains, sand hummocks, and mesquite dunes of the Coachella Valley in Riverside County, California (California Department of Fish and Game 2000d). The species requires fine, loose sand for burrowing for shelter during cold temperatures and extreme heat. The sand dunes on which the species occurs are referred to as blowsand habitat, and consist of the fine sand that accumulates at the bottom of drainages during flood events, and that is transported across the Coachella Valley by high winds that continually blow through the area. Typically, vegetation in these sand dune habitats is scarce, consisting of creosote bush and other types of scrubby growth (Stebbins 1985). However, lizards do rely on some plants (mostly perennials) for shelter and food (Durtsche 1995).

Coachella Valley fringe-toed lizards hibernate during the winter, and are most active during the daylight hours. When summer temperatures reach or exceed limits that could be lethal to lizards, lizards escape the heat by burrowing beneath the sand and restricting their activities to the early morning and late afternoon hours. In May, flowers and plant-dwelling arthropods are the primary foods for the Coachella Valley fringe-toed lizard (NatureServe Explorer 2001). After the breeding season is over in the summer, when food abundance is low, the diet broadens to include ground-dwelling arthropods and foliage (Durtsche 1992, 1995). Lizard hatchlings are also eaten, when available.

Coachella Valley fringe-toed lizards breed from late April through mid-August. Little is known about the location and timing of egg laying; however, hatchlings begin to appear from late June to early September.

The Coachella Valley fringe-toed lizard was federally listed as threatened on September 25, 1980. On the same date, the USFWS designated approximately 20,000 acres of land as critical habitat. This critical habitat includes the areas with the highest concentrations of lizards, as well as a source for the blowsand habitat on which the lizard depends. Potential threats to the continued survival of this species include proposed flood control projects in the area (USFWS 1990e), movement of windblown sand out of conservation areas where this species occurs (California Department of Fish and Game 2000d), and OHV use. About 75% of the original habitat for this species has already been lost.

Desert Tortoise

The primary reference for this section is:
USFWS. 1994j. Desert Tortoise (Mojave population) Recovery Plan. Portland, Oregon.

References cited in this section are internal to the above-referenced document. They are included in the Bibliography.

The desert tortoise (*Gopherus agassizii*) occurs in desert regions of the southwestern U.S. and northwestern Mexico. Populations of this species are found in the Mojave and Sonoran deserts. The so-called Mojave population, which includes desert tortoises north and west of the Colorado River, is currently listed under the ESA. Prior to European settlement of the Mojave Desert region, the desert tortoise's range included the Mojave and Sonoran deserts in southern California, southern Nevada, western Arizona, the southwestern tip of Utah, and Sonora and Sinaloa, Mexico. This species is also found on Tiburon Island in the Sea of Cortez (Linsdale 1940). The desert tortoise is now considerably reduced in numbers throughout much of this area, and has been extirpated from parts of its historic range (Berry 1978a, b; Spang et al. 1988).

Within the varied vegetational communities of the Mojave Desert region, desert tortoises can potentially survive and reproduce where their basic habitat requirements are met. These requirements include sufficient suitable plants for forage and cover, and suitable substrates for burrow and nest sites. Throughout most of the Mojave Desert region, desert tortoises occur primarily on flats and bajadas with soils ranging from sand to sandy-gravel, and that are characterized by scattered shrubs and abundant inter-shrub space for growth of herbaceous plants. Desert tortoises are also found on rocky terrain and slopes in parts of the Mojave Desert region, and there is substantial geographic variation in the way tortoises use available resources.

Desert tortoises spend much of their lives in burrows, emerging to feed and mate during late winter and early spring. They typically remain active through the spring, and sometimes emerge again after summer storms. During these activity periods, desert tortoises eat a wide variety of herbaceous plants, particularly grasses and the flowers of annual plants (Berry 1974, Luckenbach 1982). Desert tortoises exhibit delayed maturity and live a long life. Eggs and hatchlings are quite vulnerable, and pre-reproductive adult mortality averages 98% (Wilbur and Morin 1988, Turner et al. 1987). Adults, however, are well-protected against most predators (apart from humans) and other environmental hazards (Turner et al. 1987; Germano 1992). Their longevity helps compensate for their variable annual reproductive success, which is correlated with environmental conditions.

The Mojave population of desert tortoise (including any Sonoran Desert tortoises outside of their normal range) was federally listed as threatened on April 2, 1990. On February 8, 1994, the USFWS designated approximately 6.4 million acres of desert as critical habitat for this species. The Mojave population was listed in response to precipitous declines in desert tortoise numbers in many areas. For the most part, these declines have been attributed to direct and indirect human-caused mortality, coupled with the inadequacy of existing regulatory mechanisms to protect desert tortoises and their habitat. Impacts such as the destruction, degradation, and fragmentation of habitat result from urbanization, agricultural development, livestock grazing, mining, and roads. Furthermore, direct mortality to tortoises is caused by a number of human activities. Finally, an upper respiratory tract disease is an additional major cause of mortality and population decline, particularly in the western Mojave Desert.

New Mexican Ridge-nosed Rattlesnake

The New Mexican ridge-nosed rattlesnake (*Crotalus willardi obscurus*) is a subspecies that is endemic to the Animas and Peloncillo mountains of southern New Mexico and Arizona, and the Sierra de San Luis mountains of Mexico. The population of this subspecies within its restricted range has been reduced by collection, as it is commercially very valuable and much sought after by private herpetoculturists (New Mexico Department of Game and Fish 1994). Habitat for the New Mexican ridge-nosed rattlesnake is high elevation pine-oak woodlands and pine-fir forests, as well as foothill canyons in pinyon-juniper woodland (NatureServe Explorer 2001). Rattlesnakes seek cover and shelter to escape from bad weather and predators. Winter retreats are probably talus areas and other labyrinthian formations (e.g., rock outcrops, cliffs/ledges) that allow the snakes to move below the frost line. Similar sites may be used at other times of the year, although in warm weather the species is often found on or near vegetated areas (Applegarth et al. 1980). Rattlesnakes hide in leaf litter among cobbles and rocks, and may climb into trees and shrubs (New Mexico Department of Game and Fish 1994). They are inactive in cold temperatures and in extreme heat, with most activity occurring during daylight hours from July through September (Ernst 1992). In the summer, activity peaks during warm humid mornings, and in the fall, activity peaks during the afternoon.

The bulk of the rattlesnake's diet is presumed to consist of small vertebrates, such as lizards, small mammals and birds (Vorhies 1948). Invertebrates may also be taken on occasion. The most frequently recorded prey species include the Yarrow's spiny lizard, the Arizona alligator lizard (Klauber 1949, Woodin 1953), and the brush mouse (Woodin 1953, Klauber 1972).

Rattlesnakes are ovoviviparous, with fertilized eggs being retained in the female until hatching occurs (Klauber 1972). The gestation period for the species is 13 months. Young disperse immediately after birth.

The New Mexican ridge-nosed rattlesnake was federally listed as threatened on August 4, 1978. Critical habitat has been designated in Hidalgo County, New Mexico, at elevations between 6,200 and 8,532 feet in Bear, Indian, and Spring canyons in the Animas Mountains. Threats to this species include overcollecting and factors that alter habitat, such as heavy livestock grazing, the misuse of controlled fire (Nature Serve Explorer 2001, New Mexico Department of Game and Fish 1994), development, OHV use, pollution, mining, and timber harvesting.

Giant Garter Snake

The main reference for this section is:
USFWS. 1993l. Determination of Threatened Status for the Giant Garter Snake. Federal Register 58(201): 54053-54066.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

The giant garter snake (*Thamnophis gigas*) is endemic to valley floor wetlands in the Sacramento and San Joaquin valleys of California. The species inhabits marshes, sloughs, ponds, small lakes, low gradient streams, and other waterways and agricultural wetlands (e.g., irrigation and drainage canals and rice fields).

Giant garter snakes require adequate water during the active season (early spring through mid-fall), emergent herbaceous wetland vegetation for escape cover and forage habitat, grassy banks and openings in vegetation for basking, and higher elevation uplands that provide refuge and cover from flood waters during the winter dormant season. Giant garter snakes are absent from larger rivers and other water bodies that support introduced populations of large, predatory fish. They are also absent from wetlands with sand, gravel, or rock substrates (Hansen 1980, 1988; Rossman and Stewart 1987; Brode 1988). In addition, riparian woodlands do not offer suitable habitat because of their excessive shade, the lack of basking sites, and the absence of prey populations (Hansen 1980). Giant garter snakes feed on small fishes, tadpoles, and frogs (Fitch 1941; Hansen 1980, 1988).

The giant garter snake inhabits small mammal burrows and other soil crevices above prevailing flood elevations throughout its winter dormancy period, which occurs from November to mid-March (Hansen 1991). Snakes typically select burrows with sunny aspects and west-facing slopes. Upon emergence, males immediately begin wandering in search of mates. The breeding season extends through March and April, and females give birth to live young from late July through early September (Hansen and Hansen 1990). Brood size is variable, ranging from 10 to 46 young, with an average of about 23. Young immediately scatter into dense cover and absorb their yolk sacs, after which they begin feeding on their own.

The giant garter snake was federally listed as threatened on October 20, 1993. Critical habitat has not been designated. This species is threatened by habitat loss from urbanization, flooding, contaminants, agricultural and maintenance activities, and introduced predators.

Blunt-nosed Leopard Lizard

The primary reference for this section is:
USFWS. 1998h. Recovery Plan for Upland Species of the San Joaquin Valley, California. Region 1. Portland, Oregon.

The blunt-nosed leopard lizard (*Gambelia silus*) is endemic to the San Joaquin Valley of central California (Stejneger 1893; Smith 1946; Montanucci 1965, 1970; Tollestrup 1979a). Although the boundaries of its original distribution are uncertain, blunt-nosed leopard lizards probably occurred from Stanislaus County in the north, southward to the Tehachapi Mountains in Kern County. The currently known occupied range of the species is in scattered parcels of undeveloped land on the San Joaquin Valley floor, and in the foothills of the Coast Range. It does not occur above 2,600 feet in elevation.

Blunt-nosed leopard lizards inhabit open, sparsely vegetated areas of low relief on the San Joaquin Valley floor and in the surrounding foothills (Smith 1946, Montanucci 1965). On the valley floor, they are most commonly found in the non-native grassland and valley sink scrub communities described by Holland (1986). The valley sink scrub is dominated by low, alkali-tolerant shrubs of the chenopod family, such as iodine bush and seepweeds. The soils are saline and alkaline lake bed or playa clays that often form a white salty crust and are occasionally covered by introduced annual grasses. Valley needlegrass grassland, non-native (annual) grassland, and alkali playa also provide suitable habitat for the lizard on the Valley floor. Valley needlegrass grassland is dominated by native perennial bunchgrasses, which are associated with native and introduced annual plants. Blunt-nose leopard lizards also inhabit valley saltbush scrub, a low shrubland, with an annual grassland understory, that occurs on the gently sloping alluvial fans of the foothills of the southern San Joaquin Valley and the adjacent Carrizo Plain.

Blunt-nosed leopard lizards feed primarily on insects (mostly grasshoppers, crickets, and moths) and other lizards, although some plant material is eaten rarely, or perhaps consumed unintentionally with animal prey. The lizard appears to feed opportunistically on animals, eating whatever is available in the size range they can overcome and swallow. Young of its own species are also eaten (Montanucci 1965; Kato et al. 1987; Germano and Williams 1994a).

Breeding activity begins within a month of emergence from dormancy and lasts from the end of April through the end of June. During this period, and for a month or more afterward, the adults often are seen in pairs and frequently occupy the same burrow system (Montanucci 1965, Germano and Williams 1994b). Male territories may overlap those of several females, and a given male may mate with several females. Copulation may occur as late as June (Montanucci 1965). Females lay two to six eggs in June and July, in a chamber either excavated specifically for a nest or already existing within the burrow system (Montanucci 1965, 1967). Females typically produce only one clutch of eggs per year, but some may produce three or more under favorable environmental conditions (Montanucci 1967; USFWS 1985m; Germano and Williams 1992; Willams et al. 1993a). After about 2 months of incubation, young hatch from July through early August, and rarely to September. Sexual maturity is reached at between 9 and 21 months, depending on the sex and environmental conditions (USFWS 1985m).

The blunt-nosed leopard lizard was federally listed as endangered in 1967. Critical habitat has not been designated. The greatest threats to this species are habitat disturbance, destruction, and fragmentation. Construction of facilities related to oil and natural gas production, such as well pads, wells, storage tanks, sumps, pipelines, and their associated service roads degrade habitat and cause direct mortality to leopard lizards, as do oil leakage from pumps, transport pipes, and storage facilities, surface mining, and OHV traffic (Mullen 1981, USFWS 1985m, Kato and O'Farrell 1986, Madrone Associates 1979, Chesemore 1980). Livestock grazing can result in the removal of herbaceous vegetation and shrub cover, destruction of rodent burrows used by lizards for shelter, and associated soil erosion if the stocking rate is too high or animals are left on the range too long after annual plants have died (Chesemore 1981, Williams and Tordoff 1988). However, unlike the cultivation of row crops, which precludes use by leopard lizards, light or moderate grazing may be beneficial (Chesemore 1980, USFWS 1985m, Germano and Williams 1993). The lizards are believed to prefer lightly-grazed grasslands, which are dominated by a low, sparsely growing annual grass, rather than the taller, denser, introduced red brome that dominates ungrazed sites.

The use of pesticides may directly and indirectly affect lizards. In addition, lizard mortality is known to occur as a result of automobile traffic and OHV use (Tollestrup 1979b; Uptain et al. 1985; Williams and Tordoff 1988). Roads also bisect remaining fragments of habitat, increasing the risks of mortality by vehicles and increasing the effects of isolation of populations.

Effects of Treatment Activities on Amphibians and Reptiles

Effects Common to All Treatment Methods

Indirect Effects

Removal of hazardous fuels could negatively affect amphibians and reptiles by eliminating important sources of cover (e.g., large woody debris) over the short term. However, removal of weeds and a reduction in the risk of a future catastrophic wildfire would likely have positive long-term effects on habitat components.

Prescribed Fire Treatments

Direct Effects

Direct injury to herpetofauna by prescribed fire is thought to be uncommon, even in species with limited mobility (Russell et al. 1999). Species or life phases of species (including the Sonora tiger salamander, California tiger salamander, Wyoming toad, arroyo toad, California red-legged frog, and giant garter snake) that occupy aquatic habitats can continue their activities with little interruption by fire. In addition, the wetlands and other moist habitats occupied by a number of these species are likely to burn less severely than upland sites (Smith 2000).

Some of the species discussed occur solely in upland habitats (desert tortoise, blunt-nosed leopard lizard, Coachella Valley fringe toed lizard, and New Mexican ridge nosed rattlesnake), or spend some portion of their lives in upland areas. The California red-legged frog, for example, may travel cross-country outside of riparian corridors during the spring and fall and would be most susceptible to injury from fire during these times. Even in upland habitats, however, most herpetofauna are often able to survive fires by burrowing into the soil. In addition, in desert and semi-desert habitats, sparse fuel loads cause patchy spreads of fire, possibly protecting herpetofauna from fire-related injury and mortality (Smith 2000). In areas with heavy fuel loads, however, the risks associated with mortality would be higher.

Out of all the TEP species discussed above, the desert tortoise is probably the most at risk for direct injury from prescribed fire, because it is slow moving and unable to quickly flee an area. Despite its ability to burrow into the soil, fragments of tortoise shells have been found in recently burned areas (Woodbury and Hardy 1948).

Prescribed fires may result in a large influx of heated slag and ash into aquatic systems, which can have both immediate and direct impacts (Fresques et al. 2002). These materials may briefly elevate water levels to lethal temperatures. In addition, they impact water quality, as phosphate leaches from the ash and pH is altered.

Indirect Effects

Prescribed burning is likely to affect amphibian and reptile habitats, with the nature of effects depending not only on the severity of the fire itself, but on the habitat requirements of each particular species. Fire in isolated wetlands usually increases the area of open water and enhances the vegetation structure favored by many aquatic and semiaquatic herpetofauna (Russell et al. 1999). Fire typically returns a community to an early-successional, more open state, resulting in small short-term increases in populations of species that prefer open sites, and decreases in populations of species that use or can tolerate dense vegetation (Simovich 1979). Desert species, such as the desert tortoise and Coachella Valley fringe-toed lizard, for instance, often require open, grassy conditions for optimum food and nesting, habitat that can be improved by fire. Forest species like the New Mexican ridge-nosed rattlesnake, on the other hand, often utilize litter and woody material on the forest floor—that is burned during a fire—for cover. Prescribed fire would also benefit the habitat of many species over the long term by decreasing fuel loads and reducing the risk of future catastrophic fire.

Most herpetofauna feed on insects and other invertebrates, which may be killed during prescribed fire. However, fire is unlikely to cause enough of a shortage in invertebrate populations to negatively affect populations of TEP species. The New Mexican ridge-nosed rattlesnake, the giant garter snake, and the blunt-nosed leopard lizard feed on other small vertebrates, which could also experience some mortality during a fire. The desert tortoise feeds on native desert plants, so prescribed fire could burn suitable forage if conducted during the summer, when the species is actively foraging. Impacts would be greatest if burns were conducted repeatedly, at a frequency greater than the postburn recovery time of forbs and grasses (Snyder 1991a). Over the long term, fire can have both positive and negative effects on the availability of forage in desert habitats, depending on the situation. Prescribed fires may be destructive to woody plants and cacti without increasing the amount of forbs and grasses (Bock and Bock 1987), although in grassland areas that have been invaded by desert scrub, such fires may help to restore grassland. There have also been observations of reduced productivity for 10 years or longer (Wright 1980).

Herpetofauna in aquatic habitats may be indirectly affected by influxes of ash or sediment into occupied waters. An inflow of these materials could smother eggs, clog the gills of larvae, and inhibit respiration in the invertebrates on which they feed (Agyagos et al. 2001). Factors that affect eggs and larvae could potentially reduce the number of animals in the population that reach a reproductive age.

Prescribed fire could result in the loss of some riparian vegetation. Riparian vegetation provides cover to aestivating adults, and provides shade and cover to the adjacent aquatic habitats. Loss of riparian vegetation can cause increased fluctuations in water temperature, decreased water storage capacity, and increased erosion potential. A resulting increase in sedimentation into aquatic habitats from erosion could reduce the amount of suitable habitat for certain species (USDA Forest Service 2002). Over the long term, there could also be increases in runoff and higher peak flows until adequate vegetation stabilizes the soil and retains water.

Mechanical Treatment Methods

Direct Effects

Equipment used during mechanical treatments can directly affect herpetofauna in upland habitats by killing or injuring individuals, including those seeking cover in shallow burrows. During removal of downed woody material, placing the material into piles could also crush animals.

Indirect Effects

Mechanical treatments would be expected to increase the potential for erosion over the short term, resulting in some sediment inflow into aquatic habitats. Like ash and sediment resulting from fire, this sediment could cause mortality by smothering eggs and larvae, and inhibit respiration in invertebrates on which the herpetofauna feed. Use of equipment may also crush other invertebrates and vertebrates upon which certain species feed.

Manual Treatment Methods

Direct and Indirect Effects

Manual treatments would be unlikely to affect TEP herpetofauna populations. Most reptiles and amphibians would be able to move away from treatment sites, or would be hidden in burrows or aquatic habitats that would not likely be disturbed during treatments.

Biological Control Treatments

Domestic Animals

Direct Effects. Use of domestic animals to contain weeds in upland or aquatic habitats occupied by listed reptiles and amphibians can cause death injury to animals through trampling. Domestic animals may also disturb egg masses and larvae, potentially reducing the number of individuals that reach reproductive age.

Indirect Effects. Use of domestic animals could adversely affect aquatic and riparian habitats utilized by listed reptile and amphibian species. One study indicated that exclusion of cattle grazing resulted in re-establishment of native trees and native wetland herbs, re-establishment of creek pools, and an expansion of frog populations into streams and ungrazed stock ponds (Dunne 1995). When cattle drink from small ponds and streams, they can draw down water levels, leaving egg masses above the water surface, thereby subjecting them to desiccation and/or disease (USDA Forest Service 2002).

Other effects of grazing on aquatic habitats include nutrient loading; reduction of shade and cover, which result in increases in water temperature; more intermittent flows; changes in stream channel morphology; and sedimentation caused by bank degradation and off-site soil erosion (USDA Forest Service 2002). Presence of domestic animals in riparian vegetation can cause mass erosion from trampling, hoof slide, and streambank collapse, all of which cause soils from the bank to enter the stream, reducing the quality of habitat. Trampling can also compact the soils and reduce infiltration, which in turn may decrease the recharge of the saturated zone and increase peak flow discharge. Removal of streambank vegetation, in addition to causing greater fluctuations in temperature, can also result in decreased water storage capacity and increased erosion potential. The removal of vegetation in upland areas can also increase erosion, as well as reducing water infiltration and accelerating runoff.

There would likely be long-term positive effects from using domestic animals to contain weeds, provided guidelines to increase protection of riparian vegetation and streambanks were followed. Grazing can result in reduced erosion through the growth of stabilizing vegetation and improvement of aquatic habitats by increasing the number and size of woody shrubs along streams. Over the long-term, there might also be a reduction of sediment loading into streams for most flow regimes, and a reduction of summer stream temperatures as woody vegetation along streambanks began to provide increasing levels of shade.

Desert tortoises could be adversely affected by treatments involving domestic animals, as tortoises depend on herbaceous forage for food. However, blunt-nosed leopard lizards could receive benefits from light or moderate grazing, as they are believed to prefer grasslands that are dominated by low, sparsely growing annual grasses over taller, denser, grassland habitats.

Other Biological Control Agents

Direct and Indirect Effects. Few effects from the use of biological control agents are expected. There could be some minor disturbance from the presence of workers in reptile and amphibian habitat, but it would be of short duration. In addition, there are always risks associated with the release of biological control agents into a natural ecosystem. All biological control agents are tested under laboratory conditions prior to their approval for release in the wild. Impacts to native ecosystems from their use not reasonably foreseeable.

Herbicides

Direct Effects

Herbicide treatments in upland habitats could result in the crushing of herpetofauna, primarily by vehicles, which could injure or kill individuals. Although most herpetofauna would attempt to escape work crews, many individuals would do so by seeking cover in shallow burrows, where they would not necessarily be protected from crushing. Most animals in such hiding places would not be exposed directly to herbicides during the application, but direct spray of some animals could occur. Reptiles and amphibians in terrestrial life history stages could potentially be exposed to direct spray of chemicals, come into contact with sprayed vegetation after a treatment, or ingest sprayed prey items after a treatment. Amphibians in aquatic environments could be exposed to herbicides entering the water through various exposure pathways (direct spray of herbicides directly into a water body, off-site drift of herbicides applied to adjacent uplands into a water body, runoff from upland areas, or an accidental spill of herbicides directly from a truck/ATV or helicopter into a water body). Amphibian TEP species that occur in aquatic habitats for at least a portion of their lives include all the amphibians addressed in this BA, with the exception of the desert slender salamander.

For scenarios that assess the risks of ingesting contaminated food, ERAs used animals of a similar trophic guild as surrogates to assess risks. For those species that consume more than one type of food (e.g., small mammals and invertebrates), more than one surrogate species was used in the analysis.

Terrestrial Scenarios

For scenarios evaluating dermal contact with herbicides (direct spray or contact with contaminated foliage), ERAs primarily utilized small mammals to represent terrestrial vertebrate species for risk calculations. Data pertaining to reptiles and amphibians is largely unavailable, so a more accurate calculation of risk to TEP herpetofauna was not possible. However, since ERAs utilize very conservative assumptions (e.g., 50% of the animal's surface would be exposed to direct spray, and 100% of the herbicide would be absorbed through the skin), it is unlikely that they would underestimate risks to reptiles and amphibians. Therefore, it is assumed that the terrestrial vertebrate analysis is adequate to extrapolate risks for TEP reptiles and amphibians.

Based on information in the ERAs, direct spray of herpetofauna by 2,4-D, clopyralid, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, or triclopyr could potentially result in adverse health effects to herpetofauna. This information is summarized in Table 6-2, which provides additional information on the application rates for which risks to terrestrial vertebrates were predicted, as well as the relative level of risk for each herbicide. As shown in the table, dermal contact with vegetation treated by glyphosate, hexazinone, or triclopyr at the maximum application rate, or vegetation treated by 2,4-D at the typical application rate, could potentially result in adverse health effects to herpetofauna as well.

As discussed in ERAs completed by the BLM, very few laboratory studies have been conducted to assess the adverse effects of herbicides on reptiles and amphibians (ENSR 2005a-j). However, it is assumed that the potential toxicological effects of herbicides on reptiles and amphibians would be similar to those on other terrestrial species, and would include mortality and sublethal effects. According to the limited laboratory data that are available, sublethal effects may include behavioral alteration, slowed growth, developmental effects, and illness (Sparling et al. 2000). It is assumed that sublethal effects could also include reduced reproductive success. In this discussion, the term "adverse health effects" refers to the abovementioned or similar toxicological effects at the level of the organism. In addition, it is assumed that for TEP reptiles and amphibians, these adverse health effects would potentially result in population-level effects for the species in question. Because many TEP herpetofauna species already have reduced, sensitive populations, mortality of individuals or reduced reproductive output could reduce the size of affected populations further, perhaps even leading to extirpation. Furthermore, if individuals were to become more physiologically predisposed to mortality from environmental stresses (such as predation, exposure to harsh environmental conditions), the risk for future population-level effects, including extirpations, would be increased.

Reptiles and adult amphibians could ingest vegetation or prey items that were sprayed during herbicide treatments. Table 6-3 lists the TEP herpetofauna addressed in this BA, their dietary components during terrestrial phases, and the potential risks associated with herbicides proposed for use by the BLM. For species that strictly eat invertebrates, ingestion of prey items that have been sprayed by 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate, or by clopyralid or imazapyr at the maximum application rate could result in adverse health effects. For species that also eat small vertebrates, ingestion of vertebrate prey items that have been sprayed by bromacil at the maximum application rate could potentially result in adverse health effects as well.

Since ingestion of vertebrate prey contaminated by 2,4-D, glyphosate, hexazinone, picloram, or triclopyr was not examined in the ERAs for these herbicides, the potential for adverse effects to reptiles from exposure to these chemicals via this exposure pathway cannot be determined. In the case of the herbaceous desert tortoise, consumption of plant materials that have been treated by 2,4-D, diquat, glyphosate, hexazinone, or triclopyr at the typical application rate, or by bromacil, clopyralid, diuron, or imazapyr at the maximum application rate, could result in adverse health effects.

TABLE 6-2
Summary of Effects to Terrestrial Vertebrates

Herbicide	Direct Spray	Level of Risk[1]	Dermal Contact with Sprayed Vegetation	Level of Risk
2,4-D	Adverse effects	Typical rate: M Maximum rate terrestrial: M Maximum rate aquatic: H	Adverse effects	Typical rate: L Maximum rate terrestrial: L Maximum rate aquatic: M
Bromacil	No effects	--	No effects	--
Chlorsulfuron	No effects	--	No effects	--
Clopyralid	Adverse effects	Typical rate: L Maximum rate: L	No effects	--
Dicamba				
Diflufenzopyr	No effects	--	No effects	--
Diquat	No effects	--	No effects	--
Diuron	No effects	--	No effects	--
Fluridone	No effects	--	No effects	--
Glyphosate	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L
Hexazinone	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L
Imazapic	No effects	--	No effects	--
Imazapyr	Adverse effects	Typical rate: N/A Maximum rate: L	No effects	--
Metsulfuron methyl	Adverse effects	Typical rate: N/A Maximum rate: L	No effects	--
Overdrive®	No effects	--	No effects	--
Picloram	Adverse effects	Typical rate: L Maximum rate: L	No effects	--
Sulfometuron methyl	No effects	--	No effects	--
Tebuthiuron	No effects	--	No effects	--
Triclopyr acid	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L
Triclopyr BEE	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L

[1] L = low risk; M = medium risk; H = high risk; N/A = ERAs did not predict risk at this application rate.
Note: Diquat and fluridone are aquatic herbicides that would not be used by the BLM in terrestrial applications. For 2,4-D, the maximum terrestrial application rate, rather than the maximum aquatic application rate, is the maximum rate that would be used in terrestrial applications.

Aquatic Scenarios

For aquatic scenarios, fish were used as surrogates to predict risk to amphibian species. Available toxicity information for some herbicides indicates that amphibians and fish have a similar sensitivity to herbicides. Given

TABLE 6-3
Summary of Effects to Herpetofauna from Ingestion of Food Contaminated by Herbicides

Species	Food (During Terrestrial Stage)	Summary of Effects
Desert slender salamander	Arthropods	Adverse effects from 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate; adverse effects from clopyralid or imazapyr at the maximum application rate.
Sonora tiger salamander[1]	Invertebrates, fish, amphibians, and small mammals	Adverse effects from 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate; adverse effects from bromacil, clopyralid, or imazapyr at the maximum application rate.
Chiricahua leopard frog	Invertebrates	Adverse effects from 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate; adverse effects from clopyralid or imazapyr at the maximum application rate.
Wyoming toad	Insects	Adverse effects from 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate; adverse effects from clopyralid or imazapyr at the maximum application rate.
California tiger salamander	Invertebrates	Adverse effects from 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate; adverse effects from clopyralid or imazapyr at the maximum application rate.
Arroyo toad	Invertebrates	Adverse effects from 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate; adverse effects from clopyralid or imazapyr at the maximum application rate.
California red-legged frog	Invertebrates	Adverse effects from 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate; adverse effects from clopyralid or imazapyr at the maximum application rate.
Coachella valley fringe-toed lizard	Arthropods	Adverse effects from 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate; adverse effects from clopyralid or imazapyr at the maximum application rate.
Desert tortoise	Herbaceous plants	Adverse effects from 2,4-D, diquat, glyphosate, hexazinone, or triclopyr at the typical application rate; adverse effects from bromacil, clopyralid, diuron, imazapyr, or tebuthiuron at the maximum application rate.
New Mexican ridge-nosed rattlesnake[1]	Vertebrates, invertebrates	Adverse effects from 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate; adverse effects from bromacil, clopyralid, or imazapyr at the maximum application rate.
Giant garter snake[1]	Fish, vertebrates, invertebrates	Adverse effects from 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate; adverse effects from bromacil, clopyralid, or imazapyr at the maximum application rate.
Blunt-nosed leopard lizard[1]	Insects, lizards	Adverse effects from 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate; adverse effects from bromacil, clopyralid, or imazapyr at the maximum application rate.

1 – For these species, the ERA for hexazinone did not address exposure via ingestion of small mammals and other vertebrates.

the conservative approach taken in completing risk assessments, it was assumed that fish calculations were suitable for predicting risks to amphibians, even for species that are slightly more sensitive to the herbicides analyzed than fish. As discussed in the effects analysis for aquatic organisms, ERAs addressed the potential for effects to aquatic species via multiple exposure pathways (i.e., direct spray, off-site drift [BLM ERAs only], surface runoff, and accidental spill).

Direct Spray

Based on information provided in the ERAs (see Tables 5-2 through 5-5), direct spray of bromacil, diquat, diuron, fluridone, glyphosate, picloram, or triclopyr BEE into a water body could potentially result in adverse health effects to the aquatic amphibians addressed in this BA. In the case of fluridone, these effects were only predicted for direct spray at the maximum application rate. In the case of glyphosate, these effects were only predicted for direct spray of the more toxic formulations of the herbicide, or the less toxic formulations applied at the maximum application rate.

Off-site Drift

Of the terrestrial herbicides considered in the BLM ERAs, only diuron applied at the maximum rate would potentially cause adverse health effects to TEP amphibians as a result of off-site drift into a water body from a nearby upland treatment site. Since this exposure pathway was not examined in Forest Service ERAs, it is assumed that off-site drift of glyphosate, picloram, or triclopyr BEE would also have the potential to result in adverse health effects to aquatic amphibians.

Accidental Spill

According to the ERAs, an accidental spill of 2,4-D, bromacil, clopyralid, diquat, diuron, fluridone, glyphosate, imazapyr, metsulfuron methyl, picloram, tebuthiuron, or triclopyr into a water body could potentially result adverse health effects to the aquatic amphibian species addressed in this BA. Adverse effects to aquatic amphibians were assumed for an accidental spill of hexazinone as well.

Surface Runoff

According to ERAs, surface runoff of bromacil, diuron, tebuthiuron, or triclopyr BEE into a water body from upland areas could result in adverse health effects to TEP amphibians present in that water body.

Indirect effects

Treatment of upland sites with herbicides could result in a substantial, though temporary, reduction in vegetative cover, particularly if a site was broadcast sprayed with a broad-spectrum formulation. Such a loss of vegetation could indirectly impact TEP reptiles and amphibians by removing cover. However, other important components for cover, such as duff and woody debris would be maintained, and could even increase in quantity. It is possible that prey items, such as invertebrates, could also be reduced temporarily as a result of crushing, toxicity from spraying, or loss of habitat. However, long-term adverse effects to habitat should not occur. Furthermore, treatments to reduce weedy species could benefit herpetofauna habitat by returning it to a more native state.

Conservation Measures

Many local BLM offices already have management plans in place that ensure the protection of these species during activities on public lands. In addition, the following conservation measures are the minimum steps required of the BLM to ensure that treatment methods would be unlikely to adversely affect TEP species.

Conservation measures:

- Survey all areas that may support TEP amphibians and/or reptiles prior to treatments.
- Conduct burns during periods when the animals are in aquatic habitats or are hibernating in burrows.
- For species with extremely limited habitat, such as the desert slender salamander, avoid prescribed burning in known habitat.
- Do not use water from aquatic habitats that support TEP amphibians and/or reptiles for fire abatement.
- Install sediment traps upstream of aquatic habitats to minimize the amount of ash and sediment entering aquatic habitats that support TEP species.
- In potential desert tortoise habitat, conduct prescribed burns during mid-winter, when tortoises are not dependent on forage and are hibernating in burrows.
- In habitats where aquatic herpetofauna occur, implement all conservation measures identified for aquatic organisms in Chapter 4.
- Within riparian areas, wetlands, and aquatic habitats, conduct herbicide treatments only with herbicides that are approved for use in those areas.
- Do not broadcast spray herbicides in riparian areas or wetlands that provide habitat for TEP herpetofauna.
- Do not use diquat, fluridone, glyphosate, or triclopyr BEE to treat aquatic vegetation in habitats where TEP amphibians occur or may potentially occur.
- When conducting herbicide treatments in upland areas adjacent to aquatic or wetland habitats that support TEP herpetofauna, do not broadcast spray during conditions under which off-site drift is likely.
- In watersheds where TEP amphibians occur, do not apply bromacil, diuron, or triclopyr BEE in upland habitats upslope of aquatic habitats that support (or may potentially support) TEP amphibians under conditions that would likely result in surface runoff.
- Follow all instructions and SOPs to avoid spill and direct spray scenarios into aquatic habitats that support TEP herpetofauna.
- Do not use 2,4-D in terrestrial habitats occupied by TEP herpetofauna; do not broadcast spray 2,4-D within ¼ mile of terrestrial habitat occupied by TEP herpetofauna.
- When conducting herbicide treatments in or near terrestrial habitat occupied by TEP herpetofauna, avoid using the following herbicides, where feasible: clopyralid, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr.
- When conducting herbicide treatments in upland habitats occupied by TEP herpetofauna, do not broadcast spray 2,4-D, clopyralid, glyphosate, hexazinone, picloram or triclopyr; do not broadcast spray these herbicides in areas adjacent to habitats occupied by TEP herpetofauna under conditions when spray drift onto the habitat is likely.
- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in upland habitats occupied by TEP herpetofauna, utilize the typical, rather than the maximum, application rate.
- If spraying imazapyr or metsulfuron methyl in or adjacent to upland habitats occupied by TEP herpetofauna, apply at the typical, rather than the maximum, application rate.
- If conducting herbicide treatments in or near upland habitats occupied by TEP herpetofauna, consult Table 6-3 on a species by species basis to determine additional conservation measures that should be enacted to avoid adverse effects via ingestion of contaminated prey.

Summary of Effects

Using the assumption that any of the proposed vegetation treatments could occur anywhere on public lands, these treatments would be **likely to adversely affect** any and all of the TEP amphibian and reptile species and/or their critical habitat discussed in this section. However, if the proper precautions were taken at the local level during the development of treatment programs, impacts to these species could be avoided and the effects determination could be reduced to **not likely to adversely affect**. The previous section, Conservation Measures, provides general guidance at the programmatic level. Project-specific BAs completed at the local level would provide any additional conservation measures necessary to protect the species from the proposed treatments.

Birds

The following birds, and the ecoregion they are typically found in, are considered in this BA:

Steller's eider – Tundra
Spectacled eider - Tundra
Cactus ferruginous pygmy-owl – Subtropical Desert
Northern Aplomado falcon – Subtropical Desert
Yuma clapper rail – Subtropical Desert
Southwestern willow flycatcher – various in SW
Mexican spotted owl – Numerous in SW
Least tern (interior) – Temperate Steppe
Piping plover – Temperate Steppe
Western snowy plover – Mediterranean/Marine
Least Bell's vireo – Mediterranean
Inyo California towhee – Mediterranean
Coastal California gnatcatcher – Mediterranean
Brown pelican – Mediterranean
California condor – Mediterranean
Northern spotted owl – Mediterranean/Marine
Marbled murrelet – Various in NW
Whooping crane – Numerous
Bald eagle – Numerous

Note: in the discussions that follow, the general term "adverse health effects" is used in reference to exposure to certain herbicides under certain scenarios. The potential toxicological effects of herbicides on terrestrial wildlife, which were examined in ERAs, include mortality and sublethal effects. Examples of sublethal effects include harm to vital organs, changes in body weight, reduced reproductive success, and altered behavior, which may increase the animal's susceptibility to predation (USDA Forest Service 2004). Sublethal effects to an animal's health may also increase the severity of impacts associated with unrelated environmental stresses and other disturbances. In all of the effects assessments for birds found in this chapter, the term "adverse health effects" refers to the abovementioned or similar toxicological effects at the level of the organism. In addition, it is assumed that for TEP birds, these adverse health effects would potentially result in population-level effects for the species in question. Because many TEP bird species already have reduced, sensitive populations, mortality of individuals or reduced reproductive output could reduce the size of affected populations further, perhaps even leading to extirpation. Furthermore, if individuals were to become more physiologically predisposed to mortality from environmental stresses (such as predation or exposure to harsh environmental conditions), the risk for future population-level effects, including extirpations, would be increased.

Alaskan Waterfowl: Steller's Eider and Spectacled Eider

Steller's Eider
The primary reference for this section is:
USFWS. 1997i. Threatened Status for the Alaska Breeding Population of the Steller's Eider. Federal Register 62(112): 31748-31757.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS, Ecological Services Fairbanks Field Office, Fairbanks, Alaska.

The Steller's eider (*Polystricta stelleri*) is a sea duck that spends the majority of the year in shallow, near-shore marine waters where it feeds by diving and dabbling for mollusks and crustaceans (Petersen 1980). The current breeding distribution of the Steller's eider encompasses the Arctic coastal regions of northern Alaska from

Wainwright to Prudhoe Bay up to 54 miles inland (King and Brackney 1993), and Russia from the Chukotsk Peninsula west to the Taimyr, Gydan and Yamal peninsulas (American Ornithologists' Union 1983; Yesou and Lappo 1992).

Principal foods in marine areas include bivalves, crustaceans, polychaete worms, and mollusks (Petersen 1980, Troy and Johnson 1987, Metzner 1993). During the breeding season, Steller's eiders move inland in coastal areas, where they nest adjacent to shallow ponds or within drained lake basins (King and Dau 1981; Flint et al. 1984; Quakenbush and Cochrane 1993). In inland areas, their diet includes aquatic insects (primarily chironomid larvae), plant materials, crustaceans, and mollusks (Cottam 1939, Quakenbush and Cochrane 1993). Actual numbers nesting in Alaska and Russia are unknown, but the majority of Steller's eiders nest in Arctic Russia (Palmer 1976, Bellrose 1980). After the nesting season, Steller's eiders return to marine habitats where they molt (Jones 1965; Petersen 1980, 1981).

Concentrations of molting Steller's eiders have been noted in Russia (Gerasimov *cited in* Kistchinski 1973), near St. Lawrence Island in the Bering Sea (Fay 1961), and along the northern shore of the Alaska Peninsula (Jones 1965; Petersen 1980, 1981). In some years, groups of tens of thousands may molt in the bays and lagoons along the Alaska Peninsula, in particular Nelson Lagoon and Izembek Lagoon (Petersen 1980). In other years, many of the birds complete their molt before arriving on the Peninsula (Jones 1965). During winter, most of the world's Steller's eiders concentrate along the Alaska Peninsula from the eastern Aleutian Islands to southern Cook Inlet in shallow, near-shore marine waters (Palmer 1976). They also occur, although in lesser numbers, in the western Aleutian Islands and along the Pacific coast, occasionally to British Columbia. A small number also winter along the Asian coast, from the Commander Islands to the Kuril Islands, and some are found along the north Siberian coast west to the Baltic States and Scandinavia (Dement'ev and Gladkov 1967, Frantzen 1985, Petraitis 1991, Frantzen and Henricksen 1992). In spring, large numbers concentrate in Bristol Bay before migration.

Historically, Steller's eiders nested in Alaska in two general regions: western Alaska, where the species has been essentially extirpated, and the North Slope, where the species still occurs. The breeding range of Steller's eiders in Alaska has contracted in recent decades. The species no longer nests on the Yukon-Kuskokwim (Y-K) Delta, where it was once common, or other areas in western Alaska, and is now found exclusively on the North Slope. The breeding range on the North Slope may also have contracted. In recent decades, nesting Steller's eiders have been documented in only three areas -- (1) at Barrow; (2) on the lower Colville River, where a female with young was seen in 1987 (T. Swem, unpublished data); and (3) near Prudhoe Bay, where a female with young was seen in 1993 (Johnson 1994). Near Barrow, at the northernmost tip of Alaska, Steller's eiders still occur regularly, though not annually. In some years, up to several dozen pairs may breed in a few square miles. In contrast, elsewhere on the North Slope, the species apparently occurs at extremely low densities over a huge area, and use of specific areas appears to be irregular. Current and historical population sizes remain unknown, but overall numbers have likely declined. Steller's eiders still occur over a large area on the North Slope, but at such low densities that only hundreds or a few thousand occupy the huge expanse of seemingly suitable habitat.

The Alaska breeding population of Steller's eider was federally listed as threatened on June 11, 1997. On February 2, 2001, the USFWS designated approximately 24,954,638 acres on the Y-K Delta, in Norton Sound, Ledyard Bay, and the Bering Sea between St. Lawrence and St. Matthew Islands as critical habitat for the population. The primary threats to this population are the substantial decrease in the species' nesting range in Alaska and the reduction in the number of Steller's eiders nesting in Alaska, which result in increased vulnerability of the remaining breeding population to extirpation.

Spectacled Eider

The primary reference for this section is:
USFWS. 2001k. Final Determination of Critical Habitat for the Spectacled Eider. Federal Register 66(25): 9146-9185.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS, Ecological Services Anchorage Field Office, Anchorage, Alaska.

The spectacled eider (*Somateria fischeri*) is a large sea duck, a type of waterfowl that spends at least part of its life at sea or on large waterbodies. Spectacled eiders are diving ducks that spend most of the year in marine waters where they primarily feed on bottom-dwelling mollusks and crustaceans.

In the U.S., spectacled eiders historically had a discontinuous nesting distribution from the Nushagak Peninsula in southwestern Alaska north to Barrow and east nearly to the Canadian border. Today, two breeding populations remain in Alaska. The remainder of the species breeds in Arctic Russia. On the Y-K Delta, spectacled eiders breed mostly within 9.3 statute miles of the coast, from Kigigak Island north to Kokechik Bay (USFWS 1996), with smaller numbers nesting south of Kigigak Island to Kwigillingok and north of Kokechik Bay to the mouth of Uwik Slough. The coastal fringe of the Y-K Delta is the only subarctic breeding habitat where spectacled eiders occur at high density (1.2 to 2.6 birds per square mile; USFWS 1996). Nesting is restricted to the vegetated intertidal zone, which are dominated by low wet-sedge and grass marshes and have numerous small shallow water bodies. Nests are rarely more than 680 feet from water and are usually within a few yards of a pond or lake. Presumably, nonbreeding birds remain at sea year-round until they attempt to breed at age 2 or 3. It is unknown which areas at sea are important to nonbreeding spectacled eiders.

On Alaska's North Slope, nearly all spectacled eiders breed north of 70° latitude between Icy Cape and the Shaviovik River. Within this region, most spectacled eiders occur between Cape Simpson and the Sagavanirktok River (USFWS 1996). Spectacled eiders on the North Slope occur at low densities within about 50 miles of the coast. During pre-nesting and early nesting, they occur most commonly on large shallow productive thaw lakes, usually with convoluted shorelines or small islands (Larned and Balogh 1997). Such shallow water bodies with emergent vegetation and low islands or ridges appear to be important as eider nesting and brood-rearing habitat on the North Slope (Derksen et al. 1981; Warnock and Troy 1992; Andersen et al. 1998).

Within the U.S., spectacled eiders molt in Norton Sound and Ledyard Bay, where they congregate in large, dense flocks that may be particularly susceptible to disturbance and environmental perturbations. During their time on the molting grounds (early July through October), each bird is flightless for a few weeks. During winter, spectacled eiders congregate in exceedingly large and dense flocks in pack ice openings between St. Lawrence and St. Matthew islands in the central Bering Sea (Larned et al. 1995c). Spectacled eiders from all three known breeding populations use this wintering area (USFWS 1999a); no other wintering areas are currently known.

Effects of Vegetation Treatments on Steller's Eider and Spectacled Eider

Effects Common to All Treatment Methods

Indirect Effects. During the breeding period, treatments that remove vegetation from eider nesting areas in Alaska could adversely affect the species by reducing plant cover that helps to hide nests from predators.

Treatments that reduce hazardous fuels could benefit eiders by reducing the risks that wildfire would burn nesting habitat in the future. Removal of non-native species would also benefit eiders by maintaining native habitat for use in nesting.

Prescribed Fire

Direct Effects. Burning during the nesting period could harm or kill eiders, primarily eggs and young, by burning nests. Adults would likely be able to escape harm by fleeing the site.

Indirect Effects. Burning eiders' nesting grounds could impact populations by making these areas less suitable for breeding purposes. However, these effects would only be substantial if large expanses of breeding habitat were consumed by fire.

Mechanical Treatments

Direct Effects. The use of heavy equipment in eider breeding habitat could crush nests, eggs, and newborn birds. Adults would be able to escape harm, but breeding activities could be disturbed.

Indirect Effects. Large-scale removal of vegetation from eider nesting habitats could make these areas less suitable for breeding. These effects would be short-term in duration.

Manual Treatments

Direct and Indirect Effects. Manual treatments would be unlikely to substantially affect eiders or their habitat. There could be some disturbance from the presence of workers in the area, which would be short-term in duration.

Biological Control

Domestic Animals
Direct Effects. If domestic animals (e.g., reindeer) were allowed to graze in eider breeding grounds, they could trample and destroy nests and eggs, and possibly harm young birds. They could also disturb breeding activities.

Indirect Effects. Domestic animals could adversely affect eiders by altering their nesting habitat. Domestic animals can spread weeds and alter the vegetation composition in an area.

Other Biological Control Agents
Direct and Indirect Effects. There are not likely to be effects from the release of biological control agents in eider breeding habitats. There could be some minor disturbance from the presence of workers in the breeding area, but it would not last long. In addition, there are always risks associated with the release of biological control agents into a natural ecosystem. However, adverse effects to natural systems are not reasonably foreseeable.

Herbicides

Direct Effects. It is very unlikely that the BLM would use herbicides in eider breeding areas (at this time, no herbicide treatments are proposed for Alaska base on the 2002 data call to BLM field offices). However, if herbicides were applied in eider breeding areas, nests, eggs, or newborn birds could be crushed by workers and vehicles. It is likely that adults would flee the area, but breeding activities could be disturbed. Eiders, and especially newborn birds and eggs, could inadvertently be sprayed during herbicide applications. Based on risks predicted by ERAs for terrestrial vertebrate species (see Table 6-2), inadvertent direct spray of birds by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, could conceivably have adverse health effects on eiders. Eiders could also come into contact with sprayed foliage after the application. Via this exposure pathway, adverse health effects to eiders could occur if vegetation was sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate.

Table 6-4 summarizes the risks to birds, as determined in ERAs, as a result of ingesting food items exposed to herbicides. The table lists which herbicides would potentially cause adverse effects to TEP birds via ingestion exposure pathways, and the relative risk to TEP birds at typical and maximum application rates. Since eiders primarily eat aquatic invertebrates, as well as some plant materials, indirect exposure to herbicides could occur if an eider were to consume animals or plants that had been sprayed by herbicides during vegetation treatments. According to ERAs, consumption of invertebrates exposed to 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate, or to clopyralid or imazapyr at the maximum application rate, could potentially result in adverse health effects to eiders. In addition, ERAs suggest that consumption of plant materials exposed to 2,4-D, diquat, glyphosate, hexazinone, or triclopyr at the typical application rate, or bromacil, clopyralid, diuron, imazapyr, picloram, or tebuthiuron at the maximum application rate, could potentially result in adverse health effects to eiders. Long-term consumption of contaminated vegetation sprayed by picloram at the maximum application rate could also result in adverse health effects. For all ingestion exposure scenarios, ERAs assumes that 100% of the animal's diet would come from contaminated vegetation, which is unlikely given that vegetation is a relatively minor component of eider diets.

Indirect Effects. Loss of vegetation in nesting habitats as a result of herbicide treatments would likely have an adverse effect on eiders, potentially resulting in increased predation during nesting, and reduced reproductive success at the population level. Effects would be greatest if treatments occurred just before or during the breeding season.

Conservation Measures

The following conservation measures are required to ensure that eiders would not be adversely affected by project activities:

- Prior to developing management plans associated with treatment activities, assess whether Steller's or spectacled eiders are likely to use areas proposed for treatment for nesting or brood-rearing activities.
- Do not conduct vegetation treatments during the breeding season (as determined by a qualified wildlife biologist).

Determination of Effects

Assuming that all vegetation treatments could occur anywhere on public lands, including eider breeding grounds, the proposed treatment program could affect Steller's and spectacled eiders. However, as no treatments are proposed for the tundra habitats occupied by these species at this time, the proposed treatment program is **not likely to adversely affect** eiders and/or their designated critical habitat listed in Tables 1-1. If treatments were proposed for tundra habitats in the future, their effects would be analyzed under a project-specific BA, and conservation measures to protect Steller's and spectacled eiders would be developed, as appropriate.

Cactus Ferruginous Pygmy-owl

The primary reference for this section is:
Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation, Wildland Urban Interface Fuel Treatment. Forest Service Southwestern Region. Albuquerque, New Mexico.

The cactus ferruginous pygmy-owl (*Glaucidium brasilianum cactorum*) was once common throughout much of the southern half of Arizona at elevations below 4,000 feet. However, the species has declined to the extent that it has nearly been extirpated. Surveys in the 1998-1999 season documented a total of 41 adult cactus ferruginous pygmy-owls in Arizona (USFWS 1999j).

The ferruginous pygmy-owl nests in cavities found in trees or large columnar cacti. These cavities may either be naturally formed or excavated by woodpeckers. The pygmy-owl's primary habitats are riparian cottonwood forests, mesquite-cottonwood woodlands, and mesquite bosques. Riparian habitats provide the large trees for nests and roosts, and also have a high density and diversity of animal species that constitute the pygmy-owl's prey base. Pygmy-owls also occur uncommonly and unpredictably in Sonoran Desert scrub associations comprised of paloverde, ironwood, mesquite, acacia, bursage, and columnar cacti (saguaro or organ pipe). More predictably, they are found in thick desert scrub communities found along dry washes. They also nest in mesquite-invaded grasslands in the Altar Valley area. Pygmy owls feed on a variety of animals, including birds, lizards, insects, small mammals, frogs, and earthworms.

The breeding season of the ferruginous pygmy owl runs from late winter to early spring. Between three and five eggs are laid, and incubation lasts approximately 28 days. Young fledge approximately 28 days after hatching.

TABLE 6-4
Summary of Effects to TEP Birds via Ingestion Pathways

Herbicide	Ingestion of Invertebrate Prey		Ingestion of Vegetation		Ingestion of Small Vertebrate Prey[1]		Ingestion of Fish	
	Effect	Risk Level[2]	Effect	Risk Level	Effect	Risk Level	Effect	Risk Level
2,4-D	Adverse effects	Typical rate: H Maximum rate terrestrial: H Maximum rate aquatic: H	Adverse effects	Typical rate: M Maximum rate terrestrial: H Maximum rate aquatic: H	Adverse effects	Typical rate: L Maximum rate terrestrial: L Maximum rate aquatic: M	Adverse effects	Typical rate: H Maximum rate terrestrial: H Maximum rate aquatic: H
Bromacil	No effects	--	Adverse effects	Typical rate: N/A Maximum rate: L	Adverse effects	Typical rate: N/A Maximum rate: L	No effects	--
Chlorsulfuron	No effects	--	No effects	--	No effects	--	No effects	--
Clopyralid	Adverse effects	Typical rate: N/A Maximum rate: L	Adverse effects	Typical rate: N/A Maximum rate: L	No effects	--	No effects	--
Dicamba	No effects	--	No effects	--	No effects	--	No effects	--
Diflufenzopyr	No effects	--	No effects	--	No effects	--	No effects	--
Diquat	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: L Maximum rate: H	Adverse effects	Typical rate: N/A Maximum rate: L	No effects	--
Diuron	Adverse effects	Typical rate: L Maximum rate: L	Adverse effects	Typical rate: N/A Maximum rate: M	Adverse effects	Typical rate: L Maximum rate: L	No effects	--
Fluridone	No effects	--	No effects	--	No effects	--	No effects	--
Glyphosate	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: L Maximum rate: L	No effects	--	No effects	--
Hexazinone	Adverse effects	Typical rate: M Maximum rate: M	Adverse effects	Typical rate: L Maximum rate: M	Unknown[3]	Unknown	Adverse effects	Typical rate: L Maximum rate: M

TABLE 6-4 (Cont.)
Summary of Effects to TEP Birds via Ingestion Pathways

Herbicide	Ingestion of Invertebrate Prey		Ingestion of Vegetation		Ingestion of Small Vertebrate Prey[1]		Ingestion of Fish	
	Effect	Risk Level[2]	Effect	Risk Level	Effect	Risk Level	Effect	Risk Level
Imazapic	No effects	--	No effects	--	No effects	--	No effects	--
Imazapyr	Adverse effects	Typical rate: N/A Maximum rate: L	Adverse effects	Typical rate: N/A Maximum rate: L	No effects	--	No effects	--
Metsulfuron methyl	No effects	--	No effects	--	No effects	--	No effects	--
Overdrive®	No effects	--	No effects	--	No effects	--	No effects	--
Picloram	No effects	--	Adverse effects	Typical rate: N/A Maximum rate: L (chronic risk only)	No effects	--	No effects	--
Sulfometuron methyl	No effects	--	No effects	--	No effects	--	No effects	--
Tebuthiuron	No effects	--	Adverse effects	Typical rate: N/A Maximum rate: L	No effects	--	No effects	--
Triclopyr acid	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: L Maximum rate: M	No effects	--	No effects	--
Triclopyr BEE	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: L Maximum rate: M	No effects	--	No effects	--

[1] Only ERAs for 2,4-D, clopyralid, glyphosate, metsulfuron methyl, picloram, and triclopyr assessed risks to carnivorous birds. For all other herbicides, carnivorous mammals were used as surrogates when completing risk assessments.
[2] Key: L = low risk; M = medium risk; H = high risk; and N/A = ERAs did not predict risk at this application rate.
[3] Unknown = ERAs did not assess risk to birds for this herbicide via this exposure pathway.

The cactus ferruginous pygmy-owl was federally listed as endangered on March 10, 1997. On July 12, 1999, the USFWS designated approximately 731,712 acres of riverine riparian and upland habitat in Pima, Cochise, Pinal and Maricopa counties in Arizona as critical habitat for the subspecies. The cactus ferruginous pygmy-owl is threatened primarily by past, present, and potential future destruction and modification of its habitat throughout a major portion of its range in the U.S. Areas where owls are known to exist have suffered considerable degradation, destruction, and modification caused by urban and agricultural encroachment, wood (mesquite) cutting, water diversion, channelization, livestock overgrazing, groundwater pumping, and hydrological changes resulting from various land-use practices.

Effects of Vegetation Treatments on the Cactus Ferruginous Pygmy-owl

Effects Common to All Treatment Methods

One of the primary factors responsible for the decline of the cactus ferruginous pygmy-owl in the U.S. is the loss of suitable habitat through the removal of vegetation. Therefore, all forms of vegetation treatment proposed for use by the BLM could affect owl habitat by contributing to loss of vegetation. Over the long term, however, treatment methods that target non-native species could improve habitat for pygmy-owls, provided native plant species replaced them after treatment. In addition, treatments that reduce the presence of fuels could reduce the likelihood of a future catastrophic fire that could conceivably destroy large tracts of remaining suitable habitat. In addition, there would be less likelihood that toxic fire retardants/suppressants would need to be used in pygmy-owl habitats.

Prescribed Fire

Direct Effects. Prescribed fire could cause pygmy-owl mortality by burning nesting trees and/or cacti, although adults would likely be able to escape the burn. Smoke could disturb birds and interfere with foraging and other activities.

Indirect Effects. Fire would be likely to affect pygmy-owls by removing vegetation in riparian and desertscrub habitats. Owls are dependent on overstory vegetation for nesting, roosting, perching, and catching food. In addition, mid- and lower-story vegetation is important for pygmy-owls because it provides habitat for prey items, and may also provide pygmy-owls with some protection from predation.

Mechanical Treatment Methods

Direct Effects. Vegetation treatments using mechanical methods would be unlikely to result in injury to pygmy-owl, unless nest trees or cacti were cut, which could lead to the destruction of eggs or the death of young birds. The noise and human presence associated with mechanical treatment activities would cause some disturbance to pygmy-owls, and could interfere with activities such as breeding and foraging.

Indirect Effects. Like prescribed fire, mechanical treatment methods typically remove some amount of vegetation (often shrubby species), which could indirectly affect pygmy-owls by eliminating prey species' habitat, removing vegetation used for protection from predators, and removing young vegetation that could support owl nests in the future. Heavy equipment used during treatment could also crush pygmy-owl prey items, temporarily reducing the availability of food. Removal of vegetation in riparian habitats could alter these communities by altering hydrology, as well as increasing erosion and sedimentation. Alteration of riparian areas would likely have both short- and long-term effects on pygmy-owl habitat.

Manual Treatment Methods

Direct and Indirect Effects. Manual treatment methods would be unlikely to cause direct effects to owls or owl nests. There would be some disturbance associated with the presence of humans, which would have the greatest impact on pygmy-owl populations during the breeding season, when reproductive success could be affected. However, these disturbances should be minimal and short-term in duration. There would be some removal of vegetation associated with manual treatment methods. Removal of vegetation would likely have some effects on pygmy-owl habitat, as described above, with the degree of impact dependent on the amount and types of vegetation removed.

Biological Control Treatments

Domestic Animals
Direct Effects. Use of domestic animals to control vegetation would be unlikely to directly affect cactus ferruginous pygmy-owls, which nest in the overstory and would be able to avoid contact with grazers.

Indirect Effects. Removal of vegetative cover by ingestion, as well as trampling of grass and brush would likely occur. Mid-story and ground-level vegetation has been identified as an important habitat component for pygmy-owl s, and one that may provide protection from predators and increase the density of potential prey items (USFWS 1999j). By reducing the structural complexity and altering the plant species composition of understory communities, grazing can lead to a reduced abundance of lizards, bird species, mammals, and insects. Other indirect effects of grazing are a reduced vigor of plants, and accelerated soil erosion, which can ultimately result in reduced land productivity. Removal of understory vegetation can also limit the regeneration of species that would potentially serve as future nest trees for pygmy-owls, saguaros in particular. Grazed riparian areas (riparian areas provide food, cover, and nesting habitat for cactus ferruginous pygmy-owls), typically have less ground cover, a poorly developed understory and midstory, and decreased vegetative biomass when compared to similar ungrazed riparian areas (Krueper 1995). Since riparian areas provide food, cover, and nesting habitat for cactus ferruginous pygmy-owls, weed control in these areas using domestic animals would likely have both short- and long-term negative effects on pygmy-owl habitat. The severity of effects would depend largely on the intensity, duration, and timing of treatments.

Other Biological Control Agents
Direct Effects. The release of biological control agents into pygmy-owl habitat would be unlikely to directly affect pygmy-owls or their nests.

Indirect Effects. Minor disturbances associated with the presence of humans could occur. In addition, biological control agents would act on target species, reducing the coverage of these species. This elimination of vegetation could have a negative effect on habitat, although it would occur gradually. In addition, the long-term effects on habitat could be positive if native plant species replaced the weedy target species. Finally, biological control agents could have unanticipated negative effects on pygmy-owls or their habitat of an unspecified type or duration. Such unforeseen consequences would be highly unlikely and are not expected.

Herbicides

Direct Effects. The presence of herbicide applicators and equipment in pygmy-owl habitat could temporarily disturb pygmy-owls in the area. During treatments, pygmy-owls in nesting cavities likely would be protected from direct contact with herbicides. However, pygmy-owls in exposed areas that were unable to leave the treatment site could be directly exposed to chemicals. Based on the ERAs (see Table 6-2), direct spray by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, could potentially result in adverse health effects to pygmy-owls.

Pygmy-owls could also be exposed to herbicide by touching treated plant materials or by ingesting prey items that were exposed to herbicides. According to the ERAs, contact with sprayed plant materials after an herbicide application of 2,4-D at the typical application rate, or of glyphosate, hexazinone, or triclopyr at the maximum application rate, could potentially result in adverse health effects to pygmy-owls. Based on the results of the ERAs (see Table 6-4), ingestion of prey sprayed by 2,4-D or diuron at the typical application rate, or by bromacil or diquat at the maximum application rate, would potentially result in adverse health effects to pygmy-owls. Since the ERA for hexazinone did not assess the potential risks to carnivorous birds through ingestion of contaminated prey, the potential for adverse effects to pygmy-owls from exposure to hexazinone via this exposure pathway cannot be determined.

Indirect Effects. Herbicide treatments in pygmy-owl habitats would be unlikely to alter habitat structure, since the large trees and cacti utilized by pygmy-owls for nesting would likely remain standing and continue to provide cavities even if injured or killed. Removal of lower vegetation layers could eliminate some habitat for prey species, but could also make hunting for prey easier. These effects would likely be short term in nature. If herbicide applications were to result in the mortality of young, but established saguaros, effects could last longer, as future pygmy-owl habitat would potentially be eliminated.

Conservation Measures

The following conservation measures are required to ensure that cactus ferruginous pygmy-owls would not be adversely affected by vegetation treatments:

- Prior to treatments, conduct surveys for cactus ferruginous pygmy-owls in all suitable habitat where treatments are proposed to take place.
- Limit vegetation treatments within ¼ mile of any site occupied by a cactus ferruginous pygmy-owl, or any unsurveyed suitable habitat within the project area.
- Avoid conducting vegetation treatments in pygmy-owl habitat during the nesting period (as determined by a qualified wildlife biologist).
- Do not use 2,4-D in cactus ferruginous pygmy-owl habitat; do not broadcast spray 2,4-D within ¼ mile of cactus ferruginous pygmy-owl habitat.
- Where feasible, avoid use of the following herbicides in cactus ferruginous pygmy-owl habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr.
- Avoid broadcast spraying herbicides in areas where future nesting cacti and trees occur.
- Do not broadcast spray clopyralid, diuron, glyphosate, hexazinone, picloram, or triclopyr in cactus ferruginous pygmy-owl habitat; do not broadcast spray these herbicides in areas adjacent to pygmy-owl habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, diquat, imazapyr, or metsulfuron methyl in or adjacent to pygmy-owl habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in pygmy-owl habitat, utilize the typical, rather than the maximum, application rate.

Additional conservation measures would be developed at the local level during the completion of project-specific BAs and management plans.

Determination of Effects

Assuming that vegetation treatments could occur anywhere on public lands, including habitats utilized by the pygmy-owl, the proposed treatments would be **likely to adversely affect** the cactus ferruginous pygmy-owl and/or their designated critical habitat listed in Tables 1-1. In order to avoid or minimize these effects to a **not likely to adversely affect** determination, the BLM would be required to follow the conservation measures listed in the previous section, Conservation Measures, as well as any project-specific conservation measures deemed appropriate by local BLM offices.

Northern Aplomado Falcon

The primary reference for this section is:
Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation, Wildland Urban Interface Fuel Treatment. Forest Service Southwestern Region. Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The northern aplomado falcon (*Falco femoralis septentrionalis*) inhabits the desert grasslands and savannas of Latin America. In the United States, the subspecies historically inhabited desert grasslands with mesquite and yucca, riparian woodlands in open grasslands, and sand ridges with yuccas on the coastal prairies of Texas, New Mexico, and southeastern Arizona (Henshaw 1875, Merrill 1878, Bendire 1892, Ligon 1961). In general, open landscapes with scattered trees and shrubs provide good habitat (USFWS 1990f). Other necessary habitat components include moderately low ground cover, an abundance of small to medium sized birds, and a supply of nesting platforms (e.g., large bromeliads and stick nests; Hector 1981, 1983). There are a total of 22 grassland areas within the historical range of the species in southeastern Arizona and southern New Mexico that offer suitable habitat conditions for the aplomado falcon.

Aplomado falcons prey primarily on other birds (e.g., cuckoos, doves, woodpeckers, blackbirds, flycatchers, and thrushes), supplementing their diet with insects, small mammals, and herpetofauna (e.g., grasshoppers, butterflies, crickets, wasps, frogs, lizards, bats, and rodents; USFWS 1992i). Falcons typically initiate hunting in flight, but will chase prey on foot if necessary.

Aplomado falcons do not construct their own nests, and are therefore dependent on nesting platforms constructed by other species, such as the stick nests of Swainson's hawks, crested caracaras, and Chihuahuan ravens (Merrill 1878, Bendire 1892, Strecker 1930). In desert habitats, nest availability is determined by the presence of species that build large nests, such as crows, kites, ravens, or hawks (USFWS 1990f). The breeding season lasts for 6 to 8 months, with most eggs laid between March and May. Clutches consist of two to three eggs, and the incubation period (both parents tending) lasts 32 days. Nestlings fledge after approximately 35 days, and remain within the vicinity of the nest for another month (Hector 1983).

The northern aplomado falcon was federally listed as endangered on February 25, 1986. Critical habitat has not been designated. At the time of listing, the falcon was no longer breeding in the U.S. Recently, however, there have been sightings of falcons in New Mexico, suggesting that the subspecies is dispersing from breeding locations in Mexico back into the southwest. In addition, falcons that were reintroduced to the Laguna Atacosa National Wildlife Refuge in Texas may disperse into other areas with suitable habitat. Originally subject to large population declines because of pesticides—especially DDT—applied in Mexico, the falcon has also lost large areas of suitable habitat through brush encroachment and agriculture clearing.

Effects of Vegetation Treatments on the Northern Aplomado Falcon

Effects Common to All Treatment Methods

Indirect Effects. The northern aplomado falcon is a desert grassland species that has lost large areas of suitable habitat through the encroachment of shrubs. Because it requires an open landscape, with scattered trees and an abundance of small- to medium-sized birds, this species will benefit from any treatment that removes shrubby plants from its habitat and helps to maintain open conditions for foraging. Use of vegetation treatments to restore desert grasslands could have a long-term beneficial effect by potentially increasing the acreage of suitable habitat and leading to the repopulation of historical habitats. In addition, fuels reduction treatment would likely benefit the species by decreasing the chance that a catastrophic wildfire would destroy existing habitat. Finally, any treatment that reduces the presence of non-native species should help to restore grassland structure and function, which would benefit not only falcons, but also their prey species.

Prescribed Fire Treatments

Direct Effects. Prescribed burning is not expected to cause direct mortality to adult falcons. Depending on the conditions of the burn and its distance from falcons, there could be some smoke disturbance. Smoke may temporarily obscure the landscape, interfering with foraging activities. A fire occurring during the breeding season could destroy young birds and/or eggs.

Indirect Effects. A prescribed fire could result in the destruction of raptor nests and nesting structures, such as mesquite trees and yuccas. Northern aplomado falcons are dependent on nesting structures built by other large bird species. Therefore, even the destruction of an old nest that was once used by another species could reduce the suitability of a habitat for the falcon.

A prescribed fire would likely have a short-term impact on the presence of prey species, such as ground- or shrub-nesting birds. Populations of these species would be expected to decrease as a result of fire, which could indirectly affect the northern aplomado falcon. Over the long-term, however, the removal of shrubs through burning would have a positive effect on the habitat of the falcon.

Mechanical Treatment Methods

Direct Effects. Machinery and personnel associated with mechanical treatments could cause auditory and visual disturbances to falcons. The risk for impacts would be greatest during the breeding season, when reproductive success could be affected.

Indirect Effects. Mechanical treatment methods could be very beneficial by removing a large amount of invading brush from falcon habitat, or from land that could be falcon habitat in the future. However, large-scale removal of vegetation from the site could also result in the obliteration of (or damage to) raptor nests and nesting structures, which would reduce the suitability of the site for falcons.

Over the short term, there could be minor impacts to falcon prey species, such as birds, small mammals and herpetofauna, which could be crushed by equipment.

Manual Treatment Methods

Direct and Indirect Effects. Vegetation treatments using manual methods would be unlikely to have high direct or indirect effects on aplomado falcons. There would be some minor disturbances associated with the presence of workers.

Biological Control Treatments

Domestic Animals

<u>Direct Effects</u>. The introduction of domestic animals into falcon habitat is unlikely to have direct effects on falcons, or to cause high amounts of disturbance.

<u>Indirect Effects</u>. Effects to falcon habitat resulting from weed containment using domestic animals would be dependent on the intensity of the treatment. Light to moderate controlled grazing could benefit falcon habitat by helping to halt the succession of shrubs onto desert grasslands. However, increased levels of grazing would have the opposite effect, as the continued removal of grass species would begin to encourage the invasion of shrubs into the habitat (USDI BLM 1996b). Over time, this increased shrub density would make the site less desirable for the falcon. Grazing might also encourage the spread of non-native species and increase erosion, which could degrade the quality of habitat for falcon prey species.

Other Biological Control Agents

Direct and Indirect Effects. The release of biological control agents into northern aplomado falcon habitat would be unlikely to have major direct or indirect effects on aplomado falcons. There is a chance that the release of a foreign organism or pathogen into the wild could have unanticipated effects to non-target organisms that would result in ecosystem-wide changes. However, such an occurrence is not reasonably foreseeable.

Herbicides

Direct Effects. The presence of workers and vehicles in aplomado falcon habitats could cause some minor, temporary disturbances to falcons. During the application of herbicides, most birds would be able to flee the area to avoid contact with the sprayed chemicals. However some falcons, including young flightless birds, might be unable to avoid such an inadvertent exposure. Based on the results of risk assessments, direct spray of birds by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, could potentially cause adverse health effects to northern aplomado falcons (see Table 6-2).

Immediately after an herbicide treatment, aplomado falcons could contact foliage or other plant materials that were exposed to herbicides. Via this exposure pathway, adverse health effects to falcons could occur if vegetation was sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate. According to the ERAs, ingestion of prey items sprayed by 2,4-D or diuron at the typical application rate, or by bromacil or diquat at the maximum application rate, could result in adverse effects to falcons (see Table 6-4). Since the ERA for hexazinone did not assess the potential risks to carnivorous species through ingestion of contaminated prey, the potential for adverse effects to falcons from exposure to hexazinone via this exposure pathway cannot be determined.

Indirect Effects. Herbicide treatments in aplomado falcon habitats would be expected to benefit the species by creating more open grassland conditions, which are conducive to finding prey in flight. If herbicide treatments were to reduce overall bird populations in falcon habitat, adverse effects to falcons could include reduced prey and reduced nesting opportunities. The conservation measures to protect falcon populations (listed below) should be protective of other bird populations on the treatment site.

Conservation Measures

The following conservation measures are the minimum steps required to protect the northern aplomado falcon from being adversely affected by the proposed vegetation treatments. Additional conservation measures would also be developed at the local level.

- Prior to conducting vegetation treatments, survey the project area for northern aplomado falcon nests.
- Do not burn or cut trees within ¼ mile of northern aplomado falcon nests.
- Avoid conducting vegetation treatments in northern aplomado falcon habitat during the nesting period.
- Avoid cutting mesquite trees, yuccas, and other trees that may support aplomado nests in the future.
- Do not use 2,4-D in northern aplomado falcon habitats; do not broadcast spray 2,4-D within ¼ mile of northern aplomado falcon habitat.
- Where feasible, avoid use of the following herbicides in northern aplomado falcon habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr.
- Avoid broadcast spraying herbicides in areas where future falcon nesting trees occur.
- Do not broadcast spray clopyralid, diuron, glyphosate, hexazinone, picloram, or triclopyr in northern aplomado falcon habitat; do not broadcast spray these herbicides in areas adjacent to northern aplomado falcon habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, diquat, imazapyr, or metsulfuron methyl in or adjacent to northern aplomado falcon habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in northern aplomado falcon habitat, utilize the typical, rather than the maximum, application rate.

Determination of Effects

Assuming that all vegetation treatments could be done anywhere on public lands, the proposed action would be **likely to adversely affect** the northern aplomado falcon. However, with the implementation of conservation measures to reduce or prevent these adverse effects (discussed in the previous section, Conservation Measures), vegetation treatments would likely have long-term positive effects on the species' habitat. Therefore, the effects determination could be reduced to **not likely to adversely affect** the northern aplomado falcon.

Yuma Clapper Rail

The Yuma clapper rail (*Rallus longirostris yumanensis*) is a subspecies of clapper rail that occurs in inland habitats in the southwestern United States. Yuma clapper rails are found in shallow, freshwater marshes containing dense stands of cattails and bulrushes, along the Colorado River from California and Arizona south into Mexico (California Department of Fish and Game 2000e). They also occur in dense, near-monotypic stands of cattail at the Salton Sea in Imperial County, California (USDI BLM 2001b). Unlike other clapper rails, which are associated with tidal marshes, the Yuma clapper rail occupies freshwater marshes during the breeding season. Until recently, most of the population was thought to retreat to Mexico during the winter; it is now estimated that over 70% of the breeding population winters along the Lower Colorado River (USFWS 1997i). The Yuma clapper rail feeds on crayfish and other crustaceans, and it is believed that the abundance of food animals at a particular site is a better predictor of rail population densities than is vegetation (USDI BLM 2001b).

Yuma clapper rails breed from March through July, with most eggs hatching during the first week of June. Nests are built in three major microhabitats: at the base of living clumps of cattail or bulrush, under wind thrown bulrush, or on the top of dead cattails remaining from the previous year's growth (USFWS 1997i). Nesting materials and cover are obtained from mature cattail/bulrush stands. Both adults care for eggs and young, and clutch size is typically six to eight eggs.

The Yuma clapper rail was federally listed as endangered on March 11, 1967. Critical habitat for this subspecies has not been designated. This subspecies is threatened by loss and degradation of habitat by activities such as water projects and draining or filling of marshes for development or agriculture (California Department of Fish and Game 2000e). Other threats to this species include catastrophic flooding, invasion of non-native plant species such as tamarisk, and pollution from urban runoff, industrial discharges, and sewage effluent. Although population numbers of the species appear to be stable, habitat throughout the species' range is not secure (USFWS 1997i).

Effects of Vegetation Treatments on the Yuma Clapper Rail

Effects Common to All Treatment Methods

Indirect Effects. Yuma clapper rails are associated primarily with dense marsh vegetation (USFWS 1997i). Therefore, any treatment method that reduces the cover of herbaceous vegetation in clapper rail habitats would be expected to negatively affect the species. However, activities that reduce the likelihood of wildfire and the coverage of non-native species in Yuma clapper rail habitat would benefit the species. Wildfire has been identified as a threat to Yuma clapper rail habitat, as the invasive species saltcedar and arrowweed tend to dominate post-fire recovery. These species exclude the dense marsh vegetation required by Yuma clapper rails, and reduce the suitability of wetland habitat for the species.

Prescribed Fire Treatments

Direct Effects. A prescribed fire could kill or injure Yuma clapper rails, with the greatest risks of injury during the breeding season. Although adult birds would be able to flee a fire, eggs or newly-hatched birds would be less mobile and less able to escape.

Indirect Effects. Apart from reducing cover and potentially destroying nests in clapper rail habitat, prescribed fire could facilitate the expansion of saltcedar and arrowweed into wetland habitats, reducing their suitability for clapper rails. A burn could also indirectly affect the Yuma clapper rail by temporarily altering aquatic habitats, where clapper rail prey items are found. The effects of prescribed fire on aquatic habitats are discussed in more detail in the effects discussion for aquatic organisms in Chapter 5. These effects would be localized and of short duration, and would not likely have a great effect on Yuma clapper rails, as they would be able to forage for food in other areas.

Mechanical Treatment Methods

Direct Effects. Equipment associated with mechanical treatments could crush eggs and destroy nests. However, it is unlikely that large equipment could be used directly in the wetland habitats that Yuma clapper rails occupy. Noise and personnel associated with these treatments could disturb breeding birds, and potentially interfere with reproductive success.

Indirect Effects. The use of large equipment in and near wetland habitats could result in some leakage of oil and other fuels into aquatic habitats that support Yuma clapper rail prey species. These effects are described in more detail in Chapter 5. These effects would be localized and of short duration, but there would be some risks of clapper rails foraging in contaminated waters.

As discussed under Effects Common to All Treatment Methods, large-scale removal of herbaceous vegetation would make habitat less suitable for Yuma clapper rails. However, removal of saltcedar and arrowweed, either directly from Yuma clapper rail habitats or from adjacent riparian habitats, would benefit the species.

Manual Treatment Methods

Direct and Indirect Effects. Use of manual treatments to control hazardous fuels and unwanted vegetation could result in some disturbance to Yuma clapper rails from the presence of humans and the use of loud equipment (e.g., chainsaws). Workers removing vegetation could disturb nests and flush hiding birds from protective cover. The presence of humans in nesting habitat could also temporarily interfere with breeding activities, causing stress to nesting birds. These effects would likely be of short duration.

Biological Control Treatments

Domestic Animals
<u>Direct Effects</u>. Domestic animals would be unlikely to harm or injure birds, nests, or, eggs, unless they were allowed to walk directly in the wetlands.

<u>Indirect Effects</u>. Domestic animals could affect Yuma clapper rail prey items by altering the aquatic habitats in which they occur. The feces of domestic animals can degrade water quality, and intensive grazing in riparian areas can alter water levels and channel widths, and increase sedimentation, all of which could negatively affect Yuma clapper rail habitat. For more information on the effects of treatments using domestic animals on aquatic habitats, see Chapter 5.

Other Biological Control Agents
<u>Direct and Indirect Effects</u>. There would be minor, short-term disturbances associated with the presence of humans in the area. The biological control agents themselves would not adversely affect Yuma clapper rails or their habitat, unless unforeseen effects were to result from their release.

Herbicides

Direct Effects. The presence of workers and vehicles associated with herbicide treatments would likely cause a temporary disturbance to Yuma clapper rails on site, which would cause minor behavioral modifications. Most birds would likely flee the site and so avoid direct exposure to herbicides during the treatment, but it is expected that some birds could be directly sprayed by herbicides. Based on the results of the ERAs, this direct spray of Yuma clapper rails by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, could potentially result in adverse health effects (see Table 6-2).

After an area was treated with herbicides, clapper rails could touch plant materials or ingest food items that had been contaminated by herbicides during the application. Indirect exposure through contact with foliage sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate, could potentially result in adverse health effects to clapper rails, according to the ERAs. In addition, ingestion of invertebrates sprayed by 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate, or to clopyralid or imazapyr at the maximum application rate, could potentially result in adverse health effects to Yuma clapper rails.

Indirect Effects. Use of herbicide treatments in Yuma clapper rail habitat could adversely affect the species by eliminating suitable nesting habitat and reducing the amount of vegetative cover available to the species. Effects would be greatest if treatments during the nesting season exposed nests, eggs, and/or newly-hatched birds. Over the long term, removal of non-native plant species such as tamarisk from clapper rail habitat would be expected to make treated areas more suitable for Yuma clapper rails.

Conservation Measures

To avoid adverse impacts to the species during treatments, the following programmatic level conservation measures are required:

- Conduct surveys prior to vegetation treatments within potential or suitable habitat.
- Where surveys detect birds, do not implement treatments during the breeding season.
- In habitats where Yuma clapper rails occur, follow the riparian/aquatic habitat protection measures discussed in Chapter 5.
- Closely follow all application instructions and use restrictions on herbicide labels; in wetland habitats use only those herbicides that are approved for use in wetlands.
- Do not use 2,4-D in Yuma clapper rail habitats; do not broadcast spray 2,4-D within ¼ mile of Yuma clapper rail habitat.
- Where feasible, avoid use of the following herbicides in Yuma clapper rail habitat: clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr.
- Do not broadcast spray clopyralid, diquat, diuron, glyphosate, hexazinone, picloram, or triclopyr in Yuma clapper rail habitat; do not broadcast spray these herbicides in areas adjacent to Yuma clapper rail habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying imazapyr or metsulfuron methyl in or adjacent to Yuma clapper rail habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in Yuma clapper rail habitat, utilize the typical, rather than the maximum, application rate.

Additional conservation measures would be identified at the local level, as necessary.

Determination of Effects

Assuming that all treatments proposed by the BLM could occur anywhere within Yuma clapper rail habitat on public lands, the proposed action would be **likely to adversely affect** Yuma clapper rails and/or their habitat. Following the guidance provided in the previous section, Conservation Measures, local offices would be able to implement suitable conservation measures at the local level that would result in a **not likely to adversely affect** determination for the Yuma clapper rail in BAs completed at the project level.

Sand Nesters: Western Snowy Plover, Piping Plover, Least Tern

Western Snowy Plover
The primary reference for this section is:
USFWS. 1993m. Determination of Threatened Status for the Pacific Coast Population of the Western Snowy Plover. Federal Register 58(42): 12864-12874.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Sacramento Field Office, Sacramento, California.

There are two distinct populations of western snowy plover (*Charadrius alexandrinus nivosus*), only one of which is a federally listed. The Pacific coast population of the western snowy plover, which is genetically isolated from interior-breeding western snowy plovers, is defined as those individuals that nest adjacent to or near tidal waters, including all nesting colonies on the mainland coast, peninsulas, offshore islands, adjacent bays, and estuaries. It is the Pacific coast population that is addressed in this document.

In the U.S., three breeding areas currently exist in southern Washington, and nesting birds have been recorded in nine locations in Oregon (USFWS 2001). In California, eight geographic areas support over three-quarters of the breeding population in that state: San Francisco Bay, Monterey Bay, Morro Bay, the Callendar-Mussel Rock Dunes area, the Point Sal to Point Conception area, the Oxnard lowland, Santa Rosa Island, and San Nicolas Island (Page et al. 1991).

The coastal population of the western snowy plover consists of both resident and migratory birds. Some birds winter in the same areas used for breeding, while other birds migrate either north or south to wintering areas (Warriner et al. 1986), the majority of which are south of Bodega Bay, California. Pacific coast western snowy plovers breed primarily on coastal beaches from southern Washington to Mexico. It is estimated that, at most, about 2,000 snowy plovers breed along the U.S. Pacific Coast (Page et al. 1995). Nest sites occur in flat, open areas with sandy or saline substrates, usually in areas where vegetation and driftwood are sparse or absent (Widrig 1980, Wilson 1980; Stenzel et al. 1981). Nesting habitat is unstable and ephemeral as a result of unconsolidated soil characteristics influenced by high winds, storms, wave action, and colonization by plants. Other, less common nesting habitats include salt pans, coastal dredged spoil disposal sites, dry salt ponds, and salt pond levees. Sand spits, dune-backed beaches, unvegetated beach stands, open areas around estuaries, and beaches at river mouths are the preferred habitats for nesting (Wilson 1980; Stenzel et al. 1981). Snowy plovers forage on invertebrates in the wet sand and amongst surf-cast kelp within the intertidal zone; in dry, sandy areas above the high tide; on salt pans; at spoil sites; and along the edges of salt marshes and salt ponds.

Snowy plovers breed in loose colonies that range in size from two to 318 adults. Based on concentrations of breeding birds along the coast, it is believed that the center of the plovers' coastal distribution lies close to the southern boundary of California (Page and Stenzel 1981). The breeding season of coastal western snowy plovers extends from mid-March through mid-September. The majority of snowy plovers are site-faithful, returning to the same breeding site each year, and often nesting in exactly the same locations. Nest initiation and egg laying occurs from mid-March through mid-July (Wilson 1980; Warriner et al. 1986). Typically, the clutch size is three eggs, and incubation averages 27 days, with both sexes incubating the eggs (Warriner et al. 1986).

The Pacific coast population of the snowy plover was federally listed as threatened on March 5, 1993. On December 7, 1999, the USFWS designated 28 areas along the coast of California, Oregon, and Washington (totaling approximately 18,000 acres and 180 miles of coastline) as critical habitat for this population segment. Declines in snowy plover populations have been attributed to poor reproductive success resulting from human disturbance, predation, and inclement weather, combined with habitat loss resulting from urban development and the encroachment of introduced European beachgrass. These factors continue to threaten existing coastal populations of this species.

Piping Plover

The primary reference for this section is:
USFWS. 2001l. Final Determination of Critical Habitat for Wintering Piping Plovers. Federal Register 66(132): 36037-36086.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS, Ecological Services Field Office, Corpus Christi, Texas.

The piping plover (*Charadrius melodus*) is a small North American shorebird. Piping plovers breed in three discrete areas of North America: The northern Great Plains, the Great Lakes, and the Atlantic Coast. There is only one breeding population in the project area: the northern Great Plains population. The northern Great Plains breeding range extends from southern Alberta, northern Saskatchewan, and southern Manitoba, south to eastern Montana, the Dakotas, southeastern Colorado, Iowa, Minnesota, and Nebraska, and east to north-central Minnesota. The majority of the U.S. pairs in this population are in the Dakotas, Nebraska, and Montana (USFWS 1994). Occasionally, Great Plains birds nest in Oklahoma and Kansas. Generally, piping plovers favor open sand, gravel, or cobble beaches for breeding. Breeding sites are generally found on islands, lakeshores, coastal shorelines, and river margins.

Piping plovers winter in coastal areas of the U.S. from North Carolina to Texas. They also winter along the coast of eastern Mexico and on Caribbean islands from Barbados to Cuba and the Bahamas (Haig 1992). Wintering habitats include beaches, mud flats, sand flats, algal flats, and washover passes (areas where breaks in the sand dunes result in an inlet).

Piping plovers begin arriving on the wintering grounds in July, with some late-nesting birds arriving in September. A few individuals can be found on the wintering grounds throughout the year, but sightings are rare in late May, June, and early July. Migration is poorly understood, but most piping plovers probably migrate non-stop from interior breeding areas to wintering grounds (Haig 1992). Most of the time on wintering grounds is spent foraging (Nicholls and Baldassarre 1990b; Drake 1999a, 1999b), which usually takes place on moist or wet sand, mud, or fine shell. In some cases, this substrate may be covered by a mat of blue-green algae. Primary prey includes polychaete marine worms, various crustaceans, insects, and occasionally bivalve mollusks (Nicholls 1989, Zonick and Ryan 1995). When not foraging, plovers can be found roosting, preening, bathing, in aggressive encounters (with other piping plovers and other species), and moving among available habitat locations (Zonick and Ryan 1996). Individual plovers tend to return to the same wintering sites year after year (Nicholls and Baldassarre 1990b, Drake 1999a). In late February, piping plovers begin leaving the wintering grounds to migrate back to breeding sites. Northward migration peaks in late March, and by late May most birds have left the wintering grounds (Eubanks 1994).

The population of piping plovers that breeds in the Great Lakes States is listed as endangered, while all other piping plovers are threatened species. All piping plovers are considered threatened species when on their wintering grounds. Critical habitat was designated for wintering populations on August 9, 2001, and includes 137 areas along the coasts of North Carolina, South Carolina, Georgia, Florida, Alabama, Mississippi, Louisiana, and Texas. This critical habitat includes approximately 1,800 miles of mapped shoreline and approximately 165,200 acres of mapped area along the Gulf and Atlantic coasts and along margins of interior bays, inlets, and lagoons. Critical habitat for the northern Great Plains breeding population was proposed on December 28, 2001, but has not yet been designated. The proposed designation includes 11 areas of prairie alkali wetlands and reservoir lakes in 5 counties in Montana, 18 counties in North Dakota, and 1 county at Lake-of-the-Woods, Minnesota, totaling approximately 196,576 acres. It also includes five areas on portions of four rivers in the States of Montana, North Dakota, South Dakota, and Nebraska, totaling approximately 1,338 miles of river.

Breeding census results show a marked decline of the population breeding in the northern Great Plains of the U.S. (Plissner and Haig 1997). Shoreline development, river flow alteration, channelization, and reservoir construction, have all resulted in the loss of plover breeding habitat. Overall winter habitat loss is difficult to document; however, a variety of human-caused disturbance factors have been noted that may affect plover survival or utilization of wintering habitat (Nicholls and Baldassarre 1990a, Haig and Plissner 1993). These factors include recreational activities (motorized and pedestrian), inlet and shoreline stabilization, dredging of inlets that can affect spit (a small point of land, especially sand, running into water) formation, beach maintenance and renourishment (renourishing the beach with sand that has been lost to erosion), and pollution (e.g., oil spills; USFWS 1996).

Least Tern (Interior)
The primary reference for this section is:
Lackey, J. 1997. The Interior Least Tern, an Endangered Species. NEBRASKAland Magazine and the Nebraska Game and Parks Commission. Lincoln, Nebraska.

The least tern (*Sterna antillarum*), the smallest member of the tern family, is represented by three distinct subspecies. The interior least tern (*Sterna a. athalassos*) breeds locally along the major tributaries of the Mississippi River drainage basin from eastern Montana south to Texas and east to western Illinois, Missouri, Arkansas and Louisiana. The interior least tern has distinct breeding and wintering areas. Most breeding occurs on interior rivers, and wintering is thought to occur on beaches along the Central American coast and along the northern coast of South America from Venezuela to northeastern Brazil. Wintering least terns have been reported in Guyana, El Salvador, and Guatemala. The occurrence of breeding least terns is localized and is highly dependent on the presence of dry, exposed sandbars and favorable river flows that support a forage fish supply and isolate the sandbars from the riverbanks. Characteristic riverine nesting sites are dry, flat, sparsely vegetated sandbars and gravel bars within a wide, unobstructed, water-filled river channel.

Interior least terns consume small fish captured in the shallow water of rivers and lakes. They hunt by hovering, searching and then diving from a height of a few feet to 30 feet above the surface to snatch small fish in their bill. Interior least terns nesting at sandpits and other off-river sites often fly up to 2 miles to forage at river sites. Interior least terns nesting on riverine sandbars usually forage close to the nesting colony. Fish of 1 to 3 inches long are consumed by adults.

Interior least terns usually arrive on their breeding grounds in early to mid-May and begin to establish feeding and nesting territories. During the breeding season, the terns' home range is generally limited to a 2-mile stretch of river associated with the nesting colony. Interior least terns nesting at sandpits along rivers use the adjoining river as well as the sandpit lake itself for foraging. Interior least terns are semi-colonial nesters that benefit from the anti-predator behavior exhibited by the entire colony when the nesting territory is invaded. The piping plover, a state and federally threatened shorebird species, is often found nesting in the midst of interior least tern colonies in Nebraska. Presumably the piping plover benefits from the defensive group behavior of the nesting terns as well.

Upon arrival on breeding grounds, interior least terns begin to engage in aerial courtship displays. During the ground phase of courtship, male terns offer small fish to females to help secure the pair bond. Courtship feeding is one of the most important parts of the courtship process and is continued through the incubation period. Nests are initiated only after spring and early summer flows recede and dry areas on sandbars are exposed, usually on higher elevations away from the water's edge. Artificially created nesting sites, such as sand and gravel pits, dredge islands, reservoir shorelines and power plant ash disposal areas, also are used.

Soon after pair formation, both sexes participate in making many shallow nest scrapes dispersed in open, gravelly or sandy areas. Although several scrapes might be built by each pair, only one is used for nesting. Nest scrapes are sometimes located near small pieces of wood or debris or near clusters of small stones. After the female selects a suitable scrape, two or three eggs are laid on consecutive days. Both adults begin to alternate incubation duties after the first egg is laid. If a first clutch of eggs is lost, interior least terns will renest up to two times, each renesting attempt taking place at a new site. Incubation lasts about 21 days, after which the eggs begin to hatch on consecutive days. The newly hatched young are weak and helpless and are continuously brooded by the adults during the first day. The nesting season ends by early August, and departure from breeding areas usually is complete by early September.

Following the breeding season, interior least terns gather in small flocks along rivers to feed and prepare for migration. In fall they probably follow the same migration routes that they use in spring, but their movements are less regular and more casual.

The interior least tern was federally listed as endangered on May 28, 1985. Critical habitat has not been designated. Loss of habitat has contributed to the decline of this species. River channelization, irrigation diversions, and the construction of dams have contributed to the destruction of much of the terns' sandbar nesting habitat. In addition, human-related disturbances (e.g., foot traffic, unleashed pets, swimmers, canoeists, and OHVs) can limit the reproductive success of this species.

Effects of Vegetation Treatments on the Western Snowy Plover, Piping Plover, and Least Tern

Effects Common to All Treatment Methods

All three of these species nest in sparsely vegetated, sandy habitats next to water, and require bare sand for nesting. In some places, the invasion of non-native beach grasses, or other vegetation (including native species) that encroaches onto suitable nesting areas has reduced the amount of available breeding habitat for these species. Although the natural disturbances that created habitat for these species were primarily flooding and other water-based disturbances, their net result was the removal of vegetation to expose bare sand. Therefore, any vegetation treatment method that removes invading plant species from beach/sandbar habitats would be expected to have a long-term positive effect on the western snowy plover, the piping plover, and the interior least tern.

Prescribed Fire Treatments

Direct Effects. A prescribed fire could directly affect shore birds, especially if it were to occur during the nesting season. Fire could destroy nests, eggs, and newborn chicks, which remain flightless for 20 to 30 days after hatching. However, since vegetation in areas that support these shore birds is sparse, prescribed fire is unlikely to be used as a treatment.

Indirect Effects. A prescribed fire could destroy the small amounts of cover that these species sometimes use to hide their nests from predators.

Mechanical Treatment Methods

Direct Effects. Heavy equipment and machinery used to remove vegetation in plover and tern habitats could crush nests, eggs, and newborn chicks.

Indirect Effects. The noise and human presence associated with mechanical treatments could severely impact the success of breeding, with the extent of this impact dependent on the scale and duration of the treatment. Disturbances to plovers and terns interfere with nesting, feeding, and roosting, all of which can reduce the success of the birds. These birds are highly susceptible to human interference, and if disturbed, may be chased off their nest, exposing eggs and chicks to environmental stresses and/or predators (USFWS 2001m). Mechanical control could also result in large-scale removal of vegetation, which could destroy vegetation used for cover from predators.

Manual Treatment Methods

Direct and Indirect Effects. Because of the high sensitivity of plovers and terns to human disturbances, the use of manual control during the breeding season would likely have some effect on bird populations. The presence of humans in breeding areas could cause birds to abandon their nests. In addition, since eggs and chicks are camouflaged, even careful workers may be unable to spot them, and could trample them.

Biological Treatment Methods

Domestic Animals

Direct and Indirect Effects. Domestic animals could trample nesting and brood-rearing habitat, destroy eggs, and disturb nesting birds. It is likely that animals released close to a water source would approach the water's edge to drink, and that these animals would therefore walk back and forth through plover and/or tern nesting habitat. The presence of herds of animals in shore bird habitat could also cause disturbances to nesting birds, potentially interfering with reproductive success. Disturbances can also prevent plovers from feeding and flush them from roost sites (USFWS 2001m).

Other Biological Control Agents

Direct and Indirect Effects. The release of biological control agents into plover habitats would likely entail the presence of humans in these areas, which could disturb birds (see above). These disturbances would be of short duration. The biological control agents themselves are unlikely to affect birds, as they target particular non-native species, and have a gradual effect on the vegetation. However, given the limited knowledge in the arena of biological control, there is still a chance that unforeseen effects to native species and the ecosystem in general could occur.

Herbicides

Direct Effects. The presence of workers and vehicles in plover or tern habitats during herbicide treatments would temporarily disturb some birds. If treatments were to occur near nesting birds, adverse effects to breeding success could occur. Although most birds would flee the area, some birds (particularly young, flightless birds) could inadvertently be exposed to direct spray of herbicides. Based on risks predicted by the ERAs for terrestrial vertebrate species (see Table 6-2), direct spray of birds by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, would potentially result in adverse effects to listed plovers or terns.

After an herbicide treatment program, plovers and/or terns in or near the treated area could be exposed to herbicides through contact with contaminated foliage. Via this exposure pathway, adverse health effects to birds could occur if vegetation was sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate. Birds could also be exposed to herbicides by consuming contaminated food items. In the case of the western snowy and piping plovers, food would include various aquatic invertebrates, and in the case of the least tern, food would include fish. According to the ERAs, exposure to herbicides by consumption of fish exposed to 2,4-D or hexazinone at the typical application rate would potentially result in adverse effects to birds (see Table 6-4). Birds that ingested aquatic invertebrates sprayed by 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate, or by clopyralid or imazapyr at the maximum application rate, could potentially experience adverse health effects.

Indirect Effects. Because the western snowy plover, piping plover, and interior least tern nest in open, sandy areas, vegetation removal through herbicide treatments would be unlikely to adversely affect the habitat of these species. Furthermore, treatments that control invasive plant species to maintain or recover the open conditions favored by these species could have a long-term positive effect by increasing the suitability of habitat.

Conservation Measures

The following conservation measures are the minimum steps required of the BLM to ensure that treatment methods would be unlikely to adversely affect TEP species.

- Survey for western snowy plovers, piping plovers, and interior least terns (and their nests) in suitable areas on proposed treatment areas, prior to developing treatment plans.
- Do not treat vegetation in nesting areas during the breeding season (as determined by a qualified biologist).
- Do not allow human (or domestic animal) disturbance within ¼ mile of nest sites during the nesting period.

- Ensure that nest sites are at least 1 mile from downwind smoke effects during the nesting period.
- Conduct beachgrass treatments during the plant's flowering stage, during periods of active growth.
- Closely follow all application instructions and use restrictions on herbicide labels; in wetland habitats use only those herbicides that are approved for use in wetlands.
- Do not use 2,4-D in western snowy plover, piping plover, or interior least tern habitats; do not broadcast spray 2,4-D within ¼ mile of western snowy plover, piping plover, or interior least tern habitat.
- Where feasible, avoid use of the following herbicides in western snowy plover and piping plover habitat: clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr; in interior least tern habitat avoid the use of clopyralid, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr.
- Do not broadcast spray clopyralid, diquat, diuron, glyphosate, hexazinone, picloram, or triclopyr in western snowy plover or piping plover habitat; do not broadcast spray these herbicides in areas adjacent to western snowy plover or piping plover habitat under conditions when spray drift onto the habitat is likely.
- Do not broadcast spray clopyralid, glyphosate, hexazinone, picloram, or triclopyr in interior least tern habitat; do not broadcast spray these herbicides in areas adjacent least tern habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying imazapyr or metsulfuron methyl in or adjacent to western snowy plover, piping plover, or interior least tern habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in western snowy plover, piping plover, or interior least tern habitat, utilize the typical, rather than the maximum, application rate.

Additional, project-specific conservation measures would be developed at the local level, as appropriate.

Determination of Effects

Assuming that any of the proposed vegetation treatments could occur anywhere on public lands, these treatments would be **likely to adversely affect** the western snowy plover, piping plover and/or their designated critical habitat listed in Tables 1-1; and the interior least tern. However, if the proper precautions were taken at the local level during the formulation of treatment programs, impacts to these species or their critical habitat could be avoided, reducing these effects to a **not likely to adversely affect** determination. General conservation measures for the species have been provided in the previous section, Conservation Measures.

Riparian Species: Least Bell's Vireo, Inyo California Towhee, Southwestern Willow Flycatcher

Least Bell's Vireo

The primary references for this section are:
USFWS. 1994k. Designation of Critical Habitat for the Least Bell's Vireo. Federal Register 59(22): 4845-4867;

and

USDI BLM. 2001c. Biological Evaluation on Effects of CDCA Plan as Amended and Proposed to be Amended by the NEMO and NECO Preferred Alternatives and with Other Interim Measures on Southwestern Willow Flycatcher, Least Bell's Vireo, Arroyo Southwestern Toad, and Coachella Valley Fringe-toed Lizard. BLM California Desert District, Riverside, California.

References cited in this section are internal to the above-referenced documents. Full citations have been included in the Bibliography.

The least Bell's vireo (*Vireo bellii pusillus*) is a subspecies of the Bell's vireo that occurs in riparian habitats in the southwestern U.S. and northwestern Mexico. This subspecies was once widespread and abundant throughout the Central Valley and other low elevation riverine areas of California. The least Bell's vireo historically bred in riparian woodlands from the interior of northern California (near Red Bluff, Tehama County) to northwestern Baja California, Mexico. Its current breeding distribution is restricted to a few localities in southern California and northwestern Mexico (Franzreb 1989). The least Bell's vireo winters primarily in Baja California, with occasional individuals remaining through the winter in cismontane southern California.

Least Bell's vireos nest primarily in willows, but also use a variety of other shrub and tree species for nesting (Gray and Greaves 1984, Salata 1987). Similar habitats are used by the vireos in winter months. They forage in riparian and adjoining upland habitats (Kus and Miner 1987, Salata 1987). Studies conducted along the Santa Ynez River and within the Mono Creek Basin (Santa Barbara County) indicated that a large percentage of their foraging may occur in the adjacent chaparral community up to 300 or more yards from the nest. Least Bell's vireos feed almost exclusively on arthropods, with insects and spiders comprising over 99% of their diet (Brown 1993).

The least Bell's vireo arrives on its breeding grounds in mid-March (Brown 1993), with males arriving slightly before females (Nolan 1960, Barlow 1962). Nesting takes place from early April through the end of July, and two broods are usually attempted during this period. Nests are suspended from forks in dense bushes or small trees – usually willows, although over 60 species of plants have been used for nest sites (Brown 1993). Most birds depart the nesting grounds by September, although some may remain until late November (Rosenberg et al. 1991).

The least Bell's vireo was federally listed as endangered on May 2, 1986. On February 2, 1994, about 38,000 acres of land in 10 localities of 6 counties in Southern California were designated as critical habitat. Included are areas along the Santa Ynez River in Santa Barbara County; the Santa Clara River in Ventura and LA counties; the Santa Ana River in San Bernardino and Riverside counties; and the Santa Margarite, San Luis Rey, San Diego, Sweetwater, and Tijuana rivers and Coyote and Jamul-Dulzura creeks in San Diego County. The reduction of least Bell's vireo numbers and distribution is associated with widespread loss of riparian habitats and brood parasitism by the brown-headed cowbird. The least Bell's vireo is threatened by loss and degradation of habitat by a number of factors, including agricultural, urban, and suburban development, flood control efforts, military activities, fires, OHV use, livestock activities, and the invasion of non-native plant species. Nest parasitism by the brown-headed cowbird can also have a huge negative impact on the breeding success of the subspecies (Goldwasser 1978, Beezley and Rieger 1987, Clark 1988).

Inyo California Towhee
The primary reference for this section is:
USFWS. 1998u. Recovery Plan for the Inyo California Towhee. Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Inyo California towhee (*Pipilo crissalis eremophilus*) is an isolated subspecies of the California towhee that is only found in riparian habitats in the southern Argus Range of Inyo County, California (Cord and Jehl 1979). The Inyo California towhee is a relict population of a species that was historically widespread in the southwestern U.S. and northern Mexico (Davis 1951). The subspecies became restricted to mountain areas on the northern Mojave Desert as a result of prehistoric climatic changes. Inyo California towhees are non-migratory, holding their territories year-round.

Inyo California towhees nest and forage in areas of dense riparian vegetation dominated by willows, Fremont cottonwood, and desert olive, with associated rubber rabbit brush and squaw waterweed. They also nest in shrubs of the upland community adjacent to riparian habitat, and use the upland habitat as their principal foraging grounds. This habitat consists of Mojave creosote bush scrub or Mojave mixed woody scrub. Plants associated with the creosote bush community include burro brush, allscale, and indigo bush. The mixed shrub community consists of a wide variety of plants, including antelope brush, green ephedra, Nevada ephedra, bush lupine, blackbrush, bush pea, big sagebrush, bladder sage, and brittlebush (LaBerteaux 1994).

Inyo California towhees are omnivorous, opportunistic feeders, foraging primarily in open, rocky and sandy desert hillsides on just about any seed or invertebrate they encounter. They will also forage on the low branches of large shrubs and in the leaf litter and foliage of dense riparian vegetation (LaBerteaux 1989). When foraging, towhees primarily peck and glean, but will also engage in scratching, flycatching, chasing, and harvesting to find or capture food.

Inyo California towhees mate for life, and only when one bird dies does the other pursue another mate. Sexual maturity is generally attained in the first breeding season after hatching. Initiation of nesting coincides with local plant growth and flowering periods, which are influenced by rainfall and temperature, factors that also affect insect abundance. Inyo California towhees nest in both riparian habitats and in a variety of desert shrubs in adjacent upland communities. Their nests are bulky cups made of thin twigs, grasses, and forb stems with leaves and flower heads. The nests are lined with fine stems, grasses, and hairs. Nests are constructed in a variety of plants, such as shining willow, arroyo willow, desert olive, antelope brush, bladder sage, four-winged saltbush, and green ephedra (Cord and Jehl 1979, LaBerteaux 1989). These plant types help provide nest sites off the ground that offer protection from ground predators, as well as dense canopies that hide nests from aerial predators. These trees also provide shade from extreme desert temperatures.

The breeding season generally starts early in spring, with courtship and nest building commencing in March. The first clutches are typically laid in April, although they may be laid as early as late March. Replacement clutches may be laid as late as May or early June. If the first clutch fails, the pair will recycle, but breeding behavior usually ceases for the pair when the first clutch is successful.

Clutch sizes range from two to four eggs. Only the female incubates the eggs, but both parents share in the brooding and feeding of the young. Eggs hatch after 14 days of incubation, and the young fledge 8 days after hatching. Parents continue to feed the young for at least 4 weeks after fledging. The young are fully independent of the parents at 6 weeks, but remain within their natal nest area through the following fall and winter (LaBerteaux 1989).

The Inyo California towhee was federally listed as threatened, and critical habitat was designated, on August 3, 1987. Critical habitat for the Inyo California towhee encompasses approximately 5,600 acres of habitat near springs, streambeds, and uplands in the following areas: Margaret Ann Springs, Snooky Spring, Ruby Spring, Quail Spring, Benko Spring, Bainter Spring, Indian Joe Spring, Great Falls Basin, Mountain Springs Canyon, Mumford Springs, Austin Springs, and three unnamed springs. The primary threat to the continued existence of this subspecies is the degradation and destruction of riparian habitat. These riparian habitats have been and continue to be threatened by the export of water, mining, recreational and military activities, rural development, controlled burns, and grazing.

Southwestern Willow Flycatcher

The primary reference for this section is:

USFWS. 1995e. Final Rule Determining Endangered Status for the Southwestern Willow Flycatcher. Federal Register 60(38): 10693-10715.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS, Ecological Services State Office, Phoenix, Arizona.

The southwestern willow flycatcher (*Empidonax traillii extimus*) is a subspecies of willow flycatcher that breeds in southern California, southern Nevada, southern Utah, Arizona, New Mexico, western Texas, and extreme northwest Mexico (Hubbard 1987, Unitt 1987, Browning 1993). It may also breed in southwestern Colorado, but nesting records are lacking. All willow flycatchers are migratory, most likely wintering in southern Mexico and Central America.

The southwestern willow flycatcher occurs in riparian habitats along rivers, streams, or other wetlands, where there are dense growths of willows, baccharis, cottonwood, buttonbush, and other deciduous shrubs and trees (Grinnell and Miller 1944; Phillips 1948; Phillips et al. 1964; Whitmore 1977; Hubbard 1987; Unit 1987; Brown and Trosset 1989; Whitfield 1990; Brown 1991; Sogge et al. 1993; Muiznieks et al. 1994). Throughout the range of the subspecies, these riparian habitats tend to be rare, small, and/or linear locales, widely separated by vast expanses of arid lands. Flycatchers nest in thickets of trees and shrubs approximately 13 to 23 feet or more in height, with dense foliage from approximately 13 feet above the ground, and often a high percentage of canopy cover. The diversity of nest site plant species may be low or comparatively high, and nest site vegetation may be even- or uneven-aged, but is usually dense and structurally homogeneous (Brown 1988; Whitfield 1990; Sogge et al. 1993; Muiznieks et al. 1994). Although the southwestern willow flycatcher historically nested in native plant communities, and still does so when such vegetation is available, the species is now known to nest in thickets dominated by the non-native species tamarisk and Russian olive (Hubbard 1987; Brown 1988; Sogge et al. 1993; Muiznieks et al. 1994).

The subspecies virtually always nests near surface water or saturated soil (Phillips et al. 1964; Muiznieks et al. 1994). At some nest sites surface water may be present early in the breeding season, but only damp soil is present by late June or early July (Griffith 1993, Whitfield 1993, Muiznieks 1994). Ultimately, a water table close enough to the surface to support riparian vegetation is necessary.

The southwestern willow flycatcher is an insectivore. It forages within and above dense riparian vegetation, taking insects on the wing or gleaning them from foliage (Wheelock 1912, Bent 1960). It also forages in areas adjacent to nest sites, which may be more open (Sogge 1993). No information is available on specific prey species.

Southwestern willow flycatchers arrive at breeding sites and begin singing by mid-May, and build nests in late May and early June. Birds construct nests in a fork or horizontal branch of a medium-sized bush or small tree, approximately 3.2 to 15 feet above the ground (Brown 1988; Whitfield 1990; Muiznieks et al. 1994). Typically, there is dense vegetation above and around the nest. The nest is a compact cup of fiber, bark, and grass, typically with feathers on the rim, lined with a layer of grass or some other fine, silky plant material (Harrison 1979). The southwestern willow flycatcher is present and singing on breeding territories by mid-May. The subspecies builds nests and lays eggs in late May and early June, and fledges young in early to mid-July (Willard 1912; Ligon 1961; Brown 1988; Whitfield 1990; Sogge and Tibbits 1992; Sogge et al. 1993; Muiznieks et al. 1994). Some variation in these dates has been observed (Carothers and Johnson 1975; Brown 1988; Muiznieks et al. 1994), and may be related to altitude, latitude, and renesting.

The southwestern willow flycatcher was federally-listed as endangered on February 27, 1995. On July 22, 1997, approximately 599 river miles of waterways and their adjacent riparian habitats in Arizona, California, and New Mexico were designated as critical habitat. Extensive loss of this subspecies' habitat has occurred through the conversion of floodplains to agriculture, flood-control projects, and urban developments. Other threats include overgrazing and brood-parasitism by the brown-headed cowbird. The subspecies is also highly sensitive to any disruptions to its breeding cycle, because its breeding season is among the shortest of any North American songbird.

Effects of Vegetation Treatments on the Least Bell's Vireo, Inyo California Towhee, and Southwestern Willow Flycatcher

Effects Common to All Treatment Methods

Direct Effects. Removal of vegetation for fuels reduction or weed control could directly affect riparian TEP bird species if nesting trees or shrubs were cut or burned.

Indirect Effects. Vegetation treatment methods could alter the species composition and structure of a riparian habitat, which could in turn affect its suitability for these bird species. Thinning of understory vegetation, for example, may reduce the suitability of a riparian site for nesting, as birds generally require dense vegetation above and around the nest for cover.

A treatment program that reduces invasive species, allowing natives (such as cottonwoods and willows) to increase in abundance, would be expected to have a long-term positive affect on riparian bird habitat. Fuels reduction treatments, which would potentially reduce the risk of future catastrophic wildfire, would also be likely to have a long-term positive effect on these three riparian-dwelling bird species. Furthermore, there would be less likelihood that toxic fire retardant/suppressant chemicals would need to be applied to the habitats of these birds.

Indirect effects to birds would also occur from the removal of vegetation, as seeds, berries, and other plant materials utilized as food could decrease in abundance. However, over the long term, effects of vegetation removal could be positive if the species composition of the area changed to favor species of greater food value to birds. Indirect effects could also occur if prey items, such as insects, were affected. In general, the larger the scale of vegetation removal, the greater the risks to riparian TEP bird populations.

Prescribed Fire Treatments

Direct Effects. A prescribed burn could cause mortality to TEP bird species. In most instances, adults would be able to escape fire, but nestlings and fledglings could be killed. In addition, nests and eggs could be destroyed and in some cases abandoned, resulting in reduced reproductive success. Smoke from prescribed fires could also cause bird mortality, particularly of young, by smoke inhalation or carbon monoxide poisoning. All of these impacts would be more likely and/or more severe if burning occurred during the breeding season.

Indirect Effects. Depending on the intensity of the fire, a large component of the brushy understory habitat on which the riparian birds rely could be destroyed. As a result, the suitability of the habitat for the bird species would be reduced over the short term, forcing birds to relocate. Over the long term, however, habitat suitability could be increased through increased plant diversity. Fire can stimulate the rejuvenation of early successional species, which may provide food or habitat conditions not found in later successional stages. Removal of non-native species would also likely be beneficial over the long-term, provided native species replaced them post fire.

Mechanical Treatment Methods

Direct Effects. The use of heavy equipment and machinery to carry out vegetation treatments could potentially kill or injure riparian TEP species, especially if equipment was used in nesting habitat. Although adults would be likely to escape through flight, nests could be destroyed and eggs or fledging birds could be harmed. The noise and human presence associated with mechanical treatments would also be expected to disturb birds. The severity of these effects (which could lower nesting success and productivity) would depend on their duration, their vicinity to nesting habitat, and the season.

Indirect Effects. Prolonged disturbances during the nesting period could cause birds to abandon nests, thus impacting their reproductive success. Use of some mechanical treatments may also disturb the soil enough to have a negative impact on soil-dwelling prey items, such as insects and earthworms.

Manual Treatment Methods

Direct and Indirect Effects. The use of manual control methods in riparian areas would be expected to have few effects on TEP bird species. During manual control, the presence of humans in the area could create enough of a disturbance to disrupt activities such as breeding or feeding. However, these effects should be temporary.

Biological Control Treatments

Domestic Animals

Direct Effects. Foraging domestic animals in and near riparian areas can harm or destroy nests, eggs, and nestlings. Domestic animals sometimes make physical contact with nests or supporting branches, resulting in destruction of nests and spillage of eggs and nestlings (USDA Forest Service 2002).

Indirect Effects. Use of domestic animals to contain weeds could alter riparian habitat, making it less suitable for the bird species considered in this section. Overuse by livestock has been a major factor in the degradation and modification of riparian habitats in the western U.S. (USDA Forest Service 2002). Grazing reduces the diversity and density of riparian plant species, especially cottonwoods and willows, which are often utilized as nesting trees by riparian bird species (USDI BLM 1996b). Cottonwood and willow seedlings may be grazed or trampled, thus reducing survival rates. Under heavier grazing treatments, established vegetation may be hedged to a height of 6 to 7 feet, resulting in a marked reduction in understory vegetation on which these bird species rely. It has been noted that most of the areas still known to support southwestern willow flycatchers have low to nonexistent levels of grazing by domestic animals (Suckling et al. 1992 *cited in* USDA Forest Service 2002).

Use of domestic animals to contain weeds may also indirectly affect habitat by improving conditions for nest parasitism by brown-headed cowbirds (Tibbits et al. 1994). Brown headed cowbirds prefer bare ground and open areas, conditions that can be created by extensive grazing. The southwestern willow flycatcher and least Bell's vireo are particularly susceptible to reduced reproductive success caused by parasitism by cowbirds.

Other Biological Control Agents

Direct and Indirect Effects. The release of biological control agents could cause disturbances associated with the temporary presence of humans in the area. These effects should be minimal and last for a very short time. Biological control agents, even those that have been tested and approved for release could cause future unanticipated impacts to birds or their habitat. However, such impacts are not expected to occur.

Herbicides

Direct Effects. The presence of workers and vehicles in habitats that support riparian TEP bird species would result in temporary disturbances to birds. The severity of these effects would depend on the season, and the vicinity of disturbances to nesting habitat. Although adult birds would be able to fly away from treatment sites, some birds could inadvertently be exposed to herbicides, as could nests, eggs, and young, flightless birds. According to the ERAs, direct spray by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, could potentially result in adverse health effects to riparian TEP bird species.

Since the bird species considered in this section occur in habitats with dense riparian vegetation, birds could be exposed to herbicides indirectly through contact with plants that have been sprayed. Via this exposure pathway, adverse health effects to birds could occur if vegetation was sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate.

Riparian TEP bird species could also consume food items that have been sprayed by herbicides. Based on the ERA results, ingestion of invertebrates sprayed by 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate, or by clopyralid or imazapyr at the maximum application rate, could potentially result in adverse health effects to birds. Since the Inyo California towhee also eats plant materials, adverse effects to this species could occur if plant food items were sprayed directly by 2,4-D, diquat, glyphosate, hexazinone, or triclopyr at the maximum application rate, or by bromacil, clopyralid, diuron, imazapyr, picloram, or tebuthiuron at the maximum application rate, However, this exposure scenario assumes that 100% of the bird's diet would come from contaminated plant material, which is highly unlikely.

Indirect Effects. As discussed under Effects Common to All Treatment Methods, herbicide treatments in habitats that support the least Bell's vireo, Inyo California towhee, or southwestern willow flycatcher could adversely affect these habitats if substantial loss of vegetation occurred. Effects would be greatest if vegetation around nests were injured or killed. These effects would likely be short-term in nature, unless older trees and shrubs were killed. The three bird species could also be adversely affected by a short-term reduction in food items, such as seeds and berries. Over the long term, a reduction in non-native plant species could benefit the least Bell's vireo, Inyo California towhee, and southwestern willow flycatcher by making habitat more suitable for these bird species.

Conservation Measures

To minimize or avoid adverse effects to the least Bell's vireo, Inyo California towhee, and southwestern willow flycatcher, the BLM would be required to implement the following programmatic-level conservation measures in habitats utilized by these three species.

- Conduct surveys prior to vegetation treatments within potential or suitable habitat.
- Where surveys detect birds, do not burn, broadcast spray herbicides, use domestic animals to control weeds, or conduct mechanical treatments.
- Do not conduct vegetation treatments within ½ mile (or further if deemed necessary to prevent smoke from inundating the nest area) of known nest sites or unsurveyed suitable habitat during the breeding season (as determined by a qualified wildlife biologist).
- Closely follow all application instructions and use restrictions on herbicide labels; in wetland habitats use only those herbicides that are approved for use in wetlands.
- Do not use 2,4-D in least Bell's vireo, Inyo California towhee, or southwestern willow flycatcher habitats; do not broadcast spray 2,4-D within ¼ mile of least Bell's vireo, Inyo California towhee, or southwestern willow flycatcher habitat.
- Where feasible, avoid use of the following herbicides in least Bell's vireo, Inyo California towhee, and southwestern willow flycatcher habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, tebuthiuron, and triclopyr.
- Do not broadcast spray clopyralid, diquat, diuron, glyphosate, hexazinone, picloram, or triclopyr in least Bell's vireo, or southwestern willow flycatcher habitat; do not broadcast spray these herbicides in areas adjacent to least Bell's vireo or southwestern willow flycatcher habitat under conditions when spray drift onto the habitat is likely.
- Do not broadcast spray clopyralid, diquat, glyphosate, hexazinone, picloram, or triclopyr in Inyo California towhee habitat; do not broadcast spray these herbicides in areas adjacent to Inyo California towhee habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying imazapyr or metsulfuron methyl in or adjacent to least Bell's vireo or southwestern willow flycatcher habitat, apply at the typical, rather than the maximum, application rate.
- If broadcast spraying bromacil, diuron, imazapyr, metsulfuron methyl, or tebuthiuron in or adjacent to Inyo California towhee habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in least Bell's vireo, Inyo California towhee, or southwestern willow flycatcher habitat, utilize the typical, rather than the maximum, application rate.

Additional, project-specific conservation measures would be developed at the local level to ensure protection of these species during treatment activities.

Determination of Effects

Assuming that all vegetation treatments could occur anywhere on public lands, including riparian areas, the proposed vegetation treatments would be **likely to adversely affect** the least Bell's vireo, Inyo California towhee, and southwestern willow flycatcher, and/or their designated critical habitat listed in Tables 1-1. In order to achieve a **not likely to adversely affect** determination, the BLM would be required to implement the programmatic-level conservation measures listed in the previous section, Conservation Measures, as well as any project-level conservation measures developed during project-specific BAs.

Coastal California Gnatcatcher

The primary reference for this section is:
USFWS. 2000k. Final Determination of Critical Habitat for the Coastal California Gnatcatcher. Federal Register 65(206): 63679-63743.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS, Carlsbad Fish and Wildlife Office, Carlsbad, California.

The coastal California gnatcatcher (*Pilioptila californica californica*), a subspecies of the California gnatcatcher, is a small, insectivorous songbird that typically occurs in various coastal sage scrub plant communities. These habitats are composed of relatively low-growing, dry-season deciduous, and succulent plants. Characteristic plant species include coastal sagebrush, various species of sage, California buckwheat, lemonadeberry, California encelia, and prickly pear and cholla cactus. The gnatcatcher exhibits a strong affinity to coastal sage scrub vegetation dominated by coastal sagebrush, although in some portions of its range (e.g., western Riverside County) other plant species may be more abundant. Sage scrub often occurs in a patchy, or mosaic, distribution pattern throughout the range of the subspecies. Gnatcatchers also use chaparral, grassland, and riparian habitats where they occur in proximity to sage scrub. Availability of these non-sage scrub areas may be essential during certain times of the year for dispersal, foraging, or nesting, particularly during drought conditions and following disturbance of habitat from fire.

The coastal California gnatcatcher is non-migratory and defends breeding territories ranging in size from 2 to 14 acres (Atwood 1990). Reported home ranges vary in size from 13 to 39 acres for this species (Mock and Jones 1990). The breeding season extends from late February through July, with the peak of nest initiations (startups) occurring from mid-March through mid-May. Nests are composed of grasses, bark strips, small leaves, spider webs, down, and other materials, and are often located in California sagebrush about 3 feet above the ground. Clutch size averages four eggs, and incubation and nestling periods encompass about 14 and 16 days, respectively. Both sexes participate in all phases of the nesting cycle. Juveniles are dependent upon, or remain closely associated with, their parents for up to several months following departure from the nest and dispersal from their place of birth territory. Dispersal of juveniles generally requires a corridor of native vegetation providing certain foraging and shelter requisites to link larger patches of appropriate sage scrub vegetation (Soule 1991).

This subspecies is restricted to coastal southern California and northwestern Baja California, Mexico, from Ventura and San Bernardino Counties, California, south to approximately El Rosario, Mexico, at about 30 degrees north latitude (American Ornithologists' Union 1957, Garrett and Dunn 1981, Atwood 1991, Banks and Gardner 1992). In the mid-1940s, the subspecies was considered locally common, but by the 1960s had apparently experienced a substantial population decline in the U.S. resulting from the widespread loss and fragmentation of its habitat. Recent taxonomic research has called into question the status of the coastal California gnatcatcher as a subspecies.

The coastal California gnatcatcher was federally listed as threatened on March 30, 1993. A total of approximately 513,650 acres in Los Angeles, Orange, Riverside, San Bernardino, and San Diego Counties, California were designated as critical habitat for the subspecies on October 24, 2000. The species remains threatened by habitat loss and fragmentation resulting from urban and agricultural development, and the synergistic effects of cowbird parasitism and predation.

Effects of Vegetation Treatments on the Coastal California Gnatcatcher

Effects Common to All Treatment Methods

Indirect Effects. Any treatment activity used to reduce the accumulation of fuels in coastal sage scrub and other associated plant communities (i.e., chaparral, grassland, and riparian areas) used by gnatcatchers would be expected to have long-term beneficial effects by reducing the risk of a future catastrophic fire. Because habitat for this species is small and fragmented, an uncontrolled wildfire could destroy enough habitat to have a severe impact on populations. In addition, there would be less likelihood of carrying out fire suppression activities, which can impact habitat and nesting birds (i.e., through the construction of firelines and application of fire retardants). Coastal sage scrub is a fire-prone habitat type, and much of it occurs at the wildland urban interface, where emergency fire suppression measures are a necessity to prevent loss of property. Treatment methods that reduce the coverage of non-native species would also be likely to have a beneficial effect on coastal California gnatcatchers by helping to return habitats to a more native condition. Non-native species, such as red brome, invade coastal sites and exclude the shrubs and native grasses found in coastal sage scrub habitat. Reduction of non-native species in areas that do not currently support coastal California gnatcatchers could also potentially benefit the species by increasing the amount of suitable habitat.

Given their very limited, fragmented habitat, coastal California gnatcatchers may be unable to disperse to other areas if a core habitat area is degraded. In their final determination of critical habitat for the coastal California gnatcatcher, the USFWS (2000k) identifies the following as activities that may directly or indirectly affect critical habitat: “removing, thinning, or destroying coastal California gnatcatcher habitat, whether by burning or mechanical, chemical, or other means (e.g., woodcutting, grubbing, grading, overgrazing, construction, road building, mining, herbicide application, etc.).” Therefore, there are also concerns with the use of the proposed treatment methods in coastal California gnatcatcher habitats; these concerns are described below.

Prescribed Fire Treatments

Direct Effects. Fire in coastal sage scrub and other plant communities used by the coastal California gnatcatcher could result in direct mortality to birds. As shrub-nesters that construct nests approximately 3 feet off the ground, fire could easily destroy eggs or chicks, as well as nests. In addition, smoke from fires could harm nesting birds or their young.

Indirect Effects. Coastal California gnatcatchers prefer coastal sage scrub that was burned 8 or 9 years previously, so there are some potential long-term benefits from the use of prescribed fire. However, in recent years, fire frequencies have been unnaturally high, and have destroyed habitat for the species. Habitat loss by burning directly affects the ability of an area to provide as much food, cover, and area for social spacing as it did previously (USFWS 2000k). Requiring substantial shrub cover (typically greater than 50%), coastal California gnatcatchers have been observed to avoid using burned areas for breeding purposes for a minimum of 4 to 5 years, and for as long as 12 years (Beyers and Wirtz 1997). Frequent fires also contribute to the competitive exclusion of native shrubs by exotic annual grasses and forbs.

The scale and intensity of a prescribed burn is very important. Disturbances that occur at the same scale as avian territory sizes and that occur within large, unbroken habitat areas may have no effect on the breeding densities of gnatcatchers, and may actually enhance local diversity. Coastal California Gnatcatchers tend to avoid dense and/or tall stands of coastal sage scrub. In addition, high fire intensities suppress resprouting of coastal sage shrubs, allowing the herb layer to become dominant, whereas less intense fires favor resprouting and lead to suppression of the herb layer (Westman 1981). Therefore, small, closely controlled fires at intervals that resemble the natural disturbance regime could benefit the species over the long term by rejuvenating habitat, provided suitable coastal California gnatcatcher habitat was also present in the area.

Mechanical Treatment Methods

Direct Effects. The use of mechanical equipment and machinery to control non-native species and reduce fuels could cause mortality to gnatcatchers. Equipment could rip up nest shrubs or cause other sorts of physical damage to nests, eggs, and young birds. In addition, the noise and human presence associated with the operation of machinery could disturb nesting birds and interfere with breeding activities.

Indirect Effects. Use of mechanical treatment methods over a large area could destroy enough coastal California gnatcatcher habitat to have a severe impact on the species population. Completing fuels reduction and weed removal activities outside of the critical habitat area for gnatcatchers could provide long-term benefits to the species by improving nearby habitat and potentially making it more suitable for occupation by gnatcatchers.

Manual Treatment Methods

Direct and Indirect Effects. Few direct effects would be expected from the use of manual methods to control weeds and complete other vegetation treatments. The scale of these activities would likely be small, and, unless nesting shrubs were destroyed they would have a minor physical impact on habitats. There could be some disturbance to nests, and the presence of humans could temporarily disrupt breeding activities.

Biological Control Treatments

Domestic Animals
Direct Effects. Introduction of domestic animals into coastal California gnatcatcher habitat would be unlikely to cause mortality or injury to adult birds, as the birds would be able to avoid the domestic animals. Nests occur in shrubs off of the ground, so the chances of trampling would be minimal. However, the domestic animals could make physical contact with nests or supporting branches, resulting in damage to nests and spillage of eggs and young birds onto the ground, where they could then be trampled.

Domestic animals could also disturb nesting birds and cause behavioral alterations that would potentially interfere with breeding success.

Indirect Effects. Use of domestic animals to control weeds could facilitate the spread of non-native species in coastal California gnatcatcher habitat by bringing in propagules from other sites. Where the propagules of non-native species are present, such as areas grazed by domestic animals, post-fire recovery may result in a site dominated by non-natives such as red brome rather than the sage scrub habitats required by coastal California gnatcatchers (O'Leary and Westman 1988).

Domestic animals are often associated with the presence of brown-headed cowbirds, nest parasites that reduce the reproductive success of coastal California gnatcatchers. Domestic animals also contribute to the spread of non-native species, which degrade coastal California gnatcatcher habitat.

Other Biological Control Agents
Direct and Indirect Effects. No direct or indirect adverse effects are expected from the use of biological control agents. These agents would target non-native species and would have a gradual effect on target plant populations. Given the limited information on the long-term effects of biological control agents, it is possible, though not reasonably foreseeable, that unanticipated adverse effects to gnatcatcher habitat could result from their release.

Herbicides

Direct Effects. Although most birds would be able to fly out of an area to avoid an herbicide application, some birds could be exposed to direct spray of herbicides inadvertently. Given the location of **coastal California** gnatcatcher nests (approximately 3 feet above the ground), young birds and eggs could also be sprayed during a treatment if nests were present in the area. Based on the results of the ERAs, direct spray by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, could potentially result in adverse health effects to coastal California gnatcatchers (see Table 6-2).

Coastal California gnatcatchers could also be exposed to herbicides through contact with sprayed foliage, or through ingestion of sprayed insects. According to the ERAs, adverse health effects to coastal California gnatcatchers could occur if birds came into contact with vegetation that was sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate. Ingestion of insects sprayed by 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate, or by clopyralid or imazapyr at the maximum application rate, could also cause adverse health effects to coastal California gnatcatchers (see Table 6-4).

Indirect Effects. Indiscriminate use of herbicides in coastal California gnatcatcher habitat, such that mortality of multiple plant species occurred over a large area, could result in a loss of key habitat components. Should equally or more suitable habitat not be available nearby, lasting population-level effects could occur, despite the temporary nature of the reduction in vegetative cover. Over the long-term, habitat could be made more suitable for coastal California gnatcatchers, and additional habitat for the species could potentially be created, by reducing the cover of non-native species through herbicide treatments.

Conservation Measures

In order to avoid or minimize potential effects to the coastal California gnatcatcher, the BLM would be required to implement, at a minimum, the programmatic-level conservation measures listed below.

- Prior to implementing vegetation treatments, survey areas in which treatments would occur for coastal California gnatcatchers.
- Where gnatcatchers occur, do not conduct treatments during the breeding season (as determined by a qualified wildlife biologist).
- Do not conduct treatments with domestic animals in habitats utilized by coastal California gnatcatchers, or in coastal sage scrub areas not dominated by non-native species.
- Ensure that prescribed burns and mechanical treatments are of minimal size and intensity, and do not affect greater than 30% of the coastal sage scrub habitat in a given area.
- Revegetate coastal sage habitats with native species.
- Do not broadcast spray herbicides in areas where coastal California gnatcatchers occur.
- Do not use 2,4-D in coastal California gnatcatcher habitats; do not broadcast spray 2,4-D within ¼ mile of coastal California gnatcatcher habitat.
- Where feasible, avoid use of the following herbicides in coastal California gnatcatcher habitat: clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr.
- Do not broadcast spray clopyralid, diquat, diuron, glyphosate, hexazinone, picloram, or triclopyr in areas adjacent to coastal California gnatcatcher habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying imazapyr or metsulfuron methyl in areas adjacent to coastal California gnatcatcher habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in coastal California gnatcatcher habitat, utilize the typical, rather than the maximum, application rate.

Additional conservation measures would be developed, as appropriate, during the preparation of project-level NEPA documents and management plans.

Determination of Effects

Assuming that all treatments could occur anywhere on public lands, including the coastal sage scrub habitats utilized by coastal California gnatcatchers, the proposed treatment program would be **likely to adversely affect** the species. In order to avoid or minimize these potential effects, the BLM would be required to implement the conservation measures listed in the previous section, Conservation Measures, as well as any conservation measures developed at the local level. Avoidance of these adverse effects would likely result in long-term beneficial effects for the coastal California gnatcatcher, as the risks of wildfire would be reduced. Thus, the effects determination for the proposed treatment program would be reduced to **not likely to adversely affect** the coastal California gnatcatcher.

Brown Pelican

The primary reference for this section is:
USFWS. No Date. Brown Pelican, (*Pelicanus occidentalis*). Available at http://species.fws.gov.

The brown pelican (*Pelicanus occidentalis*), also called the American brown pelican or common pelican, inhabits the Atlantic, Pacific, and Gulf coasts of North and South America. On the Atlantic Coast, pelicans can be found from Virginia south to the mouth of the Amazon River in Brazil; on the Pacific, they range from central California to south-central Chile and the Galapagos Islands; and on the Gulf of Mexico, they are found in Alabama, Louisiana, and Texas. Brown pelicans are rarely seen either inland or far out at sea.

Pelicans are primarily fish eaters, and require up to 4 pounds of fish a day. Their diet consists mainly of "rough" fish – species considered unimportant commercially. Examples of rough fish species are menhaden, herring, sheepshead, pigfish, mullet, grass minnows, topminnows, and silversides. Brown pelicans have also been known to eat some crustaceans, usually prawns. Brown pelicans have extremely keen eyesight. As they fly over the ocean, sometimes at heights of 60 to 70 feet, they can spot a school of small fish, or even a single fish. Diving steeply into the water, they may submerge completely or only partly, depending on the height of the dive, and come up with a mouthful of fish. Air sacs beneath the pelican's skin cushion the impact and help it surface.

Pelicans are social and gregarious. Males and females, juveniles and adults, congregate in large flocks for much of the year. The only breeding area in the western U.S. is in Channel Islands National Park in California. Pelicans nest in large colonies on the ground, in bushes, or in the tops of trees. On the ground, a nest may be a shallow depression lined with a few feathers and a rim of soil built up 4 to 10 inches above ground, or it may be a large mound of soil and debris with a cavity in the top. A treetop nest is built of reeds, grass, and straw heaped on a mound of sticks interwoven with the supporting tree branches. In most of the pelican's U.S. nesting range, peak egg-laying occurs in March and April. Two or three chalky white eggs hatch in approximately 1 month. Like many birds, newly hatched pelicans are blind, featherless, and completely dependent upon their parents. Average age at first flight is 75 days.

The brown pelican was federally listed as endangered on June 2, 1970. Critical habitat has not been designated. On February 4, 1985, brown pelican populations on the Atlantic Coast of the U.S. (including all of Florida and Alabama), had recovered to the point that the species could be removed from the Endangered Species List in that part of its range. The U.S. Gulf Coast population, which is still considered endangered, was recently estimated at nearly 6,000 breeding pairs. The brown pelican is also endangered in the Pacific Coast portion of its range, and in Central and South America. The southern California population of brown pelicans today is estimated at 4,500 to 5,000 breeding pairs. Brown pelicans have few natural enemies. Although ground nests are sometimes destroyed by hurricanes, flooding, or other natural disasters, the biggest threat to pelican survival comes from human activities. Pelican populations have been heavily affected by past hunting to protect commercial fishery resources, as well as the use of DDT and other pesticides. Current threats to the species include human development along the coast, abandoned fishing lines and tackle, and potential future oil spills.

Effects of Vegetation Treatments on the Brown Pelican

Effects Common to All Treatment Methods

Because the only breeding locations of the brown pelican in the western U.S. is in Channel Islands National Park, which is not managed by the BLM, treatments would only potentially affect wintering or "resting" habitat. Removal of vegetation, including non-native plant species, and hazardous fuels, would not be likely to have substantial effects on brown pelicans or their habitat in non-breeding areas.

Prescribed Fire Treatments

Prescribed burns would be unlikely to affect brown pelicans, as the birds would be able to move to other areas to avoid fires.

Mechanical Treatments

Mechanical treatments would be unlikely to affect brown pelicans, as the birds would be able to move to other areas to avoid workers and vehicles and other machinery.

Manual Treatments

Manual treatments would be unlikely to affect brown pelicans, as the birds would be able to move to other areas to avoid workers.

Biological Control Treatments

Domestic Animals. Use of domestic animals would be unlikely to affect brown pelicans, as the birds would be able to move to other areas to avoid these animals.

Other Biological Control Agents. The release of biological control agents would be unlikely to affect brown pelicans, as the birds would be able to move to other areas to avoid workers. Given the lack of knowledge about the long-term effects of biological control agents, unanticipated effects to habitats utilized by brown pelicans would be possible, though not reasonably foreseeable.

Herbicides

Direct Effects. The likelihood of a pelican being exposed to an herbicide treatment would be very low, since birds would be able to move out of the area to avoid them. Nonetheless, exposure of pelicans to herbicides could occur. In such a scenario, adverse health effects to brown pelicans could potentially occur if birds were directly sprayed by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate (see Table 6-2).

After an area was sprayed, pelicans could touch plant materials or ingest fish that were contaminated by herbicides during the treatments. Via the first exposure pathway, adverse health effects to pelicans could occur if birds came into contact with vegetation that was sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate. Furthermore, if pelicans were to consume fish from a water body into which 2,4-D or hexazinone was spilled, adverse health effects could potentially occur (see Table 6-4). Since pelicans typically consume fish from marine environments, which should not be subject to herbicide treatments, exposure to herbicides via this pathway would be unlikely.

Indirect Effects. Herbicide treatments in brown pelican wintering habitat would be unlikely to have indirect effects on pelicans.

Conservation Measures

Although treatment activities are unlikely to adversely affect the brown pelican or its habitat, extra steps could be taken by the BLM to ensure that herbicide treatments conducted in brown pelican wintering habitat did not result in adverse effects to the species:

- If feasible, conduct vegetation treatments in brown pelican wintering habitat outside the period when pelicans are likely to be present.

If herbicide treatments in brown pelican habitats must be conducted during the wintering period:

- Do not use 2,4-D in pelican wintering habitat.
- Prior to conducting herbicide treatments on pelican wintering habitat, survey the area for pelicans. Wait for pelicans to leave the area before spraying.
- Do not broadcast spray clopyralid, glyphosate, hexazinone, picloram, or triclopyr in pelican wintering habitats.
- If broadcast spraying imazapyr or metsulfuron methyl in pelican wintering habitats, use the typical rather than the maximum application rate.
- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in brown pelican wintering habitat, utilize the typical, rather than the maximum, application rate.

Determination of Effects

Assuming that all vegetation treatment activities could occur anywhere on public lands, including habitats utilized by the brown pelican, adverse effects to pelicans as a result of the proposed treatment programs are unlikely. However, should herbicide treatments occur on wintering habitat when pelicans are present, the proposed activities would be **likely to adversely affect** the brown pelican. In order to avoid or minimize these potential effects, the BLM would be required to implement the conservation measures listed above, such that the proposed treatment program would be **not likely to adversely affect** the brown pelican.

California Condor

The primary reference for this section is:
USFWS. 1996g. California Condor Recovery Plan, Third Revision. Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

At the time of the arrival of European man in western North America, the California condor (*Gymnogyps californianus*) occupied a narrow Pacific coastal strip from British Columbia, Canada to Baja California Norte, Mexico (Koford 1953, Wilbur 1978). Prior to the capture of the last free-flying, wild condor in 1987, the species used a wishbone-shaped area encompassing six counties just north of Los Angeles, California. Following several years of increasingly successful captive breeding, captive-produced condors were first released back to the wild in early 1992. The wishbone-shaped area remains an important habitat area, and has been designated as the range of primary concern by the California Condor Recovery Team.

California condors nest in various types of rock formations, including crevices, overhung ledges, and potholes, and, more rarely, in cavities of giant sequoia trees (Snyder et al. 1986). The factors influencing the choice of nest sites is poorly understood. Nest sites share the following characteristics: entrances large enough for the birds to fit through; a ceiling height of at least 15 inches at the egg position; fairly level floors with some loose surface substrate; unconstricted space for incubating adults; and short distance accessibility to a landing point.

California condors are opportunistic scavengers, feeding only on the carcasses of dead animals. Typical foraging behavior includes long-distance reconnaissance flights, lengthy circling flights over a carcass, and hours of waiting at a roost or on the ground near a carcass. Most foraging occurs in the open terrain of foothill grassland and oak savannah habitats. Steep terrain and brush interfere with foraging, and condors apparently do not locate food by olfactory cues (Stager 1964). It has been estimated that 95% of the condor’s diet once consisted of cattle, domestic sheep, ground squirrels, mule deer, and horses (Koford 1953).

Depending on weather conditions and the hunger of the bird, a California condor may spend most of its time perched at a roost. Cliffs and tall conifers, including snags, are generally used as roost sites in nesting areas. Birds often use traditional roosting sites near important foraging grounds (USFWS 1984h). Although most roost sites are near nesting or foraging areas, scattered roost sites are located throughout the range. While at a roost, birds devote considerable time to preening and other maintenance activities. Roosts may also serve some social function, as it is common for two or more birds to roost together and leave a roost together. California condors will tolerate more disturbance at a roost than at a nest, although the preservation of both requires isolation from human intrusion.

Based on observations of the wild condor population prior to 1987, courtship and nest site selection by breeding adults occurs from December through the spring months. Reproductively mature pairs normally lay a single egg between late January and early April. The egg is incubated by both parents and hatches after approximately 56 days. Both parents share responsibilities for feeding the nestling. At 2 to 3 months of age, condor chicks leave the actual nest cavity, but remain in the vicinity of the nest, where they are fed by their parents. The chick takes its first flight at about 6 to 7 months of age, but may not become fully independent of its parents until the following year. California condors often nest successfully only every other year (Koford 1953), although if the nestling fledges relatively early (in late summer or early fall), its parents may nest again in the following year (Snyder and Hamber 1985). Adults may lay a replacement clutch if their first (Harrison and Kiff 1980) or even their second egg is lost (Snyder and Hamber 1985).

The California condor was federally listed as endangered on March 11, 1967. Nine years later, the USFWS established critical habitat for the species on September 24, 1976. Despite decades of research, it is not known with certainty which mortality factors have been dominant in the overall decline of the species. However, there is evidence that two anthropogenic factors, lead poisoning and shooting, have contributed disproportionately to the decline of the species in recent years. In addition, thinning and ultrastructural abnormalities in eggshells, likely caused by the pesticide DDT, have resulted in reduced reproductive success in the species. The biggest threats to experimentally released populations appear to be collisions with power lines and other manmade objects, indicating that future releases should be conducted in areas remote from human settlements.

Effects of Vegetation Treatments on the California Condor

Effects Common to All Treatment Methods

Indirect Effects. Fuels reduction treatment activities would be expected to have a long-term positive effect on condors by reducing the risk of a future catastrophic wildfire that could harm birds, especially chicks, and eggs, and could burn the large trees and snags used by condors for roosting, perching, and foraging. Nesting trees (primarily giant sequoias) could also be burned, although trees are only used infrequently for this purpose. A reduced risk of future wildfire would also reduce the likelihood of future fire suppression activities that can disturb nesting condors and cause such impacts as nest abandonment or egg breakage by a disturbed adult.

Treatments that remove vegetation from young, dense forests stands would be expected to benefit condors over the long term by providing a more open, fire resilient stand (USDA Forest Service 2002). Condors require open conditions to search the surrounding area for food. A more open habitat would increase the quality of foraging, as condors prefer to forage on ridges and in areas with short vegetation so that they can easily locate prey and for facilitation of takeoff and approach (Verner 1978; Lowe et al. 1990).

Prescribed Fire Treatments

Direct Effects. A prescribed burn could cause mortality to eggs or to chicks as a result of burning, smoke inhalation, or stress. In particular, fall burning near nest sites would be expected to have some adverse effects on newly hatched condors (Nichols and Menke 1984). Adult birds would be able to avoid the burn site, and should not experience major direct effects.

Indirect Effects. A prescribed burn would remove understory vegetation, which could substantially change stand characteristics and have the potential to damage snags and trees that could be used in the future for nesting or roosting. However, only a large, severe fire would typically be capable of destroying the large trees used for roosting and nesting (Dodd 1986).

Prescribed fire could enhance the habitat of California condors by creating snags for future roost sites and improving foraging habitat. Condors occur in major fire-dependent plant associations in which fire exclusion has reduced the suitability of habitat by reducing openings and increasing shrub and/or tree cover (Tesky 1994).

Fire would also potentially modify the habitat of prey species, such as deer and small mammal species. These species typically receive some benefit from fires, although fires may have some adverse effects as well. Small mammals, in particular, typically increase in abundance following fires because of the availability of new palatable ground cover. Small mammals are an essential part of the California condor's diet because the condors obtain calcium from the bones, which are small enough to swallow.

Mechanical Treatment Methods

Direct Effects. The use of mechanical equipment to reduce fuels and remove weedy vegetation is unlikely to directly affect California condors, although the associated noise and other disturbances could have negative effects on birds, depending on the location, duration, and intensity of treatments. Noise disturbance could interfere with breeding by discouraging birds from nesting in otherwise suitable habitat, or by causing nest failure as a result of frequent long absences by adult birds. Agitated birds could also accidentally crush eggs as a result of disturbances from noisy equipment. Condors may be alarmed by loud noises or other human disturbances from distances of over 1 mile (Koford 1953). Noise can also disturb roosting condors, so disturbances late in the day could prevent nesting in that area at night (Tesky 1994). However, the short-term disturbances associated with mechanical treatments should not cause California condors to abandon regularly used roosts.

Indirect Effects. Like fire, mechanical control could remove understory vegetation and alter stand characteristics, as described above. Thinning treatments could remove future roosting trees, and snags, and future nesting trees. Mechanical control methods would also affect the habitat of prey species to some degree by removing plants used for forage, cover, and other needs.

Manual Treatment Methods

Direct Effects. Condors could be disturbed by human activities, with their reactions largely depending on the duration and intensity of the disturbance and whether condors were nesting, roosting, or foraging (USFWS 1996g). The largest potential effects would be to nesting birds, which may be discouraged from nesting in otherwise suitable habitat, or may experience nest failure as a result of frequent long absences.

Indirect Effects. Manual treatment methods would be unlikely to have major effects on California condor habitat. The scope of these treatments would likely be small and cause limited disturbance.

Biological Control Treatments

Domestic Animals

Direct Effects. The introduction of domestic animals into California condor habitat would be unlikely to have direct effects on condors.

Indirect Effects. Domestic animals could indirectly affect condors by altering their habitat or the habitat of their prey. Moderate amounts of grazing would help to keep understory vegetation short and the habitat open, which would increase the quality of foraging, as described above. Low vegetation would also be beneficial for most small mammals. Domestic animals could compete with deer for forage; however, a substantial reduction in the availability of carrion is not anticipated.

Other Biological Control Agents

Direct and Indirect Effects. Vegetation treatments using biological control agents would be unlikely to have major effects on condors or their habitat. These agents target particular invasive species and have a gradual effect on their hosts. Although the biological control agents approved for use have been tested and deemed safe for native species, there is still the risk that these agents could have unforeseen adverse effects to ecosystems in which they are released. Such an unforeseen occurrence, although very unlikely, could potentially affect condors or their habitat.

Herbicides

Direct Effects. Human presence and use of vehicles associated with herbicide treatments could disturb condors, causing behavioral modification. Although disturbance would be temporary, effects to breeding birds could be longer-lasting (i.e., decreased reproductive success). It is unlikely that a California condor would be sprayed during an herbicide application inadvertently, since condors would be able to flee the area, and typically nest and roost in cliffs and tall conifers. However, if direct spray by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, did occur, adverse health effects could potentially occur, according to the ERAs for these herbicides (see Table 6-2).

Condors would likely have minimal contact with foliage in sprayed areas, but such indirect exposure to herbicides could occur. According to the ERAs, adverse health effects could occur if birds came into contact with vegetation that was sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate.

Condors might consume carrion that was contaminated by herbicides. In such a scenario, adverse health effects to birds could occur if the prey item was sprayed directly by 2,4-D or diuron at the typical application rate, or by bromacil or diquat at the maximum application rate (see Table 6-4). Since the ERA for hexazinone did not assess the potential risks to carnivorous species through ingestion of contaminated prey, the potential for adverse effects to condors from exposure to hexazinone via this exposure pathway cannot be determined.

Indirect Effects. Use of herbicides to control vegetation in California condor habitat would be unlikely to have a negative effect on the quality of the habitat. Removal of non-native species could create more open habitat conditions, potentially allowing condors to forage more easily.

Conservation Measures

In order to avoid or minimize adverse effects to the California condor, the BLM would be required to implement the programmatic level conservation measures listed below.

- In areas where effects to breeding California condors may occur, do not burn until nesting is completed (Dodd 1986).
- Restrict human activity within 1.5 miles of California condor nest sites (Snyder et al. 1986).
- Do not use 2,4-D in California condor habitats; do not broadcast spray 2,4-D within ¼ mile of California condor habitat.
- Where feasible, avoid use of the following herbicides in California condor habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr.
- Do not broadcast spray clopyralid, diuron, glyphosate, hexazinone, picloram, or triclopyr in California condor habitat; do not broadcast spray these herbicides in areas adjacent to California condor habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, diquat, imazapyr, or metsulfuron methyl in or adjacent to California condor habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in California condor habitat, utilize the typical, rather than the maximum, application rate.

Additional conservation measures would be developed at the project level, as appropriate.

Determination of Effects

Assuming that all treatments could occur anywhere on public lands, including areas that support condors, the proposed vegetation treatment program would be **likely to adversely affect** California condors. However, with the conservation measures listed in the previous section in place, as well as any conservation measures deemed necessary at the project level, the proposed treatments would be likely to benefit California condors over the long term. If adverse effects were avoided or minimized, the effects determination at the local level would be **not likely to adversely affect** the California condor or its habitat.

Mature-forest Nesters: Marbled Murrelet, Northern Spotted Owl, Mexican Spotted Owl

Marbled Murrelet

The primary references for this section are:
USFWS. 1992j. Determination of Threatened Status for the Washington, Oregon, and California Population of the Marbled Murrelet. Federal Register 57(191): 45328-45337;

and

National Audubon Society. 2002a. Marbled murrelet (*Brachyramphus marmoratus*). Available at http://audubon2.org/webapp/watchlist/.

References cited in this section are internal to the above-referenced USFWS document. A complete list of these references is available from the USFWS Portland Field Office, Portland, Oregon.

The North American subspecies of marbled murrelet (*Brachyramphus marmoratus marmoratus*) is a small seabird found on the Pacific Coast of North America. Marbled murrelets are generally found in nearshore waters (within about 3 miles of shore) near their nesting sites. They nest in a narrow range along the Pacific, from the Aleutian Islands of Alaska south through British Columbia, Washington, and Oregon, to central California. The species generally occupies nesting areas on a year-round basis, although in certain places in Alaska and British Columbia, birds move to more protected waters during the winter. This species can also be found wintering south of its breeding range, along the coast of southern California to extreme northwestern Baja California. The states of California, Oregon, and Washington encompass roughly one-third of the geographic area occupied by this subspecies, comprising an important portion of its range. The amount of nesting habitat has undergone a tremendous decline since the late 1800s (most of which has taken place during the last 30 to 40 years), especially in the coastal areas of all three states. Therefore, the marbled murrelet is listed only in these three states, which together constitute a distinct population segment of the eastern Pacific subspecies.

Marbled murrelets feed primarily on fish and invertebrates in nearshore marine waters. During the summer, major food items include Pacific sand lance, northern anchovy, Pacific herring, and other small schooling fish, while during the winter, krill, amphipods, and herring are major prey items. Marbled murrelets usually forage alone, or in pairs, and are active in search of food both day and night. Although the majority of birds are found within or adjacent to the marine environment, there have been detections of marbled murrelets on rivers and inland lakes (Carter and Sealy 1986). Marbled murrelets spend the majority of their lives on the ocean, and come inland to nest, although they visit some inland stands during all months of the year. There are records of marbled murrelets up to 50 miles inland in Washington (Hamer and Cummins 1991), 35 miles inland in Oregon (Nelson 1990), 22 miles inland in northern California (Carter and Erickson 1988, Paton and Ralph 1990), and 11 miles inland in central California (Paton and Ralph 1990). However, the majority of detections were recorded closer to the coast. Marbled murrelets are semi-colonial in their nesting habits, and simultaneous detections of more than one bird are frequently made at inland sites. Nesting birds are often aggregated, with separate nests located close together.

Marbled murrelets do not reach sexual maturity until their second year, and adults have a variable reproductive rate (i.e., not all adults may nest every year). They produce one egg per nest, which the female lays on the limb of an old-growth conifer tree. Nesting occurs over an extended period from mid-April to late September (Carter and Sealy 1987). Incubation lasts about 30 days, and fledging takes another 28 days (Simons 1980; Hirsch et al. 1981). Both sexes incubate the egg in alternating 24-hour shifts (Simons 1980; Singer et al. 1991). Flights from ocean feeding areas to inland nest sites occur most often at dusk and dawn (Hamer and Cummins 1991). The adults feed the chick at least once per day, carrying one fish at a time (Carter and Sealy 1987; Hamer and Cummins 1991; Singer et al. 1992; Nelson 1992). Before leaving the nest, the young molt into a distinctive juvenile plumage. Fledglings appear to fly directly from the nest to the sea, rather than exploring the forest environment first (Hamer and Cummins 1991).

In California, Oregon, and Washington, marbled murrelets use older forest stands near the coastline for nesting. These forests are generally characterized by large trees (32 inches diameter at breast height or larger), a multi-storied stand, and a moderate to high canopy closure. In certain parts of the range, marbled murrelets are also known to use mature forests with an old-growth component. In order to provide suitable nest platforms, trees must have large branches or deformities (Binford et al. 1975; Carter and Sealy 1987; Hamer and Cummins 1990, 1991; Singer et al. 1991, 1992). Marbled murrelets tend to nest in the oldest trees in the stand. Observations of nests indicate that they tend to be located high above ground, usually with good overhead protection, in locations that allow easy access to the exterior of the forest. In Oregon and Washington, nests are located in stands dominated by Douglas-fir, and in California they are located in old-growth redwood stands.

In California, the species is restricted to old-growth redwood forests in Del Norte, Humboldt, San Mateo, and Santa Cruz Counties (Paton and Ralph 1988). In northwest Washington, marbled murrelets are mostly found at old-growth/mature sites (Hamer and Cummins 1990), and in Oregon, they occupy stands dominated by larger trees more often than those dominated by smaller trees (Nelson 1990). Large geographic gaps in offshore marbled murrelet numbers occur between central and northern California (a distance of 300 miles), and between Tillamook County, Oregon, and the Olympic Peninsula (a distance of about 120 miles), where nearly all older forest has been removed near the coast.

The marbled murrelet was federally listed as threatened in California, Oregon, and Washington on October 1, 1992. On May 24, 1996, 32 critical habitat units in Washington, Oregon, and California, encompassing approximately 3,887,800 acres of land, were designated for the species. Critical habitat areas focused on two primary constituent elements: individual trees with potential nesting platforms, and forested areas within 0.5 miles of these trees with a canopy height of at least one-half the site-potential tree height. The principal factor affecting the marbled murrelet in the three-state area, and the main cause of population decline has been the loss of older forests and associated nest sites. Older forests have declined throughout the range of the marbled murrelet as a result of commercial timber harvest, with additional losses from natural causes such as fire and windthrow. Most suitable nesting habitat on private lands within the range of the subspecies in Washington, Oregon, and California has been eliminated by timber harvest (Green 1985; Norse 1988; Thomas et al. 1990). Remaining tracts of potentially suitable habitat on private lands throughout the range are subject to continuing timber harvest operations. Mortality associated with oil spills and gill-net fisheries (in Washington) are lesser threats. It has been estimated that Marbled Murrelets are experiencing an annual population decline throughout their range as great as 4 to 7% per year. Surveys from Vancouver Island conducted 10 years apart suggest that populations there may have decreased by 40%. Populations in the northern Gulf of Alaska, meanwhile, may have declined by 50 to 73% over a 17- to 20-year period of time.

Northern Spotted Owl

The primary reference for this section is:
USFWS. 1990g. Determination of Threatened Status for the Northern Spotted Owl. Federal Register 55(123): 26114-26194.

References cited in this section are internal to the above-referenced document. A complete list of references is available from the USFWS, Fish and Wildlife Enhancement, Portland, Oregon.

The northern spotted owl (*Strix occidentalis caurina*) is one of three subspecies of the spotted owl, a nocturnal bird of forest habitats. The current range of the northern spotted owl is from southwestern British Columbia, through western Washington, western Oregon, and northern California south to San Francisco Bay. Throughout this present range, individuals are not evenly distributed. The majority of individuals are found in the Cascade Mountains of Oregon and the Klamath Mountains in southwestern Oregon and northwestern California (USDA 1989; Gould 1989). Evidently, northern spotted owls reach their highest population densities and may have their best reproductive success in suitable habitat in this part of their range (USDI 1987, 1989; Franklin and Gutierrez 1988; Miller and Meslow 1988; Franklin et al. 1989; Robertson 1989).

The northern spotted owl is known from most of the major types of coniferous forests in the Pacific Northwest (Gould 1974, 1975, 1979; Forsman et al. 1977, 1984; Garcia 1979; Marcot and Gardetto 1980; Solis 1983; Sisco and Gutierrez 1984; Gutierrez et al. 1984; Forsman and Meslow 1985). In California, northern spotted owls most commonly use the Douglas-fir and mixed conifer forest types (Marcot and Gardetto 1980, Soils 1983, Gutierrez 1985). In Washington’s coastal forests, the spotted owl is found in forests dominated by Douglas-fir and western hemlock. At higher elevations in western Washington, Pacific silver fir is commonly used by owls, whereas on the east side of the Cascades, Douglas-fir and grand fir are used (Postovit 1977). Extensive studies of spotted owls during the last 20 years have shown the species to be strongly associated with late-successional forests throughout much of its range.

Northern spotted owls have been observed over a wide range of elevations, although they seem to avoid higher elevation, subalpine forests (USDA 1986). The age of forests is not as important a factor in determining habitat suitability as are vegetational and structural components. Suitable owl habitat has moderate to high canopy closure (60 to 80%); a multi-layered, multi-species canopy dominated by large (> 30 inches diameter at breast height) overstory trees; a high incidence of large trees with various deformities (e.g., large cavities, broken tops, dwarf-mistletoe infections, and other evidence of decadence); numerous large snags; large accumulations of fallen trees and other woody debris on the ground; and sufficient open space below the canopy for owls to fly (Thomas et al. 1990). Usually, the features characteristic of owl habitat are most commonly associated with old-growth forests or mixed stands of old-growth and mature trees, which do not assimilate these attributes until 150 to 200 years of age.

Although a secretive and mostly nocturnal bird, the northern spotted owl is relatively unafraid of human beings (Bent 1938; Forsman et al. 1984; USDA 1986). The adult spotted owl maintains a territory year-round; however, individuals may shift their home ranges between the breeding and nonbreeding season. Northern spotted owls are perch-and-dive predators; over 50% of their prey items are arboreal or semi-arboreal species. They subsist on a variety of mammals, birds, reptiles, and insects, with small mammals (e.g., flying squirrels, red tree voles, and dusky-footed woodrats) making up the bulk of the food items throughout the range of the species (Solis and Gutierrez 1982; Forsman et al. 1984; Barrows 1985).

Monogamous and long-lived, northern spotted owls tend to mate for life. However, specific northern spotted owl pairs usually do not nest every year, nor are nesting pairs successful every year. Nesting behavior begins in February to March, with nesting occurring from March to June. The timing of nesting and fledging varies with latitude and elevation (Forsman et al. 1984). The number of eggs in a clutch ranges from one to four, with two eggs being most common. Fledging occurs from mid-May to late June, with parental care continuing into September. Females are capable of breeding in their second year, but it is likely that most do not breed until their third year (Barrows 1985; Miller and Meslow 1985b; Franklin et al. 1986). Males do most of the foraging during incubation, and assist with foraging during the fledging period.

The northern spotted owl was federally listed as a threatened species on June 26, 1990. On January 15, 1992, critical habitat was designated for the subspecies in 190 areas, encompassing a total of nearly 6.9 million acres of land. Throughout its range, the northern spotted owl is threatened by the loss and modification of suitable habitat as a result of timber harvesting. These threats are exacerbated by risks of catastrophic events such as fire, volcanic eruption, and wind storms. The population of the northern spotted owl is estimated at approximately 3,800 pairs and 1,000 individuals (National Audubon Society 2002).

Mexican Spotted Owl

The primary reference for this section is:
Agyagos, J., A. Telles, and R. Fletcher. 2001. Biological Assessment and Evaluation, Wildland Urban Interface Fuel Treatment. U.S. Department of Agriculture Forest Service Southwestern Region. Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Mexican spotted owl (*Strix occidentalis lucida*) occurs over a broad geographic range, from southern Utah and Colorado, south through the mountains of Arizona, New Mexico, and western Texas, and into the mountains of Mexico. The subspecies occurs in disjunct localities that correspond to isolated mountain systems and canyons. The range of the Mexican spotted owl in the U.S. has been divided into six recovery units (as identified in the recovery plan), with an additional five recovery units in Mexico. The U.S. recovery units, listed in decreasing order of number of known owls, are Upper Gila Mountain, Basin and Range-East, Basin and Range-West, Colorado Plateau, Southern Rocky Mountain-New Mexico, and Southern Rocky Mountain-Colorado.

Mexican spotted owls nest, roost, forage, and disperse in a diverse array of biotic communities. Nests and roosts are primarily found in closed-canopy forests or rocky canyons. In the northern portion of the range, most nests are in caves or on cliff ledges in steep-walled canyons. Elsewhere, the majority of nests appear to be in trees (Fletcher and Hollis 1994). Forests used for roosting and nesting often contain mature or old-growth stands that are structurally complex (Skaggs and Raitt 1988; Ganey and Balda 1989, 1994; McDonald et al. 1991). These forests are typically uneven-aged and multi-storied, with high canopy closure. Although a variety of tree species are used for nesting and roosting, Douglas-fir appears to be the most commonly utilized species for both of these activities (Fletcher and Hollis 1994).

Mexican spotted owls typically locate prey from an elevated perch by sight or sound, then pounce on the prey and capture it with their talons. In general, owls appear to forage more in unlogged forests than in selectively logged forests (Ganey and Balda 1994). Common prey items include species of rodent, bat, bird, reptile, and arthropod that use unique habitats. Thus it appears that diverse habitats for prey species provide owls with a diverse prey base.

Mexican spotted owls breed sporadically, but do not nest every year (Ganey 1998). Reproductive chronology varies somewhat across the range of the subspecies. Spotted owls observed in Arizona begin courtship and roosting in March, with eggs laid in either late March or early April. Incubation, which is performed exclusively by the female parent, begins shortly after the first egg is laid, and lasts for approximately 30 days. During incubation and the first half of the brooding period, the female leaves the nest only rarely (Forsman et al. 1984; Ganey 1998). Eggs hatch in early May, and young owls fledge 4 to 5 weeks after hatching, dispersing sometime between mid-September and early October.

The Mexican spotted owl was federally listed as a threatened species on April 15, 1993. On January 18, 2001, the USFWS designated 830,000 acres in Arizona, 525,000 acres in Colorado, 54,000 acres in New Mexico, and 3.2 million acres in Utah as critical habitat for the species. Primary threats to the subspecies are the continued alteration of habitat as a result of even-aged silvicultural practices, and the danger of catastrophic wildfire. Additional threats vary by Recovery Unit, and include such factors as indiscriminate fuelwood cutting, overgrazing, recreation, and fragmentation of habitat. There are estimated to be between 800 and 1,600 Mexican spotted owls in the southwestern U.S. (National Audubon Society 2002b).

Effects of Vegetation Treatments on the Marbled Murrelet, Northern Spotted Owl, and Mexican Spotted Owl

Effects Common to All Treatment Methods

Removal of vegetation could directly affect tree-nesting TEP bird species if nest trees were burned or cut. Treatment activities could also affect the species' habitat by altering its structure. For example, a reduction in snags, downed logs, woody debris, multi-storied canopies, dense canopy cover, or other key habitat components of these birds, could reduce the suitability of habitat, potentially resulting in the relocation of birds (Agyagos et al. 2001). For example, standing dead or down woody material, which is identified as a key habitat component of northern spotted and Mexican owls, could be removed during fuels reduction activities. Treatment activities that remove enough vegetation to increase the fragmentation of old-growth forests would negatively affect owls and murrelets, as all three species are associated with large old-growth stands.

Although treatments may have adverse effects on prey species and their habitat in the short term, the proposed treatments may increase the diversity of vegetative conditions, which would in turn provide for a diverse prey base. The prey of marbled murrelets, which feed on fish in marine habitats, would be unaffected by treatment activities.

In the absence of fuels reduction, an uncontrolled wildfire could have a large area of impact and could cause a great amount of damage to forest vegetation, not to mention forest birds and their nesting, roosting, and foraging habitats. Therefore, all treatment activities that reduce the amount of fuels that are present in forests in which northern or Mexican spotted owls, or marbled murrelets occur would likely have a long-term positive effect on these species. A reduction in the risk of future catastrophic fire would also reduce potential future needs for using toxic fire retardant/suppressant chemicals in habitats where these species occur.

Prescribed Fire

Direct Effects. A low-intensity understory burn would be unlikely to have direct effects on adult owls or murrelets, although there could be some effects from smoke inhalation. Such a fire would also be unlikely to seriously affect young birds, eggs, or nests, which should be in the upper forest layers, out of reach of the burn. A larger fire or an escaped fire would be expected to have a greater incidence of negative effects. Marbled murrelets would only be directly affected (potentially) by fires occurring during the breeding season, since they spend the majority of their time in marine environments.

Indirect Effects. The effects of fire on the habitat of these three species would depend largely on the intensity of the burn. A high intensity fire would likely result in a loss of key habitat components, such as snags and large trees that provide canopy cover. The USFWS (2002l) estimated that prescribed fire treatments proposed by the Forest Service in one area would result in the loss of 6.5% of trees greater than 24 inches diameter at breast height, a key habitat component of Mexican spotted owl habitat. Even a low-intensity prescribed fire can affect habitat by removing standing dead and down woody material, and eliminating the multi-storied canopy. Although overall habitat use tends to shift away from burned areas, spotted owls have been observed to continue to use areas of low intensity burn that maintain canopy cover (Bevis et al. 1997). The creation of a more open understory canopy can benefit spotted owls and murrelets by increasing the navigability of habitat.

Spotted owl prey items, such as woodrats and northern flying squirrels use habitat components that could be reduced by prescribed fire. Emaciated spotted owls, presumably malnourished from a lack of prey have been observed in recently-burned habitats (Bevis et al. 1997). Murrelet prey (fish) would be unaffected.

Mechanical Treatment Methods

Direct Effects. The use of heavy equipment and machinery in older forests would be unlikely to directly affect spotted owls or murrelets, unless nest trees were cut.

Indirect Effects. The noise associated with operation of the equipment could cause behavioral disturbances to owls, which could in turn prevent nesting or lead to nest failure. The use of heavy equipment could also crush or harm owl prey species, temporarily affecting food availability. Depending on the types and extent of vegetation removed, mechanical treatments could also negatively affect habitat by altering the multi-storied canopy.

Manual Treatment Methods

Direct and Indirect Effects. The use of manual control treatment methods in forested areas would be expected to have few effects on spotted owls or murrelets. There could be some disturbances associated with the presence of field crews, which could be large enough to disrupt activities such as breeding or feeding. However, these effects would likely be temporary.

Biological Control Treatments

Domestic Animals
Direct Effects. Use of domestic animals to control weeds in forested habitats used by murrelets and spotted owls would have few direct effects on birds. All of these species nest in tall, old trees that would be safely out of the way of domestic animals.

Indirect Effects. Heavy grazing could have long-term negative effects on habitat by preventing the replacement of existing old-growth habitat parameters that are necessary for/preferred by these species.

Other Biological Control Agents
Direct and Indirect Effects. The use of biological control agents to control non-native species in forested habitats would not be expected to have direct effects on spotted owls or murrelets. There could be minor disturbances associated with field crews releasing the agents, and follow-up monitoring, but these disturbances would be temporary. Unforeseen unspecified effects from biological control agents are possible but not reasonably foreseeable.

Herbicides

Direct Effects. Herbicide treatments would involve workers and the use of vehicles (trucks/ATVs) or aircraft, which could potentially disturb murrelets or spotted owls. Disturbance would be temporary, and effects would be greatest during the breeding season, when reproductive success could be reduced. While it is unlikely that murrelets or owls would be exposed to herbicides during treatments, it is conceivable that inadvertent direct exposure to herbicide spray could occur. According to the ERAs, such an exposure to 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, could potentially result in adverse health effects to murrelets or spotted owls (see Table 6-2).

Murrelets and spotted owls also could be exposed to herbicides by touching contaminated vegetation or ingesting contaminated prey. Contact with plant materials that have been sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate, could potentially result in adverse health effects to murrelets or owls, as shown in Table 6-2. Furthermore, ingestion of contaminated fish by murrelets could potential result in adverse health effects if the fish were exposed to spills of 2,4-D or hexazinone (see Table 6-4). Ingestion of prey sprayed by 2,4-D or diuron at the typical application rate, or by bromacil or diquat at the maximum application rate, by northern or Mexican spotted owls could potentially cause adverse health effects (see Table 6-4). Since the ERA for hexazinone did not assess the potential risks to carnivorous species through ingestion of contaminated prey, the potential for adverse effects to spotted owls from exposure to hexazinone via this exposure pathway cannot be determined.

Indirect Effects. Herbicide treatments should not have a substantial effect on spotted owl or murrelet habitat. Nesting trees would not be targeted during herbicide applications. Some alteration of the composition of lower canopy layers could occur, but key habitat components such as snags and woody debris would not be affected.

Conservation Measures

The following programmatic-level conservation measures are the minimum steps required of the BLM to ensure that treatment methods would be unlikely to adversely affect the marbled murrelet, northern spotted owl, or Mexican spotted owl.

- Survey for marbled murrelets, northern spotted owls, and Mexican spotted owls (and their nests) on suitable proposed treatment areas, prior to developing treatment plans.
- Delineate a 100-acre buffer around nests prior to mechanical treatments or prescribed burns.
- Do not allow human disturbance within ¼ mile of nest sites during the nesting period (as determined by a local biologist).
- Ensure that nest sites are at least 1 mile from downwind smoke effects during the nesting period.
- Protect and retain the structural components of known or suspected nest sites during treatments; evaluate each nest site prior to treatment and protect it in the most appropriate manner.
- Maintain sufficient dead and down material during treatments to support spotted owl prey species (minimums would depend on forest types, and should be determined by a wildlife biologist).
- Do not conduct treatments that alter forest structure in old-growth stands.
- Do not use 2,4-D in marbled murrelet, northern spotted owl, or Mexican spotted owl habitats; do not broadcast spray 2,4-D within ¼ mile of marbled murrelet, northern spotted owl, or Mexican spotted owl habitat.
- Where feasible, avoid use of the following herbicides in northern spotted owl and Mexican spotted owl habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr.
- Where feasible, avoid use of the following herbicides in marbled murrelet habitat: clopyralid, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr.
- Do not broadcast spray clopyralid, glyphosate, hexazinone, picloram, or triclopyr in marbled murrelet, northern spotted owl, or Mexican spotted owl habitat; do not broadcast spray these herbicides in areas adjacent to marbled murrelet, northern spotted owl, or Mexican spotted owl habitat under conditions when spray drift onto the habitat is likely.
- Do not broadcast spray diuron in Mexican or northern spotted owl habitat; do not broadcast spray these herbicides in areas adjacent to Mexican or northern spotted owl habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying imazapyr or metsulfuron methyl in or adjacent to marbled murrelet, northern spotted owl, or Mexican spotted owl habitat, apply at the typical, rather than the maximum, application rate.
- If broadcast spraying bromacil or diquat in or adjacent to Mexican or northern spotted owl habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in marbled murrelet, northern spotted owl, or Mexican spotted owl habitat, utilize the typical, rather than the maximum, application rate.
- Follow all instructions and SOPs to avoid spill and direct spray scenarios into aquatic habitats, particularly marine habitats where murrelets forage for prey.

Additional conservation measures would be developed, as necessary, at the project level to fine-tune protection of these species.

Determination of Effects

Because it is assumed that any of the proposed vegetation treatments could occur anywhere on public lands, these treatments would be **likely to adversely affect** the marbled murrelet, northern spotted owl and Mexican spotted owl, and/or their designated critical habitat listed in Tables 1-1. However, if the proper precautions were taken at the local level during the formulation of treatment programs, impacts to these species and their habitats could be avoided, resulting in a **not likely to adversely affect** determination in project-level BAs.

Whooping Crane

The primary reference for this section is:
USFWS. 1994l. Whooping Crane Recovery Plan. Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The whooping crane (*Grus americana*) is a wetland bird that currently exists in four wild populations and at captive breeding locations. The only self-sustaining wild population, called the AWP, nests in Alberta, Canada and winters along the Gulf of Mexico coast at Aransas National Wildlife Refuge, Texas, and adjacent areas. A second wild flock consists of individuals reared by wild sandhill cranes. These birds spend the summer in Idaho, western Wyoming, and southwestern Montana, and winter in the middle Rio Grande Valley of New Mexico. The third wild population consists of captive-reared birds released in the Kissimmee Prairie of Florida in 1993, and is considered a nonessential experimental population. In 2001, a fourth wild population was released in Necedah National Wildlife Refuge in Wisconsin (National Audubon Society 2002c). The current population estimates of whooping cranes in the western United States are 179 individuals in the AWP population (in December 2002); and about 10 individuals in sandhill crane flock. The total species population is 243 individuals.

Whooping cranes prefer sites with minimal human disturbance for nesting. The current nesting habitat for the main wild population in the Wood Buffalo National Park lies between the headwaters of the Nyarling, Sass, Klewi, and Little Buffalo Rivers. The area is poorly drained and interspersed with numerous potholes. Wetlands vary considerably in size, shape, and depth, and most possess soft bottoms. These wetlands are separated by narrow ridges that support an overstory of white spruce, black spruce, tamarack, and willows; and an understory of dwarf birch, Labrador tea, and bearberry. bulrush is the dominant emergent in the potholes used for nesting, although cattail, sedge, and other aquatic plants are common (Allen 1956; Novakowski 1965, 1966; Kuyt 1976a, 1976b, 1981a). Nest sites are located in the rushes or sedges of marshes, sloughs, or along lake margins (Bent 1926). It is believed that mollusks and frogs are important prey items for breeding adults and their offspring (Allen 1956).

The sandhill-reared flock, which was established at Grays Lake National Wildlife Refuge in southeastern Idaho in 1975, does not breed in the wild. This population summers in the vicinity of the Greater Yellowstone Ecosystem, which includes Yellowstone National Park; Grays Lake, Island Park, and Teton Basin in Idaho; Upper Green River basin in Wyoming; and the Centennial Valley in Montana. These whooping cranes winter with greater sandhill cranes in the Rio Grande area of south-central New Mexico.

Whooping cranes use a variety of habitats during migration (Howe 1987, 1989; Lingle 1987; Lingle et al. 1991). They have been observed feeding in a variety of croplands and roosting in marshy wetlands (Howe 1987, 1989). Whooping cranes also roost in riverine habitat, most notably the Platte River, Middle Loup River, and Niobrara River in Nebraska; the Cimarron River in Oklahoma; and the Red River in Texas. Cranes roost on submerged sandbars in wide unobstructed channels that are isolated from human disturbance (Armbruster 1990). Large palustrine wetlands are used for roosting and feeding during migration.

The principal wintering grounds (salt flats on Aransas National Wildlife Refuge and adjacent islands) consist of marshes dominated by salt grass, saltwort, smooth cordgrass, glasswort, and sea ox-eye. Inland margins of the flats are dominated by Gulf cordgrass. Interior portions of the refuge are gently rolling and sandy, and are characterized by oak brush, grassland, swales, and ponds. Typical plants include live oak, redbay, and bluestem (Stevenson and Griffith 1946, Allen 1952, Labuda and Butts 1979).

Whooping cranes are omnivorous (Walkinshaw 1973), probing the soil subsurface with their bills and taking foods from the soil surface or vegetation. Young chicks are fed by their parents, and gradually become more independent in their feeding until they separate from the parents preceding the next breeding season. Summer foods include large nymphal or larval forms of insects, frogs, rodents, small birds, minnows, and berries (Allen 1956, Novakowski 1966). Foods utilized during migration are poorly documented, but include frogs, fish, plant tubers, crayfish, insects, and waste grains in harvested fields. Animal foods and the plant wolfberry predominate in the winter diet. Most foraging occurs in brackish bays, marshes, and salt flats lying between the mainland and barrier islands.

Whooping cranes are monogamous, but will re-mate, sometimes within only a few weeks, following the death of their mate (Blankinship 1976, Stehn 1992). Most pairs return to the nesting area in late April, and begin nest construction and egg laying. Experienced pairs arrive first, show considerable fidelity to their breeding territories, and normally nest in the same general vicinity each year. From the initiation of laying until chicks are a few weeks of age, the activities of pairs and family groups are restricted to the breeding territory. Eggs (from one to three per clutch) are normally laid in late April to mid-May, and hatching occurs about 1 month later. The incubation period is from 29 to 31 days. Whooping cranes may re-nest if their first clutch is destroyed or lost before mid-incubation (Erickson and Derrickson 1981, Derrickson and Carpenter 1981, Kuyt 1981b). Whooping cranes generally nest annually, but occasional pairs skip a nesting season for no apparent reason. When nesting habitat conditions are unsuitable, some pairs do not attempt to nest. Autumn migration normally begins in mid-September, with most birds arriving on the wintering grounds between late October and mid-November. Occasionally, stragglers may not arrive until late December.

The whooping crane was listed as endangered, except where designated as an experimental population, on March 11, 1967. On May 15, 1978, critical habitat was designated for the species at nine sites in six states: Monte Vista National Wildlife Refuge, Colorado; Alamosa National Wildlife Refuge, Colorado; Grays Lake National Wildlife Refuge and vicinity, Idaho; Cheyenne Bottoms State Waterfowl Management Area, Kansas; the Platte River bottoms between Lexington and Dehman, Nebraska; Bosque del Apache National Wildlife Refuge, New Mexico; Salt Plains National Wildlife Refuge, Oklahoma; and Aransas National Wildlife Refuge and vicinity, Texas. It is thought that populations declined as a result of the destruction of wintering and breeding habitat, collisions with powerlines and fences, shooting, specimen collection, and human disturbance. Current threats are similar, and include the loss of wetlands, collisions, poaching, and poor reproductive success.

Effects of Vegetation Treatments on the Whooping Crane

Effects Common to All Treatment Methods

Indirect Effects. Whooping cranes in the project area commonly occur in wetlands and some agricultural fields along their migration route. Activities that reduce the cover of non-native plant species in resting and feeding areas along this route would be likely to have at least a minor positive effect on cranes by helping to restore/maintain the native qualities of these habitats. Treatment activities that reduce the accumulation of fuels would also benefit whooping crane habitat by reducing the likelihood that a severe wildfire would burn through key migration stopover areas, destroying habitat. A reduced likelihood of fire would also reduce the potential need for fire retardant/suppressant chemicals, toxic chemicals that could be released into crane habitats if fire suppression activities were required.

Cranes roost in standing water in wetlands to avoid terrestrial predators. However, they select sites without tall trees, dense vegetation, or other visual obstructions. Therefore, treatment activities that reduce the overall coverage of vegetation and make a site more open could have a positive effect on areas currently used by cranes, or could make areas not currently used more suitable for crane use in the future.

Removal of vegetation in breeding areas could have some negative effects on cranes by making eggs and chicks more susceptible to predation (USFWS 1980).

Prescribed Fire Treatments

Direct Effects. Prescribed fire used in crane habitats could destroy nests and harm molting adults and flightless chicks. Most cranes would be able to avoid fires, although the disturbance could have minor effects on foraging or roosting behavior. Lightning-caused fires have burned large portions of the nesting area during drought, but losses of eggs, chicks, or adults have not been confirmed (USFWS 1980).

Indirect Effects. Whooping cranes are attracted to burned uplands on their wintering grounds (Tesky 1993). Fire may be beneficial to cranes by removing dense or tall vegetation, thus making the area more accessible for whooping crane use, and by recycling nutrients. On upland wintering habitats, fires burn off dead grasses, making food items such as acorns very easy to obtain. Prescribed burns may also increase the abundance of certain prey items, such as rodents, and decrease the abundance of others, such as insects.

Mechanical Treatment Methods

Direct Effects. Mechanical treatments could destroy whooping crane nests or harm eggs, flightless young, or molting adults. Most birds, however, would be able to avoid areas where there was a human presence and work was taking place. However, such a disturbance could interfere with roosting and foraging activities.

Indirect Effects. Vehicles and equipment used directly in wetlands could cause habitat degradation. Whooping cranes tend to occur in remote, isolated areas that are not easily accessed by people. Bringing heavy equipment into crane habitats could increase the accessibility of these sites and potentially make them less suitable for use by cranes in the future.

Manual Treatment Methods

Direct and Indirect Effects. Manual vegetation treatment methods would be very unlikely to directly affect migrating cranes. A human presence in roosting and foraging sites would likely cause cranes to avoid these areas while work was occurring. Some negligible associated effects could occur.

Biological Control Treatments

Domestic Animals
<u>Direct Effects</u>. A herd of domestic animals could disturb whooping cranes, which would be likely to avoid areas where these animals were present. Such disturbances could interfere with roosting and foraging activities.

<u>Indirect Effects</u>. There could be indirect effects to whooping cranes caused by the degradation of wetland communities through the trampling of vegetation and the increased spread of non-native species.

Other Biological Control Agents
<u>Direct and Indirect Effects</u>. Use of biological control agents would be very unlikely to directly affect migrating cranes. There would be some disturbance associated with human presence (see Manual Methods above). Given the lack of knowledge about the long-term effects of biological control agents, future unanticipated effects of an unknown magnitude are possible, though extremely unlikely.

Herbicides

Direct Effects. The human presence and vehicles associated with herbicide treatments could disturb whooping cranes, particularly during roosting and foraging activities. In addition, use of trucks/ATVs in whooping crane habitat could destroy nests or harm eggs, flightless young, or molting adults. It is possible that some cranes could be exposed inadvertently to a direct spray of herbicides. Based on the results of the ERAs, such an exposure to 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or to imazapyr or metsulfuron methyl at the maximum application rate, could potentially result in adverse health effects to whooping cranes (see Table 6-2).

Whooping cranes also could be exposed to herbicides by coming into contact with contaminated vegetation or ingesting contaminated food. According to the ERAs, contact with foliage sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate could potentially result in adverse health effects to cranes. Under most scenarios, ingestion of fish that was exposed to herbicides would not result in adverse health effects (Table 6-4). However, in a scenario in which 2,4-D or hexazinone was spilled into an aquatic habitat, ingestion of contaminated fish by whooping cranes could potentially result in adverse health effects. Ingestion of insects directly sprayed by 2,4-D, diquat, diuron, glyphosate, hexazinone, or triclopyr at the typical application rate, or by clopyralid or imazapyr at the maximum application rate, could potentially result in adverse health effects to whooping cranes. Ingestion of plant materials sprayed by 2,4-D, diquat, glyphosate, hexazinone, or triclopyr at the typical application rate, or bromacil, clopyralid, diuron, imazapyr, picloram, or tebuthiuron at the maximum application rate, could potentially result in adverse health effects as well. Finally, ingestion of prey animals (such as frogs) that were exposed to 2,4-D or diuron at the typical application rate, or to bromacil or diquat at the maximum application rate, could potentially result in adverse health effects to whooping cranes. Since the ERA for hexazinone did not assess the potential risks to carnivorous species through ingestion of contaminated prey, the potential for adverse effects to whooping cranes from exposure to hexazinone via this exposure pathway cannot be determined. All of these risk predictions assume that 100% of the animal's diet consists of the type of food item in question.

Indirect Effects. Herbicide treatments in would likely have a positive effect on whooping crane habitat by reducing the coverage of weeds and promoting open conditions. However, if treatments were conducted during the breeding season, control of vegetation could make nests more visible to predators.

Conservation Measures

The following conservation measures are the minimum steps required of the BLM to ensure that treatment methods would not affect the whooping crane. Additional, site-specific conservation measures would also be developed at the local level, as appropriate.

- Burn whooping crane wintering grounds in late winter, when the food supply is low.
- Avoid prescribed fire activities in whooping crane breeding areas.
- Do not allow human disturbance within 1 mile occupied whooping crane habitat (nesting, roosting foraging) or potential nesting habitat where whooping cranes have been observed within the past 3 years during periods when cranes may be present (as determined by a qualified biologist).
- During prescribed burns, ensure that nest sites or occupied habitat are greater than 1 mile from downwind smoke effects during periods when cranes may be present.
- Do not conduct herbicide treatments in whooping crane habitat during the breeding season.
- Closely follow all application instructions and use restrictions on herbicide labels; in wetlands and riparian habitats use only those herbicides that are approved for use in those areas.
- Do not use 2,4-D in whooping crane habitats; do not broadcast spray 2,4-D within ¼ mile of whooping crane habitat.
- Where feasible, avoid use of the following herbicides in whooping crane habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, tebuthiuron, and triclopyr.
- Do not broadcast spray clopyralid, diquat, diuron, glyphosate, hexazinone, picloram, or triclopyr in whooping crane habitat; do not broadcast spray these herbicides in areas adjacent to whooping crane habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, imazapyr, or metsulfuron methyl in or adjacent to whooping crane habitat, apply at the typical, rather than the maximum, application rate.

- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in whooping crane habitat, utilize the typical, rather than the maximum, application rate.

Determination of Effects

Assuming that any of the proposed vegetation treatments could occur anywhere on public lands, these treatments would be **likely to adversely affect** the whooping crane. However, if the proper precautions were taken at the local level during the formulation of treatment programs, impacts to this species could be avoided. The previous section, Conservation Measures, lists the minimum steps required of the BLM to ensure that treatment methods would be **not likely to adversely affect** the whooping crane at the local level. Additional conservation measures would be added during project-level NEPA analysis, as appropriate.

Bald Eagle

The primary reference for this section is:
USDI BLM. 2001. Biological Evaluation on Effects of CDCA Plan as Amended and Proposed to be Amended by the NEMO and NECO Preferred Alternatives and with Other Interim Measures on Nine Threatened and Endangered Species. BLM California Desert District. Riverside, California;

and

USFWS. 1999k. Proposed Rule to Remove the Bald Eagle in the Lower 48 States from the List of Endangered and Threatened Wildlife. Federal Register 64(128): 36454-36463.

References cited in this section are internal to the above-referenced documents. Complete citations are included in the Bibliography.

The bald eagle (*Haliaeetus leucocephalus*) ranges throughout much of North America, nesting on both coasts, from Florida to Baja California, Mexico in the south, and from Labrador to the western Aleutian Islands, Alaska, in the north. Within this range, bald eagles are absent as breeding birds in most of the Great Basin, the prairie and plains region, and the eastern U.S. west of the Appalachian Mountains (American Ornithologists' Union 1983, Brown 1988). The bald eagle is a bird of aquatic ecosystems, frequenting estuaries, large lakes, major rivers, and some seacoast habitats. The species may also use prairies if adequate food is available. To support bald eagles, these areas must provide an adequate food base, perching areas near the shoreline, and suitable nesting sites.

Fish is the major component of the bald eagle's diet, but waterfowl, seagulls, and carrion are also eaten. In winter (defined as the non-nesting period), bald eagles often congregate at specific wintering sites that are close to open water and offer good perch trees and night roosts. Water bodies in winter foraging areas generally contain an abundance of shallow water fish or concentrations of waterfowl, providing eagles with easily catchable prey. Large concentrations of eagles are often observed at salmon spawning rivers.

Northern bald eagles winter in areas such as the Upper Mississippi River, and shorelines and river mouths in the Great Lakes area. Mid-continent bald eagles winter in the southern states; and southern bald eagles, who nest in the winter months, forage during the non-breeding season in areas such as Chesapeake Bay or Yellowstone National Park. In all cases, eagles seek wintering areas that offer an abundant and readily available food supply and suitable night roosts. Night roosts typically offer isolation and thermal protection from winds.

Perches, used during the daytime, are located on the water, within view of prey. Eagles may use a variety of trees or rocks for perching, and they may be located on or near the ground. Roosts provide another necessary habitat component for bald eagles. Roosts are chosen for their relative proximity to feeding sites, isolation from disturbance, darkness, and protection from wind (Johnsgard 1990). They are sometimes located against steep canyon walls, or in groves of the largest trees and protected from wind. Eagles appear to prefer trees with an open branched structure that facilitates landing. Communal roosts are common, containing from a few to dozens of birds.

Bald eagles usually nest in trees near the water, but are known to nest on cliffs and (rarely) on the ground. Nest sites are usually in large trees along shorelines, in relatively remote areas that are free of disturbance. Trees must be sturdy and open to support bald eagle nests, which are often 5 feet wide and 3 feet deep. The nesting season lasts about 6 months. Breeding times for bald eagles vary by elevation as well as latitude; mating occurs in late September through November in the South, in January through March in the Central States, and in late March to early April in Alaska. 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 nest(s). It is presumed that once bald eagles mate, the bond is long-term. Bald eagle pairs begin courtship about a month before egg-laying. Incubation lasts approximately 35 days, and fledging takes place at 11 to 12 weeks of age. As they leave their breeding areas, some bald eagles stay in the same general vicinity, but most migrate for several months and hundreds of miles to their wintering grounds.

The bald eagle was once federally listed as endangered in all of the lower 48 states (March 11, 1967), with the exception of Michigan, Minnesota, Wisconsin, Washington, and Oregon, where it was designated as threatened. It has since been reclassified as threatened in all states except Alaska, where eagles are not at risk, and are not protected under the ESA. Critical habitat has not been designated. The decline of bald eagles in most of the U.S. was caused by a combination of hunting, a decline in major prey species, and DDT usage. Since a recovery program for the species was established in the mid-1970s, the bald eagle population has increased in number and expanded in range. This improvement is attributable to the banning of DDT and other persistent organochlorides, habitat protection, and other recovery efforts. On July 6, 1999, the USFWS proposed to delist the bald eagle. Since the late 1970s, the species has doubled its breeding population every 6 to 7 years. However, bald eagles are still threatened by a number of factors, primarily human disturbances at nesting and wintering sites and activities that affect the food supply.

Effects of Vegetation Treatments on the Bald Eagle

Effects Common to All Treatment Methods

Indirect Effects. The removal of vegetation in bald eagle habitat could benefit the species by creating more open conditions. As a result, eagle sight distances would be increased, facilitating hunting conditions.

Any treatment that reduces the amount of hazardous fuels in or near bald eagle habitats would be expected to have a long-term positive effect on the species by reducing the risk of a future catastrophic wildfire. A stand-replacing fire, such as the sort that may be sustained with a large fuel load, could be capable of destroying nest trees, and could also destroy stands of trees used for roosting. There is evidence that stand-replacing fires, by changing the structure of a forest, can affect bald eagle use (National Park Service 1991). Fires that destroy old-growth forest can reduce eagle populations.

The removal of non-native vegetation would likely have at least a minor positive effect on eagle habitat by helping to restore native species. In addition, prey items such as waterfowl and fish may also experience long-term positive benefits from these activities. Over the short term, however, vegetation removal could alter aquatic habitats, negatively affecting aquatic prey, as described in Chapter 5.

Prescribed Fire Treatments

Direct Effects. Prescribed fire is unlikely to directly affect bald eagles, which will tend to avoid or flee a burn area. During the breeding season, however, the disturbance associated with prescribed burns may cause eagles to leave their nests, which could reduce reproductive success. An intense fire would run the risk of burning nests, and during the breeding season could destroy eggs or kill young chicks that are unable to fly. Smoke from burns could also affect eagles by creating a visual disturbance to foraging eagles.

Indirect Effects. Prescribed low-intensity fires in forests or stands of trees that support eagles can have a positive effect on bald eagle habitat by reducing litter build-up, controlling disease, removing less vigorous species, and allowing more vigorous trees to reach maturity, thus providing more suitable habitat for bald eagles (Harrington and Sackett 1992). Fire also creates snags, which are important perching and nesting sites for bald eagles. However, snags can potentially increase the likelihood of a lightning-caused fire when standing, and can increase fuel loading when fallen (Lyon 1977).

As described above, more intense fires can have a negative effect on eagle habitat by destroying nesting and roosting trees. Although prescribed fires would be aimed at thinning understory stands, rather than the large trees in which eagles nest and roost, burning can still alter bald eagle habitat by changing the stand characteristics and damaging potential future nest trees. In addition, fire could also destroy snags that are used as perches by eagles.

There could be some short-term effects on bald eagle prey from fire. As described in Chapter 5, fire could heat water to lethal temperatures, and cause negative chemical changes capable of harming aquatic species. However, these affects to prey would be short-term and localized, and should not extensively affect eagles, which would be able to forage in other areas or consume other types of prey. See sections on aquatic species for effects on fish and waterfowl populations.

Mechanical Treatment Methods

Direct Effects. Mechanical methods of vegetation control are unlikely to directly affect eagles, except through noise and visual disturbance. The disturbance to flying or foraging eagles would be localized and of short duration and low intensity, and would not affect the overall distribution of the species (Agyagos et al. 2001). During the breeding season, however, noise/machinery disturbances could have more substantial negative effects by causing nesting eagles to leave nearby nests temporarily. Repeated entries into nest areas would likely cause the greatest harm.

Indirect Effects. Eagles prefer to roost in trees within proximity to other large trees. Therefore, fuels reduction treatments involving thinning of trees could affect nesting habitat if such treatments were to modify clumps of suitable roost trees (Agyagos et al. 2001). There could also be some positive effects of thinning vegetation in eagle habitats, as treatments would likely increase sight distances, improving hunting conditions for eagles.

As discussed in Chapter 5, mechanical treatments could have short-term negative effects on bald eagle prey sources by causing erosion and sedimentation, and potentially through the leakage of oil and other fuels into the water. These effects on prey could affect bald eagles in turn. In most cases, however, bald eagles would be able to temporarily forage in other areas for food.

Manual Treatment Methods

Direct and Indirect Effects. No major effects are expected to occur from the use of manual treatment methods in or near eagle habitats. The disturbance associated with these activities would be minimal, though the likelihood of affecting eagles would increase the closer to nesting sites the activities were, and the longer they lasted.

Biological Control Treatments

Domestic Animals

Direct Effects. The use of domestic animals in eagle habitat would be unlikely to directly affect bald eagles, as birds are highly mobile, and generally nest and roost out of the reach of these animals.

Indirect Effects. Indirect effects through the alteration of habitat are possible. Intensive use of domestic animals to contain weeds in riparian areas or other eagle habitats can inhibit the replenishment or establishment of large tree species (such as cottonwoods) that eagles use for roosting, perching, or nesting (USDI BLM 1996b). Domestic animals can also alter prey abundance by modifying the plant cover or species composition of the grazed area. In addition, overgrazing can cause erosion and siltation into streams in which the eagles' main prey species are found (see fish and waterfowl sections for more information on the effects of grazing on prey species). More controlled, less intensive grazing techniques would have less impact on eagle habitats.

Other Biological Control Agents
Direct and Indirect Effects. The use of biological agents to control undesirable species in eagle habitat is unlikely to affect eagles. Biological control agents target specific weeds and have a gradual effect on these plant populations. However, there is always a small risk for unforeseen impacts associated with the release of these agents.

Herbicides

Direct Effects. Human presence and use of vehicles associated with herbicide applications in eagle habitats would create a temporary disturbance. Outside the breeding season, the disturbance would be minor. During the breeding season, however, the disturbance could cause eagles to leave nests temporarily, potentially reducing reproductive success. It is unlikely that bald eagles or their nests would be sprayed by herbicides inadvertently. Nonetheless, if direct spray of bald eagles by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, did occur, it could potentially result in adverse health effects to eagles, according to the ERAs (Table 6-2).

Bald eagles also could be exposed to herbicides through contact with contaminated vegetation, or by ingesting contaminated prey items. Based on the results of the ERAs, contact with foliage that was directly sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate, could potentially result in adverse health effects to eagles. Under most scenarios, ingestion of fish contaminated by herbicides would not result in adverse health effects to eagles (Table 6-4). However, under a scenario in which eagles ingested contaminated fish from a water body after a spill of 2,4-D or hexazinone, adverse health effects could occur. Since fish is a major component of the bald eagle's diet, the risk of indirect effects from other herbicides is low. However, since bald eagles may also eat other types of animals, there could also be some risks to the species from 2,4-D, bromacil, diquat, and diuron. Based on an ERA scenario in which 100% of the animal's diet consisted of contaminated non-fish prey items, adverse health effects to eagles could occur if the prey items were exposed to 2,4-D or diuron at the typical application rate, or to bromacil or diquat at the maximum application rate. Since the ERA for hexazinone did not assess the potential risks to carnivorous species through ingestion of contaminated prey, the potential for adverse health effects to eagles from exposure to hexazinone via this exposure pathway cannot be determined.

Indirect Effects. Herbicide treatments could have a minor effect on bald eagle habitat by minimizing the coverage of non-native species, and potentially creating more open conditions. More open habitat conditions could benefit bald eagles by making it easier for eagles to spot prey. Large nesting trees utilized by eagles would not be targeted by treatment programs. It is possible that populations of some prey species would be reduced as a result of herbicide treatments, but these effects would be temporary, and should not have a substantial effect on bald eagles' ability to find food.

Conservation Measures

The following programmatic level conservation measures are the minimum steps required of the BLM to ensure that treatment methods would not adversely affect the bald eagle or its habitat. Additional, site-specific conservation measures would also be developed at the local level, as appropriate.

- Do not allow human disturbance within ½ mile of known bald eagle nest sites during the breeding season (as determined by a qualified wildlife biologist).
- Avoid human disturbance within ¼ mile of a winter roost during the wintering period (as determined by a qualified wildlife biologist).

- Complete treatment activities that must occur within ¼ mile of a winter roost within the hours of 9 a.m. to 3 p.m., during the winter roosting period.
- Do not allow helicopter/aircraft activity within 1 mile of bald eagle nest sites or winter roost sites during the breeding or roosting period.
- Conduct prescribed burn activities in a manner that ensures that nest and winter roost sites are greater than 1 mile from downwind smoke effects.
- Do not cut trees within ¼ mile of any known nest trees.
- Do not use 2,4-D in bald eagle habitats; do not broadcast spray 2,4-D within ¼ mile of bald eagle habitat.
- Where feasible, avoid use of the following herbicides in bald eagle habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr.
- Do not broadcast spray clopyralid, diuron, glyphosate, hexazinone, picloram, or triclopyr in bald eagle habitat; do not broadcast spray these herbicides in areas adjacent to bald eagle habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, diquat, imazapyr, or metsulfuron methyl in or adjacent to bald eagle habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in bald eagle habitat, utilize the typical, rather than the maximum, application rate.

Summary of Effects

With the assumption that any of the proposed vegetation treatments could occur anywhere on public lands, these treatments would be **likely to adversely affect** bald eagle populations and/or their designated critical habitat listed in Tables 1-1. However, if the proper precautions were taken at the local level during the formulation of treatment programs, impacts to this species could be avoided. Following the conservation measures listed in the previous section, as well as any project-specific conservation measures deemed appropriate at the local level would be likely to result in a **not likely to adversely affect** determination for the bald eagle.

Mammals

The following mammals, and the ecoregion they are typically found in, are considered in this BA:

Pygmy Rabbit – Temperate Desert
Columbian White-tailed deer – Temperate Desert/Marine
Lesser Long-nosed bat – Subtropical Desert
Mexican Long-nosed bat – Subtropical Desert
Ocelot – Subtropical Desert
Jaguar – Subtropical Desert
Sonoran Pronghorn – Subtropical Desert
Amargosa Vole – Subtropical Desert
Hualapai Mexican Vole – Subtropical Steppe
Utah Prairie Dog – Subtropical Steppe/Temperate Desert
Preble’s Meadow Jumping Mouse – Temperate Steppe
Northern Idaho Ground Squirrel – Temperate Steppe
Woodland Caribou – Temperate Steppe (mountainous areas)
Grizzly Bear – Marine/Temperate Steppe (mountainous areas)
Canada Lynx – Marine/Temperate Steppe (mountainous areas)
San Joaquin Kit Fox - Mediterranean
Giant Kangaroo Rat – Mediterranean
Fresno Kangaroo Rat – Mediterranean
Tipton Kangaroo Rat – Mediterranean
Stephens’ Kangaroo Rat – Mediterranean
Morro Bay Kangaroo Rat – Mediterranean

Bighorn Sheep – Mediterranean
Riparian (San Joaquin Valley) Woodrat – Mediterranean
Buena Vista Lake Shrew – Mediterranean
Black-footed Ferret – various
Wolves – various

Note: in the discussions that follow, the general term “adverse health effects” is used in reference to exposure to certain herbicides under certain scenarios. The potential toxicological effects of herbicides on terrestrial wildlife, which were examined in ERAs, include mortality and sublethal effects. Examples of sublethal effects include harm to vital organs, changes in body weight, reduced reproductive success, and altered behavior, which may increase the animal’s susceptibility to predation (USDA Forest Service 2004). Sublethal effects to an animal’s health may also increase the severity of impacts associated with unrelated environmental stresses and other disturbances. In all of the effects assessments for birds found in this chapter, the term “adverse health effects” refers to the abovementioned or similar toxicological effects at the level of the organism. In addition, it is assumed that for TEP birds, these adverse health effects would potentially result in population-level effects for the species in question. Because many TEP bird species already have reduced, sensitive populations, mortality of individuals or reduced reproductive output could reduce the size of affected populations further, perhaps even leading to extirpation. Furthermore, if individuals were to become more physiologically predisposed to mortality from environmental stresses (such as predation or exposure to harsh environmental conditions), the risk for future population-level effects, including extirpations, would be increased.

Pygmy Rabbit

The primary reference for this section is:
USFWS. 2001n. Emergency Rule to List the Columbia Basin Distinct Population Segment of the Pygmy Rabbit (*Brachylagus idahoensis*) as Endangered. Federal Register 66(231): 59734-59749.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Upper Columbia Fish and Wildlife Office, Spokane, Washington.

The pygmy rabbit (*Brachylagus idahoensis*) was once distributed throughout much of the semi-arid, shrub steppe region of the Great Basin and adjacent intermountain zones of the conterminous western U.S. (Green and Flinders 1980a). It’s historical range likely included portions of Montana, Idaho, Wyoming, Utah, Nevada, California, Oregon, and Washington. Pygmy rabbits are not and have never been continuously distributed, typically occurring only in areas where sagebrush cover is sufficiently tall and dense, and where soils are sufficiently deep and loose to allow burrowing (Bailey 1936, Green and Flinders 1980a, Weiss and Verts 1984, Washington Department of Fish and Wildlife [WDFW] 1995). The local distribution of these habitat patches likely shifts across the landscape in response to various sources of disturbance (e.g., fire, flooding, grazing, and crop production), combined with long- and short-term weather patterns. Historically, more dense vegetation along permanent and intermittent stream corridors, alluvial fans, and sagebrush plains probably provided travel corridors or dispersal habitat for pygmy rabbits between appropriate use areas.

Once thought to be extirpated from the State of Washington, pygmy rabbits were again located in Washington in 1979. Intensive surveys in 1987 and 1988 resulted in the discovery of five small colonies of pygmy rabbits in southern Douglas County; three of which occurred on State lands and two of which occurred on privately-owned lands (WDFW 1995). With the exception of a single site record from Benton County in 1979, pygmy rabbits have been found only in southern Douglas and northern Grant Counties, Washington since 1956 (WDFW 2000a).

Pygmy rabbits typically are found in areas of tall, dense sagebrush cover, and are highly dependent on sagebrush to provide both food and shelter throughout the year (Orr 1940, Green and Flinders 1980a, WDFW 1995). The winter diet of pygmy rabbits is composed of up to 99% sagebrush (Wilde 1978), which is unique among Leporids (hares and rabbits; White et al. 1982). During spring and summer, their diet consists of roughly 51% sagebrush, 39% grasses (particularly native bunchgrasses, such as wheatgrass and bluegrass), and 10% forbs (Green and Flinders

1980b). There is evidence that pygmy rabbits preferentially select native grasses as forage during this period in comparison to other available foods. In addition, total grass cover relative to forbs and shrubs may be reduced within pygmy rabbit colonies as a result of its use as a food source during spring and summer.

The pygmy rabbit is believed to be one of only two Leporids in North America that digs its own burrows (Nelson 1909, Green and Flinders 1980a, WDFW 1995). Pygmy rabbit burrows typically are found in relatively deep, loose soils of wind-borne or water-borne origin, and the species occasionally make use of burrows abandoned by other species, such as the yellow-bellied marmot or badger. Burrows may also occur in areas of shallower or more compact soils that support sufficient shrub cover (Bradfield 1974). During winter, pygmy rabbits make extensive use of snow burrows to access sagebrush forage (Bradfield 1974, Katzner and Parker 1997), using them as protection from predators and inclement weather (Bailey 1936, Bradfield 1974). The burrows frequently have multiple entrances, some of which are concealed at the base of larger sagebrush plants (WDFW 1995). Burrows are relatively simple and shallow, often no more than 6.6 feet long and usually less than 3.3 feet deep with no distinct chambers (Bradfield 1974, Green and Flinders 1980a, Gahr 1993).

Pygmy rabbits may be active at any time of the day or night and appear to be most active during mid-morning. Pygmy rabbits maintain a low stance, have a deliberate gait, and are relatively slow and vulnerable in more open areas. They can evade predators by maneuvering through the dense shrub cover of their preferred habitats, often along established trails, or by escaping into their burrows (Bailey 1936, Severaid 1950, Bradfield 1974). Predation is the main cause of pygmy rabbit mortality (Green 1979). Potential predators include badgers, long-tailed weasels, coyotes, bobcats, great horned owls, long-eared owls, ferruginous hawks, and northern harriers (Janson 1946; Gashwiler et al; 1960, Green 1978; Wilde 1978; WDFW 1995).

Pygmy rabbits begin breeding in their second year and, in Washington, breeding occurs from February through July (WDFW 1995). Females may have up to three litters per year and average six young per litter (Green 1978, Wilde 1978). Breeding appears to be highly synchronous in a colony, and juveniles are often identifiable to cohorts (Wilde 1978). No evidence of nests, nesting material, or lactating females with young has been found in burrows (Bradfield 1974, Gahr 1993, WDFW 1995). Individual juveniles have been found under clumps of sagebrush, although it is not known precisely where the young are born in the wild or if they may be routinely hidden at the bases of scattered shrubs or within burrows (Wilde 1978). Recent information on captive pygmy rabbits indicates that females may excavate specialized natal burrows for their litters in the vicinity of their regular burrows (P. Swenson 2001; Shipley 2001). Apparently, females begin to dig and supply nesting material (e.g., grass clippings) to these burrows several days prior to giving birth and may give birth and nurse their young at the ground surface in a small depression near the burrow's entrance. After nursing, the young return to the burrow and the female refills the burrow entrance with loose soil and otherwise disguises the immediate area to avoid detection.

The Columbia Basin population of the pygmy rabbit was both emergency listed as endangered and proposed for listing as endangered on November 30, 2001. The 240-day period of the emergency listing has since passed, leaving the status as proposed for federal listing. Critical habitat has not been designated or proposed for this species. The number of pygmy rabbit colonies and active burrows in Washington has declined over the past decade (WDFW 2001a). Four of the five colonies located in 1987 and 1988 were very small, with fewer than 100 active burrows (WDFW 1995); three of these colonies have since been extirpated. The largest colony (at the state-owned Sagebrush Flat site in Douglas County) contained roughly 588 active burrows in 1993, when it was estimated to support fewer than 150 rabbits (Gahr 1993). With an additional colony discovered on privately-owned land in northern Grant County in 1997, three known colonies remained in 1999 (WDFW 2001a). One of these sites experienced a catastrophic fire in 1999 and declined to three active burrows, while the newly discovered site declined, for unknown reasons, to two active burrows following the winter of 1999-2000. These two colonies are now thought to be extirpated (Hays and McCall 2001, WDFW 2001b). In addition, during the winter of 1997-1998, the number of active pygmy rabbit burrows at Sagebrush Flat declined by approximately 50%, and has continued to decline each year since (WDFW 2001a). The entire wild pygmy rabbit population in Washington is now considered to consist of fewer than 50 individuals, possibly from just one known colony at Sagebrush Flat in Douglas County (McCall 2001). The Columbia Basin pygmy rabbit is imminently threatened by this recent decrease in population, which has caused it to be susceptible to the combined influence of catastrophic environmental events, habitat or resource failure, disease, predation, and loss of genetic heterogeneity.

Effects of Vegetation Treatments on the Pygmy Rabbit

Effects Common to All Treatment Methods

Indirect Effects. Pygmy rabbits are highly dependent on sagebrush to provide food and cover throughout the year, and feed on other types of vegetation during the spring and summer. Therefore, any treatment that removes large amounts of vegetation from pygmy rabbit habitats is likely to negatively affect the species. Removal of dense sagebrush stands would have the greatest effect on pygmy rabbits, but removal of more marginal stands could also have adverse effects, as these stands may act as dispersal corridors for the species.

Treatments that target non-native species would be expected to improve pygmy rabbit habitats. Areas with dense cover of downy brome are apparently avoided by pygmy rabbits (Weiss and Very 1984 *cited in* USFWS 2001n). As pygmy rabbits are unlikely to be present in areas with a high coverage of non-native species, treatments that restore these areas to more native conditions could potentially improve the availability of habitat for future occupation by pygmy rabbits.

Given the small size of the existing pygmy rabbit population, a wildfire burning through the habitat could potentially extirpate the species. Therefore, any treatments that reduce the presence of fuels in pygmy rabbit habitat, or in areas near to habitat from which wildfires could spread, would likely have a long-term positive benefit for the species.

Effects of Fire

Direct Effects. A prescribed fire could cause some injury, and possibly mortality, to pygmy rabbits. However, because they live in underground burrows, most individuals would be able to seek cover during a burn. The highest risks would be for young pygmy rabbits, which may or may not reside in burrows after birth, and which may not be able to escape a fire.

Indirect Effects. Fire could negatively affect pygmy rabbit habitat, since sagebrush is easily killed by fire. Because of their close association with tall, dense stands of sagebrush, pygmy rabbits are precluded from occupying frequently burned areas (USFWS 2001n). Historically, pygmy rabbits were probably adapted to periodic fire, which would eliminate patches of habitat temporarily. However, the reinvasion of sagebrush onto these sites after fire would have eventually made them suitable for the pygmy rabbit once again. Currently, the frequency of fire in sagebrush habitats has increased, destroying sources of sagebrush seed and precluding re-establishment of sagebrush. In addition, even if sagebrush were to recolonize a burn site, the extremely small population of pygmy rabbits and the lack of suitable habitat does not allow for this cyclic burn and recolonization.

Mechanical Treatments

Direct Effects. The use of heavy equipment in pygmy rabbit habitat could cause some injury and/or mortality to pygmy rabbits by crushing burrows and potentially the animals inside them. Young rabbits would be the most at risk. The disturbance of the equipment could also interfere with foraging activities for a short time.

Indirect Effects. The use of mechanical equipment in areas that support pygmy rabbits could have long-term impacts on habitat. Vehicles and other heavy equipment could cause widespread damage to pygmy rat burrows, which are relatively shallow, and may collapse even under the weight of a human or a large animal (Wilde 1978 *cited in* USFWS 2001n). In addition, compaction and disturbance of the soil could make sites less suitable for future occupation by pygmy rabbits. Although pygmy rabbits would likely be able to repair their burrow systems after damage, the small population could be severely impacted even by such short-term effects.

Manual Treatments

Direct Effects. The use of manual treatment methods would be unlikely to injure or kill pygmy rabbits. It is possible, however, that a worker could collapse a burrow, potentially harming a pygmy rabbit inside of it. The presence of workers in the area could also temporarily interfere with foraging activities.

Indirect Effects. Workers could potentially cause structural damage to burrows and dense stands of older sagebrush, both of which are key components of pygmy rabbit habitat.

Biological Control Treatments

Domestic Animals. Populations of pygmy rabbits have coexisted with various levels of grazing throughout their historic range for many years (WDFW 1995). However, the current status of populations makes them highly susceptible to any level of mortality or population stress associated with herbivory in their habitats.

Direct Effects. Domestic animals would be unlikely to cause direct mortality to pygmy rabbits, which would be able retreat into burrows. However, domestic animals could be capable of causing burrows to collapse through trampling, which could conceivably result in mortality or injury to pygmy rabbits. The presence of domestic animals in an area could also interfere with foraging activities.

Indirect Effects. Use of domestic animals could result in damage to pygmy rabbit burrow systems through trampling (Rauscher 1997 *cited in* USFWS 2001n). In addition, some structural damage to dense stands of older sagebrush used by pygmy rabbits could also occur as a result of trampling.

Domestic animals favor some of the same food sources as pygmy rabbits, primarily native grasses and forbs. The competition with domestic animals for these resources would be an added stress on pygmy rabbit populations. Extensive grazing can also increase the density of non-native species and young sagebrush stands (Daubenmire 1988, WDFW 1995). Over the long term, this sort of disturbance could actually result in the growth of tall, dense sagebrush stands, potentially improving cover conditions for pygmy rabbits. It is currently unclear whether light or moderate levels of grazing would be compatible with pygmy rabbit conservation efforts (USFWS 2001n). It is possible that, given the current threat of species extirpation, no grazing in suitable pygmy rabbit habitat is appropriate at this time.

Other Biological Control Agents Biological control agents would be unlikely to affect pygmy rabbits or their habitat. These agents would target particular weed species, and their effects would be gradual. Burrows could collapse as a result of workers walking on them during the release of agents or monitoring. There could also be some unanticipated impacts associated with the use of these agents. However, given that agents would be pre-tested under laboratory conditions, adverse effects are not reasonably foreseeable.

Herbicides

Direct Effects. Use of trucks/ATVs to apply herbicides could cause some injury and/or mortality to pygmy rabbits by crushing burrows and potentially the animals inside them. Young rabbits would be most at risk. It is also possible that disturbances associated with herbicide application procedures would have temporary behavioral effects on pygmy rabbits.

Although it is likely that pygmy rabbits would flee or retreat into burrows during herbicide applications, it is possible that some animals would be unintentionally exposed to these chemicals. Based on the results of the ERAs for terrestrial vertebrate species (see Table 6-2), direct spray by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, could conceivably result in adverse health effects to pygmy rabbits. Pygmy rabbits could also come into contact with sprayed foliage after the application. Via this exposure pathway, adverse health effects to rabbits could occur if vegetation was sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate.

Ingestion of plant materials sprayed by 2,4-D, diquat, or diuron at the typical application rate, or by bromacil, fluridone, glyphosate, hexazinone, or tebuthiuron, at the maximum application rate, could pose health risks to pygmy rabbits (Table 6-5). Should herbicide treatments with one or more of these herbicides occur in areas where pygmy rabbits forage for food, it is reasonably foreseeable that rabbits could consume food items to which herbicides were applied. However, it is unlikely that all of a rabbit's diet would come from contaminated vegetation, as assumed by ERAs when predicting these risks.

Indirect Effects. Use of horses, ATVs, and trucks to apply herbicides, in addition to applications on foot, could result in damage to pygmy rabbit burrow systems. In addition, the physical disturbance associated with herbicide applications could cause structural damage to pygmy rabbit habitat. Use of herbicides could also cause a temporary reduction in food items, although treatment programs would not target the native grasses and forbs consumed by pygmy rabbits. Over the long term, a reduction in non-native species would likely improve the quality of treated areas, making them more suitable for supporting pygmy rabbit populations.

Mitigation Measures

In order to avoid or minimize potential effects to the pygmy rabbit resulting from the proposed vegetation treatments, the BLM would be required to implement the conservation measures listed below:

- Prior to treatments, survey all suitable habitat for pygmy rabbits.
- Address pygmy rabbits in all management plans prepared for treatments within the range of the species' historical habitat.
- Do not burn, graze, or conduct mechanical treatments within 1 mile of known pygmy rabbit habitat.
- Do not use 2,4-D, diquat, or diuron in pygmy rabbit habitats; do not broadcast spray these herbicides within ¼ mile of pygmy rabbit habitat.
- Where feasible, avoid use of the following herbicides in pygmy rabbit habitat: bromacil, clopyralid, fluridone, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, tebuthiuron, and triclopyr.
- Where feasible, spot treat vegetation in pygmy rabbit habitat rather than broadcast spraying.
- Do not broadcast spray clopyralid, glyphosate, hexazinone, picloram, or triclopyr in pygmy rabbit habitat; do not broadcast spray these herbicides in areas adjacent to pygmy rabbit habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, imazapyr, fluridone, metsulfuron methyl, or tebuthiuron in or near pygmy rabbit habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of bromacil, glyphosate, hexazinone, tebuthiuron, or triclopyr to vegetation in pygmy rabbit habitat, utilize the typical, rather than the maximum, application rate.

In addition, project-level conservation measures would also be developed by local BLM offices during the development of NEPA documents for site-specific treatment projects.

Determination of Effects

Assuming that vegetation treatments could occur anywhere on public lands, including areas occupied by pygmy rabbits, the proposed treatment program would be **likely to adversely affect** pygmy rabbits and/or their habitat. In order to avoid or minimize these potential effects, the BLM would be required to implement the conservation measures discussed in the previous section. Following both programmatic-level and project-level conservation measures would be expected to reduce treatment effects to a **not likely to adversely affect** determination for the pygmy rabbit.

TABLE 6-5
Summary of Effects to TEP Mammals via Ingestion Pathways

Herbicide	Ingestion of Vegetation – Small Mammals		Ingestion of Vegetation – Large Mammals		Ingestion of Small Vertebrate Prey		Ingestion of Invertebrate Prey[1]	
	Effect	Risk level[2]	Effect	Risk level	Effect	Risk level	Effect	Risk level
2,4-D	Adverse effects	Typical rate: L Maximum rate terrestrial: L Maximum rate aquatic: M	Adverse effects	Typical rate: M Maximum rate terrestrial: M Maximum rate aquatic: H	Adverse effects	Typical rate: L Maximum rate terrestrial: L Maximum rate aquatic: M	Adverse effects	Typical rate: M Maximum rate terrestrial: H Maximum rate aquatic: H
Bromacil	Adverse effects	Typical rate: N/A Maximum rate: L	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L	No effects	--
Chlorsulfuron	No effects	--	No effects	--	No effects	--	No effects	--
Clopyralid	No effects	--	Adverse effects	Typical rate: L Maximum rate: L	No effects	--	Adverse effects	Typical rate: L Maximum rate: L
Dicamba	Adverse effects	Maximum rate: L	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: L Maximum rate: L
Diflufenzopyr	No effects	--	No effects	--	No effects	--	No effects	--
Diquat	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L	Adverse effects	Typical rate: L Maximum rate: M
Diuron	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: H Maximum rate: H	Adverse effects	Typical rate: L Maximum rate: L	Adverse effects	Typical rate: N/A Maximum rate: L
Fluridone	Adverse effects	Typical rate: N/A Maximum rate: L (chronic risk only)	No effects	--	No effects	--	No effects	--
Glyphosate	Adverse effects	Typical rate: N/A Maximum rate: L	Adverse effects	Typical rate: L Maximum rate: M	No effects	--	Adverse effects	Typical rate: L Maximum rate: M
Hexazinone	Adverse effects	Typical rate: N/A Maximum rate: L	Adverse effects	Typical rate: L Maximum rate: M	Unknown[3]	Unknown	Adverse effects	Typical rate: M Maximum rate: M
Imazapic	No effects	--	No effects	--	No effects	--	No effects	--

TABLE 6-5 (Cont.)
Summary of Effects to TEP Mammals via Ingestion Pathways

Herbicide	Ingestion of Vegetation – Small Mammals		Ingestion of Vegetation – Large Mammals		Ingestion of Small Vertebrate Prey		Ingestion of Invertebrate Prey[1]	
	Effect	Risk level[2]	Effect	Risk level	Effect	Risk level	Effect	Risk level
Imazapyr	No effects	--	Adverse effects	Typical rate: N/A Maximum rate: L	No effects	--	Adverse effects	Typical rate: L Maximum rate: L
Metsulfuron methyl	No effects	--	Adverse effects	Typical rate: N/A Maximum rate: L	No effects	--	Adverse effects	Typical rate: N/A Maximum rate: L
Overdrive®	No effects	--	Adverse effects	Typical rate: L Maximum rate: M	No effects	--	No effects	--
Picloram	No effects	--	Adverse effects	Typical rate: L Maximum rate: M	No effects	--	Adverse effects	Typical rate: L Maximum rate: M
Sulfometuron methyl	No effects	--	No effects	--	No effects	--	No effects	--
Tebuthiuron	Adverse effects	Typical rate: N/A Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L	No effects	--	No effects	--
Triclopyr acid	No effects	--	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L	Adverse effects	Typical rate: L Maximum rate: M
Triclopyr BEE	No effects	--	Adverse effects	Typical rate: L Maximum rate: M	Adverse effects	Typical rate: N/A Maximum rate: L	Adverse effects	Typical rate: L Maximum rate: M

[1] Only the ERAs for 2,4-D, picloram, clopyralid, glyphosate, metsulfuron methyl, and triclopyr assessed risks to insectivorous mammals. For all other herbicides, insectivorous birds were used as surrogates when completing risk assessments.
[2] L = low risk; M = medium risk; H = high risk; and N/A = ERAs did not predict risk at this application rate.
[3] Unknown = ERAs did not assess risk to birds for this herbicide via this exposure pathway.
Note: risks to mammals from ingesting contaminated fish are assumed to be the same as those to birds (see Table 6-4).

Columbian White-tailed Deer

The primary reference for this section is:
USFWS. 2002l. Supplemental Proposed Rule to Remove the Douglas County Population of Columbian White-tailed Deer From the Federal List of Endangered and Threatened Wildlife. Federal Register 67(120): 42217-42229.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS, Oregon Fish and Wildlife Office, Portland, Oregon.

The Columbian white-tailed deer (*Odocoileus virginianus leucurus*) is the westernmost representative of 30 subspecies of white-tailed deer in North and Central America. The subspecies was formerly distributed throughout the bottomlands and prairie woodlands of the lower Columbia, Willamette, and Umpqua River basins in Oregon and southern Washington (Bailey 1936, Verts and Carraway 1998). It is believed that this deer was locally common, particularly in riparian areas along major rivers (Gavin 1978). With the arrival and settlement of pioneers in the fertile river valleys, the decline in Columbian white-tailed deer numbers was rapid (Gavin 1978). By 1940, a population of 500 to 700 animals along the lower Columbia River in Oregon and Washington, and a disjunct population of 200 to 300 in Douglas County survived (Crews 1939, Gavin 1984, Verts and Carraway 1998).

Columbian white-tailed deer in Douglas County are most often associated with riparian habitats, though the deer also uses a variety of lower elevation habitat types (e.g., grassland, grass shrub, oak savanna, oak-hardwood woodland, oak-hardwood savanna shrub, oak-hardwood conifer, conifer, and urban/suburban yards; Ricca 1999). Open areas are used for feeding between dusk and dawn. The Columbia River population occurs in wet bottomlands and dense forest swamps where there is little elevational relief, and which receive a large amount of precipitation. The diet of Columbian white-tailed deer consists of forbs, shrubs, grasses, and a variety of other foods, such as lichens, mosses, ferns, seeds, and nuts (Lowell Whitney 2001).

Like other types of deer, Columbian white-tailed deer breed in the winter, primarily in November and December. Most fawns are born between mid-May and mid-June. Columbian white-tailed deer first breed as yearlings (18 months), and young females typically give birth to a single fawn. After 2 years of age, twins are more common.

The Columbian white-tailed deer was federally listed as endangered on March 11, 1967. On May 11, 1999, the USFWS proposed to delist the Douglas County subpopulation. Numbers of white-tailed deer have more than doubled since the species was first listed. The Douglas County subpopulation is now estimated at over 5,000 animals, and the Columbia River subpopulation is estimated at approximately 1,000 animals. This species is primarily threatened by a lack of suitable habitat. Logging has degraded forest habitat in some areas. In addition, periodic flooding of the Columbia River, and residential development along the North Umpqua River are also threats to the subspecies.

Effects of Vegetation Treatments on the Columbian White-tailed Deer

Effects Common to All Treatment Methods

Indirect Effects. Removal of vegetation from riparian vegetation can cause degradation of these habitats (see the effects analysis for aquatic species in Chapter 5) and indirectly affect Columbian white-tailed deer populations. Control of shrub and forest communities in areas used by deer for forb and grass forage would have positive effects on habitat. In addition, creation of new edge habitat may also be beneficial for deer. The suitability of treatment methods in a given area is highly dependent on the scale of the treatment, the ecology of the surrounding area, and the primary uses of that area by deer.

Prescribed Fire Treatments

Direct Effects. Like other large mammals, deer are typically able to escape a fire by moving out of a burn area, and therefore should not experience direct mortality from prescribed fire. However, newborn fawns that are unable to move quickly could be killed if fire burned through the area.

Indirect Effects. Given that Columbian white-tailed deer occupy a variety of habitat types, prescribed fire should not eliminate appropriate habitat. Fire creates early successional communities, which support new growth of grasses, forbs, and shrubs, and provide deer with a preferred food source (Bradley et al. 1992). Preferred habitat for deer contains a combination of open and closed communities, with deer often frequenting edge habitats that provide both food and cover. Given the limited remaining habitat of this species, however, it is conceivable that a large, intense prescribed burn could reduce the amount of suitable closed canopy habitat available for this species.

Mechanical Treatment Methods

Direct and Indirect Effects. Provided that they do not affect large areas of the deer's limited remaining habitat, mechanical methods for removing vegetation and debris should not adversely affect the Columbian white-tailed deer. However, riparian areas should be protected from degradation caused by use of heavy equipment.

Manual Treatment Methods

Direct and Indirect Effects. Manual control methods are unlikely to cause adverse effects to Columbian white-tailed deer or their habitats. Depending on the extent of control, plant removal may cause some soil disturbance, especially in riparian areas.

Biological Control Treatments

Domestic Animals
<u>Indirect Effects</u>. Depending on where and how the treatment is implemented, use of domestic animals to contain weeds may have either positive effects on deer habitat by preventing reestablishment of shrub and forest communities (see above), or negative effects by reducing shrub and forest communities that provide cover. In addition, deer avoid areas where cattle are present, and may experience some negative effects through competition for forage. In woodland communities, use of domestic animals to control weeds would be expected to have negative effects on deer habitat, as domestic animals would be likely to trample important browse plants.

Other Biological Control Agents
<u>Direct and Indirect Effects</u>. Biological control methods are unlikely to have effects on Columbian white-tailed deer or their habitat. These agents target specific, undesirable plant species, and have a gradual effect on vegetation. However, since there is limited knowledge about the long-term effects of these agents, it is possible that unanticipated impacts to the ecosystem (and therefore deer or their habitat) could occur.

Herbicides

Direct Effects. Although deer would readily flee people and equipment associated with herbicide applications, it is possible that some animals would inadvertently be exposed to herbicides used on public lands. According to the ERAs, adverse health effects to Columbian white-tailed deer could potentially occur as a result of direct spray by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate (see Table 6-2). Following an herbicide treatment, adverse effects could potentially occur if deer were to come into contact with foliage treated with 2,4-D applied at the typical application rate, or with glyphosate, hexazinone, or triclopyr applied at the maximum application rate.

Risk assessments predicted that if deer were to ingest plant materials treated with 2,4-D, bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, Overdrive®, picloram, or triclopyr at the typical application rate, adverse health effects could potentially occur (see Table 6-5). Furthermore, if deer were to ingest plant materials treated with imazapyr, metsulfuron methyl, or tebuthiuron, at the maximum application rate, adverse health effects would be possible. These predictions are overly conservative in that they assume 100% of the animal's diet would consist of contaminated vegetation, which would be unlikely unless all of the animal's habitat was treated.

Indirect Effects. Herbicide treatments could affect deer indirectly by temporarily reducing the availability of forage plants. Because deer are mobile and often graze in a variety of habitats, these effects would be unlikely to affect deer populations unless a very extensive area was sprayed, or few alternate foraging sites were available. Reduction in forage could also affect deer populations if treatments occurred during a time when forage was already limited. Over the long term, a reduction in non-native species could increase the quality of deer forage.

Conservation Measures

The projected short-term adverse effects of vegetation treatments on the Columbian white-tailed deer could be avoided by implementing the following programmatic-level conservation measures.

- Prior to treatments, survey for evidence of white-tailed deer use of areas in which treatments are proposed to occur.
- Address the protection of Columbian white-tailed deer in local management plans developed in association with treatment programs.
- In areas that are likely to support Columbian white-tailed deer, protect riparian areas from degradation by avoiding them altogether, or utilizing SOPs. Consult Chapter 5 for appropriate conservation measures to be used in protected riparian areas.
- In habitats used by deer, conduct treatments that use domestic animals during the plant growing season, and remove the animals after clearing has been achieved.
- Do not use domestic animals to control weeds in woodland habitats utilized by Columbian white-tailed deer.
- In areas where Columbian white-tailed deer occur, or may possibly occur, avoid the use of fences to keep domestic animals out of sensitive habitats or to otherwise restrict their movement (fence accidents are associated with deer mortality).
- Avoid burning in deer habitats during the fawning season.
- Closely follow all application instructions and use restrictions on herbicide labels; in riparian habitats use only those herbicides that are approved for use in riparian areas.
- Avoid broadcast spray treatments in areas where Columbian white-tailed deer are known to forage.
- Do not use 2,4-D in Columbian white-tailed deer habitats; do not broadcast spray 2,4-D within ¼ mile of Columbian white-tailed deer habitat.
- Where feasible, avoid use of the following herbicides in Columbian white-tailed deer habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, Overdrive®, picloram, tebuthiuron, and triclopyr.
- Do not broadcast spray bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, Overdrive®, picloram, or triclopyr in Columbian white-tailed deer habitat; do not broadcast spray these herbicides in areas adjacent to Columbian white-tailed deer habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying imazapyr, metsulfuron methyl, or tebuthiuron in or near Columbian white-tailed deer habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, imazapyr, metsulfuron methyl, tebuthiuron, or triclopyr to vegetation in Columbian white-tailed deer habitat, utilize the typical, rather than the maximum, application rate.

In addition, site-specific and project specific conservation measures would need to be developed by local BLM offices to ensure complete protection of the Columbian white-tailed deer.

Summary of Effects

Assuming that vegetation treatments could occur anywhere in habitats used by deer on public lands, the proposed treatments would be **likely to adversely affect** Columbian white tailed deer and/or their habitat. However, with the development of project-level treatment programs that incorporate both programmatic- and project-level conservation measures (as discussed in the previous section), most treatment effects could be reduced to a **not likely to adversely affect** determination.

Bats: Lesser Long-nosed Bat and Mexican Long-nosed Bat

Lesser Long-nosed Bat
The primary reference for this section is:
USFWS. 1995f. Lesser Long-nosed Bat Recovery Plan. Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The lesser long-nosed bat (*Leptonycteris curasoae yerbabuenae*) is a nectar-, pollen-, and fruit-eating bat that migrates seasonally from Mexico to southern Arizona and southwestern New Mexico. It has been found in southern Arizona from the Picacho Mountains southwest to the Agua Dulce Mountains and southeast to the Chiricahua Mountains. It has also been found in far southwestern Mexico in the Animas and Peloncillo mountains, and throughout the drier parts of Mexico. The subspecies is a seasonal resident in Arizona, usually arriving in early April and departing in mid-to-late September. It apparently resides in New Mexico only from mid-July to early September (Hoyt et al. 1994).

Two sets of resources are critical for the lesser long-nosed bat: suitable day roosts and suitable concentrations of food plants. It is unclear precisely what factors identify potential roost sites as “suitable,” but maternity roosts tend to be very warm and poorly ventilated, at least where the young are actually raised. Such roosts reduce the energetic requirements of adult females while they are raising their young (Arends et al. 1995). Lesser long-nosed bats have been found living in caves and mines displaying a variety of microclimates (e.g., dry and hot, wet and hot, dry and cool, wet and cool). They are found in well-ventilated caves as well as those that are poorly ventilated and filled with strong ammonia fumes. The subspecies sometimes co-occurs with other species of bats. Independent of its day-roosting location, the subspecies appears to be sensitive to human disturbance, and bats may temporarily abandon their roosts and move to another in response to a single brief human visit.

The lesser long-nosed bat has specialized food requirements. Columnar cactus flowers and fruits and agave flowers are believed to represent this bat’s core diet. Its consumption of nectar and pollen produced by paniculate agave flowers is well-known (e.g., Howell 1974, 1976, 1979). Important also are nectar, pollen, and fruit produced by a variety of columnar cacti (Howell 1974; Cockrum 1991; Fleming et al. 1993). Flowers and fruits of two to three species of columnar cacti (pachycereus, saguaro, and organ pipe cactus) provide nearly all of the energy and nutrients obtained by pregnant and lactating females roosting in the Sonoran Desert in the spring and early summer. By eating nectar, pollen, and fruit, lesser long-nosed bats are important pollinators and seed dispersers of their food plants.

Female lesser long-nosed bats are thought to bear only a single young per year, and the timing of mating and parturition likely varies geographically. It is thought that periods of birth and lactation coincide with peak flower availability. Young bats have well-developed feet and are left to hang in the day roost from the day of birth while the mother leaves the roost to forage. Young probably are nursed for about 6 weeks, begin to fly at 4 weeks, and begin to leave the roost on evening flights at 6 to 7 weeks.

The lesser long-nosed bat was federally listed as endangered on September 30, 1988. Critical habitat has not been designated for the subspecies. Primary threats include human and other disturbances at roosting colonies, and loss and degradation of foraging habitat. Although population estimates are difficult for this migratory species, it has apparently made a substantial recovery since surveys in 1984-1985 (Commission for Environmental Cooperation 2000). In Arizona, populations are believed to be two orders of magnitude greater than they were in 1985, and numbers at some locations appear to be relatively stable from year to year.

Mexican Long-nosed Bat
The primary reference for this section is:
USFWS. 1994m. Mexican Long-nosed Bat (*Leptonycteris nivalis*) Recovery Plan. Albuquerque, New Mexico.

The Mexican long-nosed bat (*Leptonycteris nivalis*), also a migratory species, ranges from southern Mexico to southern Texas and New Mexico. It is found at medium to high elevations (1,550 to 9,330 feet) in desert scrub, open conifer-oak woodlands, and pine forest habitats in the Upper Sonoran and Transitional life zones. The Mexican long-nosed bat is a colonial species that usually roosts in caves, but can also be found in mines, culverts, and hollow trees.

The species feeds primarily on the nectar from agave plants and also on pollen from cacti flowers and some soft fruits. Bats become active in late evening, leaving roosts in search of their night blooming food plants. The Mexican long-nosed bat is considered a vital pollinator for some plant species, such as the agave.

Reproductive information for the Mexican-long nosed bat is limited. Most young are born in May. However, some studies indicate that this species might have two birth peaks a year, the first in spring and the second peak in September. It is suggested that the migratory nature of this species is derived from the mutualistic relationship it shares with the agave plants on which it feeds. Although the agaves, which flower only once before dying, can reproduce vegetatively by sending shoots from the bottom to the main stem, they rely on the Mexican long-nosed bat and other nectar feeders for cross-pollination to keep up an adequate amount of gene flow. The bat's migratory pattern suggests that it follows the onset of flowering agaves northward, seasonally. When climatic conditions severely limit the number of agaves that flower in any given year, the bat will range farther for additional food sources.

The Mexican long-nosed bat was federally listed as endangered on September 30, 1988. Critical habitat has not been designated for this species. The primary threats to this species are modification or destruction of roost sites and foraging habitat. A lack of suitable roost sites for this species may be a limiting factor. Other potential threats include pesticides, competition for roosts and nectar, natural catastrophes, disease, and predation. The population status of the Mexican long-nosed bat is unknown for certain, but it is suspected to be declining. Population estimates are difficult, given the migratory nature and rarity of the species, and the probability that seasonal movements are connected with climactic conditions that stimulate agave flower blooming (Texas Parks and Wildlife Department 2003).

Effects of Vegetation Treatments on the Lesser Long-nosed Bat and the Mexican Long-nosed Bat

Effects Common to All Treatment Methods

Indirect Effects. All vegetation treatments that reduce the coverage of non-native species would be expected to have a positive effect on the habitat of these two bat species. Weed removal activities would likely improve habitat for nectar plants, such as agave and cactus species. In addition, some of the most common invasive species found in bat habitat areas are fire tolerant species, such as red brome, that increase the potential for a severe wildfire by adding to the fuels base (USDI BLM 1996b). Furthermore, all vegetation treatments that reduce other forms of fuels would also provide a long-term benefit to lesser and Mexican long-nosed bats by helping to reduce the likelihood of a future damaging wildfire. A large fire could destroy large stands of nectar plants.

Prescribed Fire Treatments

Direct Effects. Few direct effects to lesser and Mexican long-nosed bats are expected from prescribed burns. Most bats roost in areas that would not be impacted by fire, such as caves and mine tunnels. However, bats that roost in hollow trees could be killed by a burn if the roost tree was consumed by the fire. There could also be some injury to bats through smoke inhalation, depending on the intensity and location of the fire.

Indirect Effects. Nectar species such as agave are fairly resistant to fire and can survive some burning. However, a large, intense fire could reduce the availability of nectar plants for these species.

Mechanical Treatment Methods

Direct Effects. Mechanical control methods would be most likely to harm lesser and Mexican long-nosed bats that roost in trees. During treatments, these bats could be disturbed, injured, or killed. The majority of bats, however, would be in caves, mines, and old buildings, sites that would not be impacted by the equipment or the vegetation removal.

Indirect Effects. Mechanical treatments could affect bat habitat by removing trees used for roosting, as well as potential future roosting trees. Large-scale removal of vegetation could also affect bats by reducing the coverage of nectar plants, thus reducing the available food supply.

Manual Treatment Methods

Direct and Indirect Effects. There are no anticipated effects from manual control treatment methods, either on bats or their habitats. There would be minimal disturbance associated with hand removal of vegetation, and nectar plants would not be targeted. The effects of removing non-native vegetation would likely be positive and minor.

Biological Control Treatments

Domestic Animals

<u>Indirect Effects</u>. Use of domestic animals to treat vegetation in bat habitats could have a number of adverse effects on the forage base of these two species. High levels of grazing can lead to the depletion of agave plants, which are an important nectar source for long-nosed bats. Domestic animals have been observed foraging on developing flower stalks (USDA Forest Service 1996). Although plants are sometimes able to sprout a new rosette, prolonged grazing in the same area would be expected to reduce flower production. Other evidence of domestic animals harming nectar plants has been observed in the trampling of saguaro seedlings, grazing seedlings, or grazing nurse plants, which are other species that provide protective cover to the seedlings (USDI BLM 1996b). Domestic animals can also impact habitat by contributing to the spread of invasive species that increase fire fuel loads and degrade the habitat, such as red brome.

Other Biological Control Agents

<u>Direct and Indirect Effects</u>. There are no anticipated effects from biological control treatment methods, either on bats or their habitat. Biological control agents target non-native species, and have a gradual effect on vegetation. Given the limited knowledge about the long-term effects of these agents, however, it is possible that unanticipated impacts to the ecosystem (and therefore bats and their habitat) could occur.

Herbicides

Direct Effects. Because lesser and Mexican long-nosed bats roost in covered areas during the day and are active in the evening, direct spray of bats during herbicide treatments would be unlikely. According to the ERAs, should bats be exposed to a direct spray of 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or of imazapyr or metsulfuron methyl at the maximum application rate, adverse health effects could potentially occur (see Table 6-2).

A more likely exposure scenario would be dermal contact with, or ingestion of, plant materials after they were sprayed by an herbicide. According to risk assessments, adverse health effects to bats could occur if vegetation was sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate. Furthermore, adverse effects could potentially occur if bats were to ingest plant materials treated with 2,4-D, diquat, or diuron at the typical application rate, or with bromacil, fluridone, glyphosate, hexazinone, or tebuthiuron at the maximum application rate (see Table 6-5). These effects would be possible if herbicide applications occurred in areas where bats forage for nectar, pollen, and/or fruit.

Indirect Effects. Adverse effects to non-target plant species are predicted as a result of direct spray by all herbicides approved for use by the BLM. In addition, non-target plants could also be impacted by off-site drift and surface runoff of several herbicides that would be used by the BLM to treat vegetation (see Tables 4-2 through 4-4 for more information on potential effects to vegetation). Since lesser and Mexican long-nosed bats depend on nectar plants for food, inadvertent mortality or reduced reproductive output of these cactus species as a result of herbicide treatments would have adverse effects on lesser and Mexican long-nosed bats.

Conservation Measures

In order to prevent or minimize the potential effects to lesser and Mexican long-nosed bats from vegetation treatments, the following conservation measures should be followed:

- Prior to treatments, survey all potentially suitable habitat for the presence of bats or their nectar plants.
- At the local level, incorporate protection of lesser and Mexican long-nosed bats into management plans developed for proposed treatment programs.
- Instruct all field personnel on the identification of bat nectar plants and the importance of their protection.
- Protect nectar plants from modification by treatment activities to the greatest extent possible. Do not remove nectar plants during treatments. Avoid driving over plants, piling slash on top of plants, burning, and using domestic animals to control weeds.
- Do not burn within a mile upwind of known bat roosts.
- To protect nectar plants and roost trees from herbicide treatments, follow recommended buffer zones and other conservation measures for TEP plant species in areas where populations of nectar plants and roost trees occur.
- Do not use 2,4-D, diquat, or diuron, in lesser or Mexican long-nosed bat habitats; do not broadcast spray these herbicides within ¼ mile of lesser or Mexican long-nosed bat habitat.
- Where feasible, avoid use of the following herbicides in lesser and Mexican long-nosed bat habitat: bromacil, clopyralid, fluridone, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, tebuthiuron, and triclopyr.
- Do not broadcast spray clopyralid, glyphosate, hexazinone, picloram, or triclopyr in lesser or Mexican long-nosed bat habitat; do not broadcast spray these herbicides in areas adjacent to lesser or Mexican long-nosed bat habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, imazapyr, or metsulfuron methyl, or tebuthiuron in or near lesser or Mexican long-nosed bat habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of bromacil, diuron, glyphosate, hexazinone, tebuthiuron, or triclopyr to vegetation in lesser or Mexican long-nosed bat habitat, utilize the typical, rather than the maximum, application rate.
- If conducting spot treatments of herbicides in lesser or Mexican long-nosed bat habitats, avoid potential roost sites.

In addition, local BLM offices would be required to prepare site-specific conservation measures to protect these species prior to conducting treatments.

Summary of Effects

Assuming that any vegetation treatments could occur anywhere on public lands, including those areas with key long-nosed bat habitat elements, the proposed action would be **likely to adversely affect** Mexican and lesser long-nosed bats. However, these effects could be avoided or minimized by following both programmatic- and project-level conservation measures, as discussed in the previous section. By developing treatment programs that incorporate these conservation measures, local BLM offices would be able to ensure that most treatment effects could be reduced to a **not likely to adversely affect** determination.

Desert Cats: Ocelot and Jaguar

Ocelot
The primary reference for this section is:
USFWS. 1990h. Listed Cats of Texas and Arizona Recovery Plan (With Emphasis on the Ocelot). Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The ocelot (*Felis pardalis*) is a medium-sized spotted cat that ranges from southern Texas and Arizona to northern Argentina. Within this area, the ocelot can be found in humid tropical and subtropical forests, coastal mangroves, swampy savannas, and semi-arid thornscrub (Leopold 1959, International Union for Conservation of Nature 1978). The species is thought to be rare and threatened in many parts of its range. Two ocelot subspecies historically ranged into the United States: the Texas ocelot in Texas, and the Sonora ocelot in Arizona. Although the Sonora ocelot historically ranged into southeastern Arizona as far north as Fort Verde (Cockrum 1960, Hall 1981), the species has been infrequently sighted in Arizona over the last 50 years, and it may be extirpated from the state. The species is still known to occur in Texas, where it is now restricted to several isolated patches of suitable habitat in three or four counties of Rio Grande Plains. Population numbers are estimated at 80 to 120 individuals (2001 estimate).

Considered more adaptable than the jaguar, the ocelot may persist in partly-cleared forests, second growth woodland, and abandoned cultivation that has gone back to brush (IUCN 1978). Ocelots are primarily active at dusk and at night, spending the day in heavy brush (Leopold 1959, Grzimek 1975, Tewes and Everett 1982). Ocelots make dens in caves, hollow trees, and other similar openings. Their prey consists of small to medium-sized mammals and birds, but may also include reptiles, fish, and invertebrates (Leopold 1959, Morris 1965, Grzimek 1975, Nowak and Paradiso 1983).

The usual age of first conception in ocelots is 2 years (Seager and Demorest 1978). The gestation period is approximately 80 days, and females appear to give birth throughout much of the year. Usually, one or two kittens are born, but litter sizes of up to four have been reported (Hall and Kelson 1959, Cahalane 1961, Morris 1965, Eaton 1977, Seager and Demorest 1978, Nowak and Paradiso 1983).

The ocelot was federally listed as endangered on July 21, 1982. Critical habitat has not been designated. Populations in many areas apparently continue to decline, and the species is threatened by habitat loss and fragmentation, exploitation for fur, and predator control.

Jaguar
The primary reference for this section is:
USFWS. 1997j. Final Rule to Extend Endangered Status for the Jaguar in the United States. Federal Register 62(140): 39147-39157.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS, Arizona Ecological Services Field Office, Phoenix, Arizona.

The jaguar (*Panthera onca*) is the largest species of cat native to the Western Hemisphere. Its range in North America includes Mexico and portions of the southwestern United States (Hall 1981). Jaguars are known from a variety of habitats (Seymour 1989, Nowak 1991). They show a high affinity to lowland wet habitats, typically swampy savannas or tropical rain forests. However, they also occur, or once did, in upland habitats in warmer regions of North and South America. Within the United States, jaguars have been recorded most commonly from Arizona, but there are also records from California, New Mexico, and Texas, and reports from Louisiana. Currently there is no known resident population of jaguars in the United States, though they still occur in northern Mexico. Nonetheless, there have been recent, confirmed records of jaguar in the United States from the New Mexico/Arizona border area and in southcentral Arizona, and the USFWS recognizes that the species continues to occur in the American Southwest, at least as an occasional wanderer from Mexico. The last survey for the species was in 1997, the same year in which two occurrences of jaguars were documented in Arizona (NatureServe Explorer 2002).

Jaguars breed year-round range-wide, but at the southern and northern ends of their range there is evidence for a spring breeding season. Gestation is about 100 days, and litters range from one to four cubs (usually two). Cubs remain with their mother for nearly 2 years. Females begin sexual activity at 3 years of age, males at 4. The list of prey taken by jaguars range-wide includes more than 85 species (Seymour 1989), such as peccaries (javelina), armadillos, turtles, and various birds and fish. Javelina and deer are presumably mainstays in the diet of jaguars in the United States and Mexico borderlands.

The jaguar was originally listed as endangered from the United States and Mexico border southward to include Mexico and Central and South America. On July 22, 1997, the jaguar was also listed as endangered in the United States. It was determined that the designation of critical habitat was not prudent. Loss and modification of the jaguar's habitat are likely to have contributed to its decline. While only a few individuals have been seen in the United States in recent years, the presence of the species in the United States is believed to be dependent on the status of the jaguar in northern Mexico. In the United States, a primary threat to this species is illegal shooting.

Effects of Vegetation Treatments on the Ocelot and Jaguar

Effects Common to All Treatment Methods

Direct and Indirect Effects. Populations of ocelots and jaguars generally occur south of the United States, in Mexico and South America, although there is a population of ocelots in Texas. In the project area, these species are only known to occur as scattered individuals in Arizona and New Mexico, which are believed to be transients from Mexico. Therefore, it is unlikely that any of the treatment methods would directly affect ocelots or jaguars. However, there is some chance that modification of lands near the Arizona and New Mexico southern borders could make them either more or less suitable for supporting these species.

Prescribed Fire Treatments

Direct Effects. A prescribed burn would be unlikely to directly affect ocelots or jaguars, since these species are unlikely to be found in the project area, and because they are large, mobile mammals that could easily move out of a burn area.

Indirect Effects. Because both species require brush or other forms of cover for foraging, prescribed fire could make habitat less suitable for them over the short term. However, there may also be some positive effects resulting from an increase in availability of prey species following the prescribed burn.

Mechanical Treatment Methods

Direct Effects. Vegetation treatment using mechanical methods would be unlikely to directly affect ocelots or jaguars, since these species are unlikely to occur in the project area, and could easily move out of the area where activities were occurring.

Indirect Effects. Removal of shrubs and brush from areas where cats have been observed in the past could make these areas less suitable for both species. However, since there are no known populations in the project area, these effects would not be great.

Manual Treatment Methods

Direct and Indirect Effects. Manual vegetation treatment methods would be unlikely to have any direct or indirect effects on ocelots or jaguars, or their habitats.

Biological Control Treatments

Domestic Animals

Indirect Effects. Domestic animals can improve habitat for these species by promoting the development of shrubs and brush, which they use for foraging. The USFWS (1990h) recommended controlled grazing as a possible management tool for the ocelot in certain areas of Texas. Although grazing promotes the development of shrub communities, these communities are often very disturbed habitats. Excessive levels of grazing may actually diminish brush regeneration and make the habitat unsuitable for ocelots and jaguars.

Other Biological Control Agents

Direct and Indirect Effects. The use of biological control agents to control non-native vegetation would be unlikely to have any direct or indirect effects on ocelots or jaguars, or their habitats.

Herbicides

Direct Effects. Herbicide treatments would be unlikely to directly affect to ocelots or jaguars. These species are unlikely to occur in the project area, and would likely move out of an area being treated by herbicides. Unintentional direct spray of these mammals is not anticipated. If such an exposure were to occur, adverse health effects could potentially occur as a result of direct spray by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, according to the ERAs (see Table 6-2). Adverse effects to these species could also occur if animals came into contact with foliage sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate.

Risk assessments indicated that adverse effects to ocelots or jaguars could potentially occur as a result of ingesting prey items sprayed by 2,4-D or diuron at the typical application rate, or by bromacil, diquat, or triclopyr at the maximum application rate (see Table 6-5). Forest Service risk assessments did not address the potential risks to carnivorous species as a result of ingesting prey contaminated by hexazinone. Therefore, the potential for adverse health effects to ocelots and jaguars as a result of exposure to hexazinone via this exposure pathway cannot be determined. It is unlikely, but possible, that a transient ocelot or jaguar could consume prey recently contaminated by 2,4-D, bromacil, diquat, diuron, hexazinone, or triclopyr, should one or more of these herbicides be used to treat vegetation on public lands in potential habitat for these species.

Indirect Effects. Herbicide treatments would be unlikely to affect ocelots or jaguars indirectly by modifying their habitat or prey populations, since they are at best transients on public lands.

Conservation Measures

The proposed vegetation treatments are unlikely to cause adverse effects to the ocelot or jaguar, or their habitats, since these species are unlikely to occur in the project area. However, the following conservation measures are suggested as extra precautions for areas in which recent sightings have occurred.

- Avoid using 2,4-D, bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr, where feasible.

Determination of Effects

Given the rarity of these species in the United States and the infrequency of sightings in Arizona and New Mexico where public lands occur, the proposed vegetation treatments would be **not likely to adversely affect** the ocelot or jaguar. However, at the local level, BLM offices should still include ocelots and/or jaguars in their vegetation management plans if the species have been observed in the project area in the past.

Sonoran Pronghorn

The primary reference for this section is:
USFWS. 1998w. Final Revised Sonoran Pronghorn Recovery Plan. Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Sonoran pronghorn (*Antilocapra americana sonoriensis*), one of five subspecies of pronghorn (Nowak and Paradiso 1983), inhabits southwestern Arizona in the U.S. and northwestern Sonora in Mexico. Two of seven identified subdivisions of the Sonoran desert encompass the habitat of this subspecies: the Lower Colorado River Valley and the Arizona Upland. Common plant species found in the Lower Colorado River Valley include creosote bush, white bursage, ironwood, blue palo verde, and mesquite. Common species in the Arizona upland include foothill palo verde, catclaw acacia, jumping cholla, and teddy bear cholla. Pronghorn appear to use flat valleys and isolated hills to a greater degree than other topographic features of the Sonoran Desert (Arizona Game and Fish Department 1985).

Washes flow briefly after rains during the monsoon season and after sustained winter rains. The network created by these washes provides important thermal cover for Sonoran pronghorn during the hot summer season. Drainages and bajadas are used during spring and summer, with bajadas used as fawning areas during the spring. Pronghorn appear to use palo verde, ironwood, and mesquite for cover. Playas provide abundant forbs during the spring, especially during good rain years. Pronghorn vacate these areas later in the season when forbs dry up (Hughes and Smith 1990). Some of the sandy areas provide a greater variety of seasonal vegetation. The openness of these areas appears to be attractive for pronghorn, as the annuals, grasses, and shrubs provide good forage species, particularly in the spring. These areas have long been considered important Sonoran pronghorn habitat in the U.S. However, the decreased palatability of annuals as summer approaches and a lack of sufficient woody vegetation for nutrition and thermal protection requirements drive pronghorns to bajada habitat in the southeast portion of the range by early summer.

The diet of Sonoran pronghorns consists of forbs, shrubs, cacti, trees, and grasses. Sonoran pronghorns drink minimal amounts of water, even when it is available. It is believed that water consumption varies inversely with the quantity and succulence of plants consumed (Beale and Smith 1970).

Pronghorn does become sexually mature at 16 months, and bucks become mature at 1 year of age (Kitchen and O'Gara 1974). Bucks congregate in the summer for breeding and to pursue females. Does break off from groups to search for fawning areas. Gestation is approximately 240 days, and fawns are born between February and May, and parturition appears to coincide with spring forage abundance. Does usually have twins, and fawns appear to suckle for about 2 months, feeding on vegetation soon after. Fawning areas have been documented in the Mohawk Dunes and the bajadas of the Sierra Pintas, Mohawk, Bates, and Growler mountain ranges.

The Sonoran pronghorn was federally listed as endangered on March 11, 1967. Critical habitat has yet to be designated for the subspecies. The decline of the species is attributable to a number of factors, including a lack of recruitment, insufficient forage and/or water, drought coupled with predation, difficulties for population expansion due to barriers to historical habitat, illegal hunting, degradation of habitat from livestock grazing, the diminishing size of the Gila and Sonoyta rivers, and human encroachment. Sonoran pronghorn numbers continue to decline. During a range-wide survey (completed in 2002), a total of 21 to 33 animals were estimated (Arizona Game and

Fish Department 2002). This number is down from estimates of 99 animals in 1999 and 142 animals in 2000. The drought of 2002 appears to have played a large part in this most recent decline in numbers.

Effects of Vegetation Treatments on the Sonoran Pronghorn

Effects Common to All Treatment Methods

Indirect Effects. The Sonoran pronghorn occurs in desert habitats, many of which have been impacted by non-native species. The invasion of exotics typically occurs at the expense of native plant species, including Sonoran pronghorn forage plants. Therefore, any treatment method that aids in returning native conditions to habitat should have a beneficial effect on the species. In addition, the removal of hazardous fuels from habitats that support pronghorns would be expected to reduce the likelihood of a future high-intensity wildfire. Such an unplanned and uncontrolled fire could consume large tracts of Sonoran pronghorn habitat, having a negative effect on species populations.

Sonoran pronghorns rely on riparian areas as habitat corridors. Therefore, removal of vegetation in these areas could have negative effects on pronghorns by reducing their ability to disperse from one habitat area to another. Individual treatment methods would vary in their potential to affect riparian areas. These potential effects are discussed in detail in Chapter 5.

Prescribed Fire Treatments

Direct Effects. There would be few direct effects to Sonoran pronghorns from prescribed fire, as these animals are large, highly mobile mammals that would typically be able to move out of a burn area with limited injury or mortality. Newborn fawns may be unable to escape fires, so fires during the breeding season could carry a higher risk of mortality than fires conducted during other seasons.

Indirect Effects. A prescribed fire would have temporary effects on habitat by reducing the amount of vegetation available for forage. Even the temporary destruction of pronghorn habitat could have an effect if there was limited suitable habitat outside the burn area. Although the long-term effects of fire would likely be positive, even short-term effects to rare species in fragmented habitats can have repercussions for these populations.

Burning of fuels would have a long-term positive effect on pronghorn habitat by reducing the likelihood of a future catastrophic wildfire. In addition, reducing the amount of vegetative cover in the area could have a beneficial effect, by reducing the number of potential predators. Coyotes, bobcats, mountain lions, and other predators often lurk unseen in areas with dense vegetation (USFWS 1998w). Finally, prescribed burns in areas near known pronghorn habitat may make these areas more suitable for future use by the species.

Mechanical Treatment Methods

Direct Effects. Noise from equipment and the presence of humans disturb pronghorn, and can lead pregnant females and those with newborns to leave fawning areas. Other pronghorns may also show some sensitivity to these disturbances.

Indirect Effects. Removal of large blocks of vegetation could temporarily reduce the amount of available pronghorn forage (succulent cacti, annuals, grasses, and shrubs), forcing pronghorns to seek food in less suitable habitat. On the other hand, thinning of vegetation and reducing the amount of vegetative cover could have a beneficial effect, by reducing the number of potential predators.

Manual Treatment Methods

Direct and Indirect Effects. There would be limited disturbance to Sonoran pronghorns from this type of treatment method. The presence of humans in fawning areas could cause some stress, as described above, but few other effects are anticipated.

Biological Control Treatments

Domestic Animals
Indirect Effects. Heavy grazing in deserts where the Sonoran pronghorn exists has caused a loss of suitable habitat for this species. Alteration of grasslands by domestic animals can affect both the quality and quantity of preferred forage that is needed to sustain healthy pronghorn herds (Ellis 1970; Howard et al. 1990). There is some speculation that livestock, sheep and pronghorns favor the same species of perennial grass, and that grazing by domestic animals may compete with or exclude Sonoran pronghorns. Therefore, use of domestic animals to contain weeds is likely to have a negative effect on pronghorn, with the severity of effects depending on the food needs of the grazer, the food resources in the area, and the intensity and duration of the treatment. Pregnant female pronghorns and those with newborns react easily to most forms of harassment, and have been observed to move out of fawning areas when cattle move in (USFWS 1998w).

Other Biological Control Agents
Direct and Indirect Effects. The release of biological control agents into Sonoran pronghorn habitat is unlikely to affect this species. There would be limited disturbance associated with the presence of humans, but it would be short term and minimal. Despite laboratory testing of approved biological control agents, there is always the chance that the release of a control agent might have an unforeseen negative effect on the entire ecosystem, although such an occurrence is not reasonably foreseeable.

Herbicides

Direct Effects. Although pronghorns would readily flee areas in which herbicide applications were occurring, it is possible that an accidental spray of Sonoran pronghorns could occur. Based on the results of the ERAs, adverse health effects to pronghorns could occur if animals were directly sprayed by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate (see Table 6-2). Since pronghorns are a large, readily visible species, the likelihood of an accidental direct spray is low. Pronghorns could also come into contact with sprayed foliage after the application. Via this exposure pathway, adverse health effects could occur if vegetation was sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate.

Risk assessments predicted that if Sonoran pronghorns were to ingest plant materials sprayed by 2,4-D, bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, Overdrive®, picloram, or triclopyr at the typical application rate, or by imazapyr, metsulfuron methyl, or tebuthiuron at the maximum application rate, adverse health effects could potentially occur (see Table 6-5). These predictions are overly conservative in that they assume all of the animal's diet would consist of contaminated vegetation, which is an unlikely, though not impossible, scenario.

Indirect Effects. Over the short term, herbicide treatments could reduce the cover of forage in pronghorn habitat. Over the long term, however, the quality of forage would improve, as non-native species would likely be less prevalent. Herbicide treatments in riparian areas used by pronghorns as corridors could reduce the overall plant cover in these areas, making them less desirable for use as corridors.

Conservation Measures

In order to prevent negative effects to the Sonoran pronghorn, the following conservation measures are required by the BLM:

- Prior to treatments, survey all suitable habitat in areas proposed for treatment for Sonoran pronghorns.
- Avoid biological treatment by domestic animals in areas used as forage by Sonoran pronghorns.
- Avoid fawning areas during treatments.
- Closely follow all application instructions and use restrictions on herbicide labels; in riparian habitats use only those herbicides that are approved for use in riparian areas.
- Avoid broadcast spraying herbicides in key pronghorn foraging areas.
- Do not use 2,4-D in Sonoran pronghorn habitats; do not broadcast spray 2,4-D within ¼ mile of Sonoran pronghorn habitat.
- Where feasible, avoid use of the following herbicides in Sonoran pronghorn habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, Overdrive®, picloram, tebuthiuron, and triclopyr.
- Do not broadcast spray bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, Overdrive®, picloram, or triclopyr in Sonoran pronghorn habitat; do not broadcast spray these herbicides in areas adjacent to Sonoran pronghorn habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying imazapyr, metsulfuron methyl, or tebuthiuron in or near Sonoran pronghorn habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, imazapyr, metsulfuron methyl, tebuthiuron, or triclopyr to vegetation in Sonoran pronghorn habitat, utilize the typical, rather than the maximum, application rate.

In addition, project-specific conservation measures will be applied at the local level, as necessary, to minimize effects to the species.

Determination of Effects

Assuming that any type of vegetation treatment could occur anywhere in Sonoran pronghorn habitat on public lands, the proposed action would be **likely to adversely affect** the Sonoran pronghorn and/or its habitat. However, by following the conservation measures discussed in the previous section, as well as any additional conservation measures deemed necessary during review of individual treatment projects at the local level, the BLM should be able to reduce or prevent impacts to the Sonoran pronghorn such that the vegetation treatments would be **not likely to adversely affect** the species or its habitat.

Small Wetland Mammals: Hualapai Mexican Vole, Amargosa Vole, Preble's Meadow Jumping Mouse, Riparian Woodrat, and Buena Vista Lake Ornate Shrew

Hualapai Mexican Vole

The primary reference for this section is:
USFWS. 1991c. Hualapai Mexican Vole Recovery Plan. Albuquerque, New Mexico.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Hualapai Mexican vole (*Microtus mexicanus hualpaiensis*) is a subspecies of the Mexican vole (*Microtus mexicanus*) that occurs in the Hualapai Mountains of Arizona. The subspecies has been found at elevations between approximately 5,400 and 8,400 feet, and is generally associated with woodland forest types containing grasses and grass-sedge habitats. Habitats tend to be dry, although when it is the only vole species present, it occurs in moister habitats as well (Spicer et al. 1985). The Hualapai vole is currently associated with moist grass-sedge areas along permanent or semi-permanent waters fed by springs or seeps in either open forest or chaparral. Good cover of grasses, sedges, and forbs is characteristic of this waterside vole habitat, which is usually found in narrow bands paralleling the watercourse.

Although there is little information on Hualapai vole food habitats, the diet of most vole species usually includes green plant material when available. It is likely that the Hualapai vole utilizes a typical vole diet of lush forbs and grasses. The subspecies has been observed during both day and night (Spicer et al. 1985), and is believed to be active year-round. Burrows and runways may be present within suitable habitat.

It is believed that the life history of the Hualapai vole is similar to that of other Mexican vole subspecies, which have small litters. Pregnant females are present from at least late spring through summer. Like other vole species, population levels may fluctuate on annual and perennial cycles. These cycles may correspond with precipitation and resulting growth of vegetation (Spicer et al. 1985).

The Hualapai Mexican vole was federally listed as endangered on November 2, 1987. Critical habitat has not been designated. It is assumed that when grassy and herbaceous habitats were more abundant in the Hualapai Mountains, the Hualapai vole was more common and widespread than it is today. In addition, the waterside habitats were once more extensive and interconnected than they are today. Grazing, mining, road construction, and recreational uses have contributed to the elimination and destruction of vole habitat in the Hualapai Mountains. At present, the primary threats to the vole and its habitat are grazing and recreation use and development. All remaining habitat areas are small and isolated from each other and are easily degraded by grazing, drought, and recreational use.

Amargosa Vole

The primary reference for this section is:
USFWS. 1997k. Amargosa Vole (*Microtus californicus scirpensis*) Recovery Plan. Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Amargosa vole (*Microtus californicus scirpensis*) is a desert subspecies of the widely distributed California vole (*Microtus californicus*) with a highly localized range in the central Mojave Desert of California. The subspecies has been found in isolated wetland habitats where bulrush is a dominant perennial overstory species. These wetlands form continuous bands along the Amargosa River, and are broken by more “characteristic” desert vegetation dominated by creosote bush, burro bush and desert holly. Perennial tributary spring sources interspersed along this section of the Amargosa River additionally create mesic habitat “islands” of cattails and bulrush, ranging in size from less than 1 to over 5 acres.

The historical range of the Amargosa vole apparently was limited to wetland pockets extending from the desert community of Shoshone, Inyo County, to the Amargosa Canyon, Inyo County, California. The largely subterranean Amargosa River and an associated series of small tributary springs maintain an isolated 10-linear mile stretch of perennial surface water. The current distribution of the subspecies extends discontinuously from a tributary spring site located in the Section 33, Township 21 North Range 7 East, and Section 15, Township 20 North Range 7 East. Within this range, the distribution of the vole appears to coincide principally with isolated bulrush-cattail pockets that are not subjected to regular inundation during heavy summer thunderstorms.

Little is known about the life history of the Amargosa vole; however, it is probably similar to that of the California vole, which is described below.

Voles are primary consumers and often the principal herbivores within occupied habitats (Rose and Birney 1985). They may excavate an extensive underground network of runways and tunnels (Wolff 1985), and in dense cover frequently develop extensive surface runways (Taitt and Krebs 1985). The inability to concentrate urine and conserve water is a major reason for the vole’s distributional restriction to mesic and wetland habitats (Getz 1985). Voles lack physiological or morphological characteristics that would allow them to tolerate high temperatures (Rose and Birney 1985). Therefore, they require a regular intake of large amounts of water, meeting or exceeding 10% of body weight per day (Batzli and Pitelka 1971).

California voles are active throughout the year. Activity usually occurs in daylight hours during winter months, although animals my become crepuscular and nocturnal through the summer (Madison 1985). The main food items consumed by voles are grasses and forbs, as well as seeds (Heske et al. 1984). When seasonally available, green emergent vegetation comprises the bulk of the diet; grass seeds predominate in the diet during the summer and autumn (Batzli and Pitelka 1971).

Reproduction may occur at any time of year, but is primarily influenced by factors, such as temperature and precipitation, that determine the availability of food and water (Hoffman 1958, Seabloom 1985). In central California, vole populations peak during the spring and begin declining in late summer (Hoffmann 1958). Reproductive maturity is reached when females attain a weight of 0.9 to 1.1 ounces and males a weight of 1.2 to 1.4 ounces. Vole nests are composed of dried grass and may be placed above or below ground (Wolff 1985). In central California, litter size increases from about three at the beginning of the breeding season in the fall, to a peak of about six in the spring (Hoffman 1958). Mean litter size for the species is 4.7 (Nadeau 1985). Young are born after a gestation of 21 days, and are weaned after 14 days. California vole populations are subject to booms and crashes on a 2- to 4-year cycle.

The Amargosa vole was federally listed as endangered with critical habitat on November 15, 1984. Critical habitat for the species encompasses an area of 4,520 acres in southeastern Inyo County, California. Within critical habitat areas the major constituent elements that are known to require special management consideration or protection are marsh vegetation (primarily bulrushes), springs, and some open water along the Amargosa River, which provide escape cover and an adequate food supply. Reasons for listing this subspecies included loss of historical habitat, rechannelization of water sources needed to perpetuate habitats, and pumping of groundwater. Threats to the Amargosa vole include diversion of surface or groundwaters, intermittent flooding, and introduction of exotic plant and wildlife species.

Preble's Meadow Jumping Mouse

The primary reference for this section is:
USFWS. 2002m. Designation of Critical Habitat for the Preble's Meadow Jumping Mouse (*Zapus hudsonius preblei*). Federal Register 67(137): 47153-47210.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Colorado Ecological Services Field Office, Lakewood, Colorado.

Preble's meadow jumping mouse (*Zapus hudsonius preblei*) is found along the foothills in southeastern Wyoming, southward along the eastern edge of the Front Range of Colorado to Colorado Springs, El Paso County (Hall 1981; Clark and Stromberg 1987; Fitzgerald et al. 1994). The subspecies is likely an Ice Age relict (Hafner et al. 1981; Fitzgerald et al. 1994) that was confined to riparian systems where moisture was more plentiful after the glaciers receded from the Front Range of Colorado and the foothills of Wyoming and the climate became drier. The semi-arid climate in southeastern Wyoming and eastern Colorado limits the extent of riparian corridors and restricts the range of the Preble's meadow jumping mouse in this region. The eastern boundary for the subspecies is likely defined by the dry shortgrass prairie, which may present a barrier to eastward expansion (Beauvais 2001). The western boundary of Preble's range in both states appears related to elevation along the Laramie Range and Front Range; the general upward limit of the subspecies' habitat in Colorado is 7,600 feet (USFWS 1998).

Typical habitat for the Preble's meadow jumping mouse comprises well-developed plains riparian vegetation with adjacent, undisturbed grassland communities and a nearby water source. Well-developed plains riparian vegetation typically includes a dense combination of grasses, forbs, and shrubs; a taller shrub and tree canopy may be present (Bakeman 1997). When present, the shrub canopy is often willow, although shrub species including snowberry, chokecherry, hawthorn, gambel oak, gray alder, river birch, skunkbrush, wild plum, lead plant, red-osier dogwood, and others also may occur (Bakeman 1997, Shenk and Eussen 1998). Preble's meadow jumping mice regularly use uplands at least as far out as 330 feet beyond the 100-year floodplain for feeding and resting (Ryon 1999, Shenk 2002). The subspecies can also move considerable distances along streams, as far as 1 mile in one evening (Ryon 1999, Shenk and Sivert 1999a).

The abundance of Preble’s meadow jumping mice at a given location is not likely to be driven by the diversity of plant species, but by the density of riparian vegetation. The tolerance of the Preble's for exotic plant species is not well understood. However, there is particular concern about non-native species such as leafy spurge that may form a monoculture, displacing native vegetation and thus reducing available habitat.

The Preble’s meadow jumping mouse constructs day nests composed of grasses, forbs, sedges, rushes, and other available plant material. They may be globular in shape or simply raised mats of litter, and are most commonly above ground, but also can be below ground. They are typically found under debris at the base of shrubs and trees, or in open grasslands (Ryon 2001). An individual mouse can have multiple day nests in both riparian and grassland communities (Shenk and Sivert 1999a), and may abandon a nest after approximately a week of use (Ryon 2001). Hydrologic regimes that support Preble’s habitat range from large perennial rivers such as the South Platte River to small temporary drainages only 3 to 10 feet in width, as at Rocky Flats and in montane habitats. Flooding is a common and natural event in the riparian systems along the Front Range of Colorado. This periodic flooding helps create a dense vegetative community by stimulating resprouting from willow shrubs and allows herbs and grasses to take advantage of newly-deposited soil. Fire is also a natural component of the Colorado Front Range and Wyoming foothills, and Preble’s meadow jumping mouse habitat naturally decreases in size in response to a fire event, and then increases in size again until the next fire. Within shrubland and forest, intensive fire may result in adverse impacts to Preble’s populations. However, grassland fires on small mammals may have little effect, or even a positive effect.

Preble’s meadow jumping mice eat insects; fungus; moss; pollen; willow; lamb’s quarters; Russian thistle; sunflowers; sedge; mullein; brome, fescue, bluegrass, dropseed and wheatgrass; bladderpod; scouring rush; and assorted seeds (Shenk and Eussen 1998, Shenk and Sivert 1999a). The diet shifts seasonally, consisting primarily of insects and fungus after emerging from hibernation, and shifting to fungus, moss, and pollen during mid-summer (July to August), with insects again added in September (Shenk and Sivert 1999a). The shift in diet along with shifts in mouse movements suggests that the Preble’s meadow jumping mouse may require specific seasonal diets, perhaps related to the physiological constraints imposed by hibernation. The Preble’s meadow jumping mouse is a true hibernator, usually entering hibernation in September or October and emerging the following May, after a potential hibernation period of 7 or 8 months. Adults are the first age group to enter hibernation because they accumulate the necessary fat stores earlier than young of the year. Similar to other subspecies of meadow jumping mouse, Preble’s meadow jumping mouse do not store food, but survive on fat stores accumulated prior to hibernation (Whitaker 1963). The Preble’s meadow jumping mouse is primarily nocturnal or crepuscular but also may be active during the day, when they have been seen moving around or sitting still under a shrub (Shenk 1998).

Preble’s meadow jumping mice usually have two litters per year, but there are records of three litters per year. An average of five young are born per litter, but the size of a litter can range from two to eight young (Quimby 1951, Whitaker 1963). The Preble’s meadow jumping mouse is long-lived for a small mammal, in comparison with many species of mice and voles that seldom live a full year. However, like many small mammals, the subspecies’ annual survival rate is low. The Preble’s meadow jumping mouse has a host of known predators including garter snakes, prairie rattlesnakes, bullfrogs, red and grey foxes, house cats, long-tailed weasels, and red-tailed hawks (Shenk and Sivert 1999a, Schorr 2001). Other potential predators include coyotes, barn owls, great horned owls, screech owls, long-eared owls, northern harriers, and large predatory fish. Other mortality factors of the Preble’s meadow jumping mouse include drowning and vehicle collision. Mortality factors known for the meadow jumping mouse, such as starvation, exposure, disease, and insufficient fat stores for hibernation, also are likely causes of death for the Preble’s meadow jumping mouse (Whitaker 1963).

The Preble’s meadow jumping mouse was federally listed as threatened on May 13, 1998. On July 3, 2003, the USFWS proposed to designate approximately 57,446 acres found along 657.5 miles of rivers and streams in Colorado and Wyoming as critical habitat. The Preble’s meadow jumping mouse is closely associated with riparian ecosystems that are relatively narrow and represent a small percentage of the landscape. If habitat for the Preble’s meadow jumping mouse is destroyed or modified, populations in those areas will decline or be extirpated. Thus, the decline in the extent and quality of Preble’s meadow jumping mouse habitat is considered the main factor threatening the subspecies (Hafner et al. 1998; Shenk 1998; USFWS 1998). Habitat alteration, degradation, loss, and fragmentation resulting from urban development, flood control, water development, agriculture, and other human land uses have adversely impacted Preble's meadow jumping mouse populations.

Riparian Woodrat

The primary reference for this section is:
USFWS. 1998h. Recovery Plan for Upland Species of the San Joaquin Valley. Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citation have been included in the Bibliography.

The riparian woodrat (*Neotoma fuscipes riparia*), also called the San Joaquin Valley woodrat, is one of 11 subspecies of the dusky-footed woodrat (Hooper 1938). There is currently only one known population of the riparian woodrat, which occurs in riparian forest on the Stanislaus River in Caswell Memorial State Park, California. Riparian woodrats are found in stands of deciduous valley oaks, and are most numerous where shrub cover is dense and least abundant in open areas. The highest densities of woodrats are often encountered in willow thickets with an oak overstory (Linsdale and Tevis 1951). For the most part, woodrats are generalist herbivores, consuming a wide variety of nuts, fruits, fungi, and foliage, as well as some forbs.

Riparian woodrats make large houses out of sticks and other litter, which range from 2 feet to 5 feet in height, and from 4 feet to 8 feet in basal diameter. These houses are typically placed in the ground, against or straddling a log or exposed roots of a standing tree, and are often located in dense brush. Nests are also placed in the crotches and cavities of trees and in hollow logs.

Riparian woodrats live in loosely-cooperative societies, and a maternal-based social structure (Kelly 1990). Whereas adjacent females are usually closely related, males disperse away from their birth den and are highly territorial and aggressive, especially during the breeding season. The effective population size (i.e., successful breeders) of riparian woodrats is generally much smaller than the actual population size.

The riparian woodrat was federally listed as endangered on February 23, 2000. Critical habitat has not been designated. As for many species occurring in the San Joaquin Valley of California, loss and fragmentation of habitat are the principal reasons for the decline of the riparian woodrat. The remaining population of the species is at an increased risk for extinction because of its small size. In addition, the population is vulnerable to flooding of the Stanislaus River, which can severely damage woodrat houses.

Buena Vista Lake Ornate Shrew

The primary reference for this section is:
USFWS. 1998h. Recovery Plan for Upland Species of the San Joaquin Valley. Portland, Oregon.

and

USFWS. 2005. Final Rule to Designate Critical Habitat for the Buena Vista Lake Shrew (*Sorex ornatus relictus*). Federal Register 70(14): 3438-3461.

References cited in this section are internal to USFWS 1998, referenced above. Full citations have been included in the Bibliography.

The Buena Vista Lake ornate shrew (*Sorex ornatus relictus*) is one of nine subspecies of the ornate shrew (*Sorex ornatus*; Merriam 1895, Hall 1981, Junge and Hoffman 1981). The species formerly occurred in wetlands around Buena Vista Lake, and presumably throughout the Tulare Basin (Grinnell 1932, 1933; Williams and Kilburn 1984; Williams 1986), but little is known about its current distribution. The shrew is known to occur in areas with dense wetland vegetative cover and an abundant layer of decomposed vegetation (Center for Conservation Biology 1990, Maldonado 1992), and like other ornate shrews may be more associated with the structure of the vegetation in suitable habitats, rather than the plant species composition. The dominant plant species present in areas where the shrew has recently been captured include Fremont cottonwood, willows, glasswort, alkali heath, wild-rye grass, and Baltic rush. Although the specific feeding and foraging habits of Buena Vista Lake ornate shrews are not known, it is likely that, like other shrews, they feed on insects and other invertebrates (Harris 1990, Maldonado 1992). Shrews are thought to burrow (Rudd 1953) and utilize the burrows of other animals (Pearson 1959).

Although details of the shrews reproduction and mating are not known, it is believed that the breeding season for this subspecies may begin in autumn and end when the dry season begins in May or June. In areas where wetlands do not dry up during the dry season, the breeding season may last longer (Center for Conservation Biology 1990). Up to two litters, each containing four to six young, are produced per year (Owen and Hoffman 1983).

The Buena Vista Lake ornate shrew was listed as endangered on March 6, 2002. On January 24, 2005, the USFWS designated approximately 84 acres within the Central Valley floor of Kern County, California, as critical habitat for the species. The major causes for the decline of the shrew are loss and fragmentation of habitat as a result of converting lands to agriculture and diverting fresh water supplies (Williams and Kilburn 1984, 1992). It is believed that there may be a single remaining population of this subspecies, existing in a very small area of suitable habitat. Threats to the species include natural or human-caused changes to the remaining habitat for this species, selenium poisoning, and a lack of other viable populations in the area to recolonize the site should an extirpation of the population occur.

Effects of Vegetation Treatments on the Hualapai Mexican Vole, Amargosa Vole, Preble's Meadow Jumping Mouse, Buena Vista Lake Ornate Shrew, and Riparian Woodrat

Effects Common to All Treatment Methods

Indirect Effects. Any treatment that removes vegetation from the habitat of these four species would be expected to have short-term negative effects. All three of these mammals rely on dense vegetation for cover from predators, even utilizing habitat with a large component of non-native plant species. The Preble's meadow jumping mouse, for example, utilizes habitats where Canada thistle, toadflax, and smooth brome are present (Colorado Natural Heritage Program 1999).

Nonetheless, removal of non-native species would have long-term benefits. In riparian and wetland habitats, where these species occur, invasion and spread of non-native plant species typically results in degraded habitat. In the case of the Amargosa vole, the invasion of habitat by tamarisk is a threat because it displaces native plant species, such as bulrush, and because salt exudation from tamarisk leaves reduces the prevalence of a lower canopy flora, on which voles rely for food and cover (USFWS 1997i). Tamarisk also contributes to water loss, and can reduce the coverage of wetlands. Russian olive and knapweed are non-native species that can potentially replace critical components of Preble's meadow jumping mouse habitat (Colorado Natural Heritage Program 1999). Treatments that reduce the coverage of non-native plant species could also increase the suitability of wetland and riparian areas within the species' ranges, potentially increasing the acreage of habitat available for these species in the future.

Treatments that reduce the presence of hazardous fuels in the habitat of these three mammal species would provide long-term benefits by reducing the likelihood of a future severe wildfire. Such a catastrophic occurrence would have the capability of eliminating a large percentage of the population of the Hualapai Mexican vole, the Amargosa vole, the Preble's meadow jumping mouse, the Buena Vista Lake ornate shrew, or the riparian woodrat.

Prescribed Fire Treatments

Direct Effects. A prescribed fire would likely result in some mortality, although small mammals are typically able to hide in moist litter, stump and root holes, and other sheltered spaces to avoid the burn (Ford et al. 1999 *cited in* Smith 2000).

Indirect Effects. Fire has been labeled as both a natural component of the habitat of these species and as a threat to the remaining habitat. There is evidence that changes in watersheds and fire management policies have altered the role of fire in wetland ecosystems that provide habitat for these small mammal species. Given the limited coverage and isolated nature of existing habitat and the small population size of these species, fire suppression may be required to ensure that a wildfire does not eliminate existing populations. Therefore, prescribed fire could be detrimental as well.

The alteration of habitat by fire could make it temporarily unsuitable for these small mammal species, causing them to leave the burned area for a few growing seasons until vegetation and a litter layer return. By destroying aboveground vegetation, fire decreases food availability and protection from predation. Fire would also be expected to destroy riparian woodrat homes, which are constructed of sticks and litter. For these endangered species, for which limited habitat remains, temporarily dispersing from a burned area may not be feasible if the burned area is large in size. Therefore, a fire in remaining habitat could potentially lead to extirpation of the species.

Mechanical Treatment Methods

Direct Effects. There could be some direct mortality to these small mammals as a result of being crushed by heavy vehicles or other equipment.

Indirect Effects. Widespread removal of vegetation could be detrimental to species populations for the reasons described above. Mechanical treatments would also be likely to destroy riparian woodrat homes. In addition, mechanical control in riparian areas, and in upland areas adjacent to wetland habitats may contribute to erosion and sedimentation, degrading the quality of habitat. Mechanical control methods that alter the structural qualities of habitat may alter essential behavior patterns of voles, shrews, mice, or woodrats, perhaps reducing their success over the short term. Nonetheless, successful removal of invasive plant species such as tamarisk and Russian olive would benefit these species over the long term by halting the degradation of habitat.

Manual Treatment Methods

Direct and Indirect Effects. Use of manual control to reduce small populations of weeds and other undesirable vegetation is unlikely to have major adverse effects on these four endangered species or their habitat. The disturbance to wetland habitats by this treatment method would be minimal. The resulting reduction in populations of non-native plant species would have positive effects on vole, mouse, shrew, and woodrat habitat.

Biological Control Treatments

Domestic Animals

Indirect Effects. Grazing has been identified as a threat to these three species, as well as to their habitats. Because these species are found wetlands and riparian areas, their habitat is easily degraded by trampling by domestic animals. Furthermore, the lush vegetation that is present in these areas is a preferred food of grazers, and therefore attracts domestic animals to the area. The resulting trampling and overgrazing removes food for the voles and mice, and reduces their ability to hide from predators. Domestic animals may also reduce water levels by drinking, and their waste products may contaminate the water.

Other Biological Control Agents
The release of other biological control agents would be unlikely to affect the Hualapai Mexican vole, the Amargosa vole, the Preble's meadow jumping mouse, the riparian woodrat, or the Buena Vista Lake ornate shrew. Disturbances associated with releasing these agents would be minimal. However, since there is limited knowledge about the long-term effects of these agents, there is a chance that their release could result in unanticipated impacts to these species.

Herbicides

Direct Effects. Use of ATVs, trucks, or horses to apply herbicides could cause some mortality or injury to these small TEP mammals as a result of crushing. Since the Hualapai Mexican vole, Amargose vole, Preble's meadow jumping mouse, riparian woodrat, and Lake Buena Vista ornate shrew all utilize vegetation for cover from predators, and may have aboveground nests, it is conceivable that some animals could be sprayed inadvertently during herbicide treatments. Based on the results of the ERAs, direct spray by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, could potentially result in adverse effects to small mammals (see Table 6-2). Furthermore, if mammals were to come into contact with vegetation that had been sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate, adverse effects could potentially occur. Therefore, it is assumed that use of these herbicides in habitats that support the Hualapai Mexican vole, Amargosa vole, Preble's meadow jumping mouse, riparian woodrat, or Buena Vista Lake ornate shrew could have adverse effects on populations of these species

If voles, mice, shrews, or woodrats were to ingest plant materials sprayed by 2,4-D, diquat, or diuron at the typical application rate, or by bromacil, fluridone, glyphosate, hexazinone, or tebuthiuron, at the maximum application rate, adverse health effects could potentially occur (see Table 6-5). If mice or shrews were to ingest insects sprayed by 2,4-D, clopyralid, diquat, glyphosate, hexazinone, imazapyr, picloram, or triclopyr at the typical application rate, or by diuron or metsulfuron methyl at the maximum application rate, adverse health effects would be possible. These scenarios assume that 100% of the animal's diet consists of contaminated food items, which is unlikely.

Indirect Effects. Herbicide treatments in mouse, vole, shrew or woodrat habitat could reduce vegetative cover, temporarily exposing animals to increased predation. In addition, the availability of food could be reduced temporarily. Use of trucks or ATVs could also crush aboveground nests present on the treatment site. Treatments would also help to maintain or improve the quality of wetland habitats, which would likely benefit these species over the long term.

Conservation Measures

In order to avert or minimize potential adverse effects to the Hualapai Mexican vole, the Amargosa vole, the Preble's meadow jumping mouse, the riparian woodrat and the Buena Vista Lake ornate shrew, the following conservation measures would be required:

- Survey suitable habitat for these species prior to developing treatment programs at the local level.

In areas where Preble's meadow jumping mouse occurs:
- Create a 300-foot buffer from the exterior boundary of the 100-year floodplain.
- Within the floodplain and the 300-foot buffer, do not allow ground disturbance from vehicles, except for access routes.
- Within the floodplain and the 300-foot buffer, use existing roads. Where existing roads are unavailable, limit the number of crossing access routes. These routes should be located in sites with little vegetation.
- At the completion of the project, revegetate access routes with native seed.
- Within the 300-foot buffer, conduct prescribed burns and broadcast spray of herbicides only during the hibernation period.
- Avoid damaging the shrub and tree components within the buffer at all times.

- Avoid weed control activities from August through October to minimize the potential loss of seed crop during the critical pre-hibernation period.
- Closely follow all application instructions and use restrictions on herbicide labels; in wetland and riparian habitats use only herbicides that are approved for use in those areas.
- Within the 300-foot buffer, broadcast spray herbicides only during the hibernation period.
- Do not use 2,4-D, diquat, or diuron within the 300-foot buffer; do not broadcast spray these herbicides within ¼ mile of the buffer.
- Where feasible, avoid use of the following herbicides in or near Preble's meadow jumping mouse habitat: bromacil, clopyralid, fluridone, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, tebuthiuron, and triclopyr.
- Outside of the hibernation period, do not broadcast spray clopyralid, glyphosate, hexazinone, imazapyr, picloram, or triclopyr in areas adjacent to Preble's jumping mouse habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, fluridone, metsulfuron methyl, or tebuthiuron near Preble's meadow jumping mouse habitat outside the hibernation period, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of bromacil, glyphosate, hexazinone, tebuthiuron, or triclopyr to vegetation in Preble's meadow jumping mouse habitat, utilize the typical, rather than the maximum, application rate.

In areas where the Hualapai Mexican vole, Amargosa vole, riparian woodrat, or Buena Vista Lake ornate shrew occur:

- Address Hulalpai Mexican voles, Amargosa voles, riparian woodrats, and Buena Vista Lake ornate shrews in all management plans prepared for treatments within areas that contain habitat for these species.
- Do not burn, graze, or conduct mechanical treatments within wetlands and/or riparian areas that support these species.
- Do not burn in areas where woodrat homes are present.
- Use manual spot application of herbicides rather than broadcast treatments.
- Closely follow all application instructions and use restrictions on herbicide labels; in wetland and riparian habitats use only herbicides that are approved for use in those areas.
- Do not use 2,4-D, diquat, or diuron in Hualapai Mexican vole, Amargosa vole, or riparian woodrat habitats; do not broadcast spray these herbicides within ¼ mile of Hualapai Mexican vole, Amargosa vole, or riparian woodrat habitat.
- Do not broadcast spray herbicides within Hualapai Mexican vole, Amargosa vole, riparian woodrat, or Buena Vista Lake ornate shrew habitat.
- Where feasible, avoid use of the following herbicides in Hualapai Mexican vole, Amargosa vole, riparian woodrat, and Buena Vista Lake ornate shrew habitat: clopyralid, fluridone, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, tebuthiuron, and triclopyr.
- Do not broadcast spray clopyralid, glyphosate, hexazinone, imazapyr, picloram, or triclopyr in areas adjacent to Hualapai Mexican vole, Amargosa vole, riparian woodrat, or Buena Vista Lake ornate shrew habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, fluridone, metsulfuron methyl, or tebuthiuron near Hualapai Mexican vole, Amargosa vole, riparian woodrat, or Buena Vista Lake ornate shrew habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of bromacil, glyphosate, hexazinone, tebuthiuron, or triclopyr to vegetation in Hualapai Mexican vole, Amargosa vole, riparian woodrat, or Buena Vista Lake ornate shrew habitat, utilize the typical, rather than the maximum, application rate.

These measures represent the minimum that is required of the BLM to protect these species from adverse impacts during vegetation treatments. Additional project-specific conservation measures would also need to be developed at the local level, as appropriate.

Determination of Effects

Under the assumption that any vegetation treatment could occur anywhere on public lands, the proposed action would be **likely to adversely affect** the Hualapai Mexican vole, the Amargosa vole and/or its critical habitat, the Preble's meadow jumping mouse, the riparian woodrat, and the Buena Vista Lake ornate shrew. However, the effects determination could be reduced to **not likely to adversely affect** if the conservation measures presented in the preceding section were followed, as well as any additional conservation measures identified at the project level.

Northern Idaho Ground Squirrel

The primary reference for this section is:
USFWS. 2000l. Determination of Threatened Status for the Northern Idaho Ground Squirrel. Federal Register 65(66): 17779-17786.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Snake River Basin Office, Boise, Idaho.

The Northern Idaho ground squirrel (*Spermophilus brunneus brunneus*) has one of the smallest ranges of any North American mainland mammal (Gill and Yensen 1992), and is known from 36 sites in Adams and Valley counties, Idaho. Populations occur at elevations ranging from 3,800 to 5,200 feet, and are associated with xeric meadows surrounded by ponderosa pine and Douglas-fir forest. However, the ground squirrel is not abundant in meadows that are surrounded by high densities of small young trees (Sherman and Yensen 1994). Soil texture and depth can be a primary factor in determining species distribution for the northern Idaho ground squirrel (Brown and Harney 1993). The subspecies often digs burrows under logs, rocks, or other objects (Sherman and Yensen 1994). Dry vegetation sites with shallow soil horizons of less than 20 inches depth above basalt bedrock to develop burrow systems are preferred (Yensen et al. 1991). Nesting burrows are found in well-drained soils greater than 3 feet deep, in areas not covered with trees or used by Columbian ground squirrels.

The northern Idaho ground squirrel eats small seeds and grain seasonally, and ingests large amounts of bluegrass and other grass seeds to store energy for the winter. The subspecies will also consume the roots, bulbs, leaf stems, and flower heads of another 45 to 50 plant species that are major components of the diet during key periods of the spring and summer.

The northern Idaho ground squirrel emerges in late March or early April, remains active above ground until late July or early August (Yensen 1991), and spends the rest of the year in hibernation underground (Yensen 1999). Seasonal torpor (a state of sluggishness or inactivity) generally occurs in early to mid-July for males and females, and late July to early August for juveniles. The subspecies normally becomes reproductively active within the first 2 weeks of emergence (Yensen 1991). Females that survive the first winter live, on average, nearly twice as long as males (3.2 years for females and 1.7 years for males). During the mating period, males move considerable distances in search of receptive females for mating, and often fight with other males for copulations, thereby exposing themselves to predation by raptors.

The northern Idaho ground squirrel was federally listed as threatened on April 5, 2000. Critical habitat for the squirrel was deemed prudent, but has not yet been designated. The subspecies is primarily threatened by habitat loss resulting from forest encroachment into former suitable meadow habitats. Such forest encroachment results in habitat fragmentation, eliminates dispersal corridors, and restricts the squirrel's population into small, isolated habitat areas. The northern Idaho ground squirrel is also threatened by competition from the larger Columbian ground squirrel, land use changes, recreational shooting, poisoning, and naturally occurring events. The current population of the Northern Idaho ground squirrel is approximately 500 individuals (Idaho Fish and Game 2003).

Effects of Vegetation Treatments on the Northern Idaho Ground Squirrel

Effects Common to All Treatment Methods

Indirect Effects. A large portion of the diet of the northern Idaho ground squirrel consists of seeds from native bunchgrasses. These fire-resistant plants, in the absence of fire, have been overwhelmed by non-native invaders, which are a poorer food source for the squirrel, and which degrade the quality of habitat. Therefore, any treatment method that reduces the coverage of non-native species occurring within squirrel habitat would be expected to have a long-term beneficial effect on the species. In addition, removal of non-native species or clearing/thinning of trees in already-degraded habitats where the squirrel no longer occurs may increase their suitability for supporting this species in the future. Thus, treatment methods could also potentially increase the acreage of northern Idaho ground squirrel habitat. Furthermore, fuels reduction activities would decrease the likelihood of a high intensity fire that would severely destroy existing habitat and result in extensive squirrel mortality.

Prescribed Fire Treatments

Direct Effects. Some injury or mortality of squirrels could occur during a prescribed fire. Squirrels are more likely to seek escape cover than to flee the site (Smith 2000). Where safe escape sites exist, the likelihood of mortality would be lowest. Prescribed fires occurring during the 8-month hibernation season, from August through March, would potentially have the lowest risk of directly affecting squirrels.

Indirect Effects. The northern Idaho ground squirrel is very dependent on fire. Because squirrels occur in open meadows and shrub/grasslands among coniferous forests, conifer invasion into these habitats resulting from fire suppression has been identified as the primary threat to the species. Therefore, fire would have a long-term positive effect on squirrel habitat. Prescribed fire has been used to improve habitat for this species in the past, and has typically consisted of removing small trees to reduce the fuel load and then burning the site during the squirrels' hibernation period. In the current fragmented habitat, however, squirrels are unable to migrate from one area of suitable habitat to another. Therefore, immediately following a burn, when plant resources are low, squirrels may be unable to migrate to a more suitable habitat through the dense stands of conifers surrounding them.

Mechanical Treatment Methods

Direct Effects. Vehicles and equipment used during treatments could harm or kill squirrels by crushing them. However, treatments completed during the hibernation period would have less of a likelihood of encountering squirrels.

Indirect Effects. Heavy equipment used during mechanical treatments might cause some disturbances to habitat by eroding the soil and potentially crushing burrows. Nonetheless, mechanical methods, including basic timber harvest, have helped improve/increase habitat for the northern Idaho ground squirrel. Mechanical methods can be used to control the encroachment of conifers onto meadow habitats and other openings, reduce the fuel loading, increase the amount of available habitat, and potentially create migration corridors for the species.

Manual Treatment Methods

Direct and Indirect Effects. Manual control methods would be unlikely to adversely affect northern Idaho ground squirrels or their habitat. Disturbances associated with this treatment method would be minimal.

Biological Control Treatments

Domestic Animals

<u>Indirect Effects</u>. Introduction of domestic animals onto northern Idaho ground squirrel habitat could have some positive effects by controlling the encroachment of conifers into meadow habitats. However, some negative effects would also be likely, and would be increasingly likely as the intensity of the treatment increased. Since squirrels

are herbivores that favor bunchgrasses, competition between this species and domestic animals is likely. In addition, domestic animals can facilitate the spread of non-native species on a site. Grazers could also potentially damage burrows, especially if large numbers of animals were allowed onto northern Idaho ground squirrel habitat at the same time.

Other Biological Control Agents
Direct and Indirect Effects. Biological control methods would be unlikely to adversely affect northern Idaho ground squirrels or their habitat. Disturbances associated with the release of agents into habitat would be minimal. However, there is always the chance that an unforeseen negative effect associated with the release of approved biological control agents could occur in the ecosystem.

Herbicides

Direct Effects. During herbicide treatments, any northern Idaho ground squirrels unable to flee the area could be inadvertently injured or killed by trucks or ATVs used to apply herbicides. Although direct spray of northern Idaho ground squirrels is unlikely, it could potentially occur during herbicide treatment programs in the species' habitat outside of the hibernation period. Based on the results of the ERAs, direct spray of 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or of imazapyr or metsulfuron methyl at the maximum application rate, could pose a health risk to the species (see Table 6-2). In addition, if ground squirrels were to come into contact with foliage that had been sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate, adverse health effects could potentially occur.

Consumption of plant materials that have been treated by 2,4-D, diquat, or diuron at the typical application rate, or by bromacil, fluridone glyphosate, hexazinone, or tebuthiuron at the maximum application rate, would potentially pose health risks to northern Idaho ground squirrels, based on the results of ERAs for these chemicals (see Table 6-5). These predictions represent the maximum potential for risk, since they assume the animal's entire diet consists of contaminated vegetation, which would be unlikely.

Indirect Effects. Herbicide treatments in northern Idaho ground squirrel habitat could affect populations if the cover of native bunchgrasses were reduced. Over the long term, however, herbicide treatments would likely improve ground squirrel habitat, since native bunchgrasses would benefit from the removal of weeds.

Conservation Measures

Implementation of the following conservation measures would ensure that the BLM's activities would not adversely affect the Northern Idaho ground squirrel.

- Prior to conducting treatments, survey the area to be treated for northern Idaho ground squirrels.
- At the local level, address northern Idaho ground squirrels and their habitat when developing management plans for proposed treatments.
- Where squirrels are detected, conduct vegetation treatments during the hibernation season, where feasible.
- Prohibit or minimize use of domestic animals in squirrel habitats.
- Design treatments so that only a portion of northern Idaho ground squirrel habitat is in a state of recovery at any one time.
- Design treatments to avoid injury to native bunchgrasses in northern Idaho ground squirrel habitat; consult plant buffer distances and other conservation measures for sensitive plants in Chapter 4 for guidance.
- Do not use 2,4-D, diquat, or diuron in northern Idaho ground squirrel habitats outside of the hibernation period; do not broadcast spray these herbicides within ¼ mile of northern Idaho ground squirrel habitat outside the hibernation period.
- Where feasible, avoid use of the following herbicides in northern ground Idaho squirrel habitat: bromacil, clopyralid, fluridone, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, tebuthiuron, and triclopyr.

- Do not broadcast spray clopyralid, glyphosate, hexazinone, picloram, or triclopyr in northern Idaho ground squirrel habitat outside of the hibernation period; do not broadcast spray these herbicides in areas adjacent to northern Idaho ground squirrel habitat outside of the hibernation period under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, imazapyr, fluridone, metsulfuron methyl, or tebuthiuron in or near northern Idaho ground squirrel habitat outside of the hibernation period, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of bromacil, glyphosate, hexazinone, tebuthiuron, or triclopyr to vegetation in northern Idaho ground squirrel habitat outside of the hibernation period, utilize the typical, rather than the maximum, application rate.

In addition, at the time of project-level NEPA review, local BLM offices would need to include additional conservation measures, specific to both the project and the site, for protecting the Northern Idaho ground squirrel.

Determination of Effects

Assuming that any vegetation treatments could occur anywhere on public lands, the proposed action would be **likely to adversely affect** northern Idaho ground squirrels or their habitats. However, implementation of conservation measures, as discussed in the previous section, would reduce the risk of any impacts that might occur as a result of vegetation treatments. With these measures in place, vegetation treatments would likely benefit northern Idaho ground squirrels and their habitat over the long term. Therefore, the effects determination could be reduced to **not likely to adversely affect** the northern Idaho ground squirrel.

Woodland Caribou

The primary reference for this section is:
USFWS. 1994n. Recovery Plan for Woodland Caribou in the Selkirk Mountains. Portland, Oregon.

References cited in this section are internal to the above-references document. Full citations have been included in the Bibliography.

The woodland caribou (*Rangifer tarandus caribou*) primarily occurs in Canada, but there is a small population—the Selkirk Mountain population—that extends into the northwestern United States. This population is generally found at elevations above 4,000 feet in the Selkirk Mountains, in Engelmann spruce/subalpine fire and western red cedar/western hemlock forest types. Prior to 1900, woodland caribou were distributed throughout northeastern, northcentral, and northwestern United States. However, since the 1960's, the last remaining population in the United States has restricted its range to the Selkirk Mountains of northeastern Washington and northern Idaho (in addition to southeastern British Columbia in Canada). The most recent aerial census count (March 2002) numbered this population at 34 animals, 9 of which were calves (The Lands Council 2003).

Woodland caribou, in general, do not make the long, mass migrations for which tundra caribou are famous. However, seasonal movements and migrations are characteristic of many, though not all, woodland caribou herds (Shoesmith and Storey 1978; Bloomfield 1980; Simpson et al. 1985; Antifeau 1987; Cichowski 1989; Servheen and Lyon 1989). Generally, the mountain ecotype of woodland caribou exhibit five distinct seasonal movements. In early winter, caribou shift to lower elevation habitats best characterized by mature to old-growth subalpine fir/Engelmann spruce and western hemlock/western red cedar forest types and the ecotone between them on moderate slopes with a high density of recently windthrown arboreal lichen-bearing trees. During early winter, these dense-canopied habitats intercept snow, reducing snow depth on the forest floor and providing green forage later in the season than more exposed forest communities at higher elevations.

The movement from early winter to late winter habitats (taking place anywhere between mid-December and mid-January) occurs as snow accumulates and hardens, allowing easier movement and lifting the caribou into the lichen-bearing canopy. The Engelmann spruce/subalpine fir forests used during this period are characterized by open canopies, and are generally above 6,000 feet in elevation (Servheen and Lyon 1989). Areas with moderate slopes on all aspects are most suitable for caribou during this period. Caribou are often located on ridge tops or open slopes with open, old-growth forests.

In spring, caribou move to areas of new growth, which are typically located at mid-elevation in open-canopied areas, often adjacent to mature forest (Scott and Servheen 1985, Servheen and Lyon 1989). These areas provide high quality forage in early spring, allowing caribou to recover from the effects of winter. Pregnant females move to typical spring habitat in April or May, then move back to snow-covered areas, often at higher elevations, to calve in early June. The areas selected for calving by the Selkirk Mountain caribou typically support old noncommercial forests with high lichen densities, open canopies, and small trees. Lichen again becomes the primary food source because green forage is unavailable at these elevations in early June (Servheen and Lyon 1989).

Caribou spend the summer in alpine and subalpine vegetative zones, primarily in areas of high forage availability. In early summer, open-canopied stands provide an abundance of forbs and huckleberry leaves (Scott and Servheen 1985), and as summer progresses the caribou move to more closed-canopy forest stands supporting forbs that mature later in the season (Servheen and Lyon 1989). In the fall, caribou shift to lower elevations and more densely canopied forest in the southern Selkirk Mountains.

Although caribou eat a wide range of foods, winter foraging is limited almost exclusively to arboreal lichens. Selkirk Mountain caribou may depend on arboreal lichens for up to 6 months of the year. During the remainder of the year, the caribou feed extensively on blackberry leaves, Sitka valerian, boxwood, and smooth woodrush.

Caribou generally have a low reproductive rate. Females usually give birth to their first calf when they are 3 years old, and single calves are the norm. The breeding season peaks in early to mid-October, and calves are born in May or June. Calf mortality during the first few months of life is high, often approaching 50% or greater. Common causes of calf mortality include inclement weather, predation, abandonment, and accidents. Selkirk Mountain caribou are polygamous, with adult males defending harems of six to 10 cows with calves. The breeding season is unusually short, and peaks during early or mid-October.

The woodland caribou was federally listed as endangered on February 29, 1984. Critical habitat has not been designated. Threats to caribou include habitat alteration caused by logging, mining, road construction, severe winter weather, and fire; predation by wolves; and low reproductive potential. In addition, overhunting and poaching, collisions with motor vehicles, and disease and genetic problems from inbreeding are also potential threats.

Effects of Vegetation Treatments on the Woodland Caribou

Effects Common to All Treatment Methods

Indirect Effects. Because of the potential effects to the limited remaining habitat in the Selkirk Mountains from fire, the Selkirk Mountain Woodland Caribou Recovery Plan (USFWS 1994n) calls for improving methods for fire protection and control. Therefore, any treatment method that reduces fuel loading in caribou habitat would be expected to have a long-term positive effect by reducing the likelihood, intensity, and area of influence of a future wildfire. In addition, treatments that reduce the cover of non-native species would also be expected to improve habitat quality, and to have a long-term positive effect.

Removal of forest vegetation through treatments, and creation of access routes into habitat could negatively affect caribou populations. Predation is thought to be the major source of mortality in caribou populations (Kinley and

Apps 2001). Given the already fragmented habitat of the species, any activity that increases the ability of predators to find caribou (e.g., access routes, increased visibility) would likely exacerbate this problem.

Prescribed Fire Treatments

Indirect Effects. As for most species, the effects of prescribed fire are highly dependent on numerous factors that are impossible to predict for this analysis. It is generally agreed that historically, wildfire was the primary disturbance factor in the Selkirk Mountains, where the listed caribou are located (USFWS 1994n). In the past, fire has destroyed caribou cover and winter food, and has altered habitat. Although such disturbances may not have a major impact when a large acreage of habitat is available, in the present conditions of limited, fragmented habitat, a fire could burn a large percentage of the remaining available habitat. Thus, prescribed burns in caribou habitat could have extensive adverse effects on caribou populations caused by an overall reduction in habitat.

Caribou are dependent on lichens for winter forage, and fire destroys lichens in the forest floor and on trees (Viereck and Schandelmeier 1980). After a burn, caribou may continue to avoid the area for 50 years or more, until lichens become established in the new forest (Smith 2000). However, some amount of fire may also be good for lichen production by eventually increasing lichen cover, or by rejuvenating older stands in which lichen quantities have begun to decrease. Thus, some amount of prescribed burning may have a long-term beneficial effect on caribou habitat. In addition, vegetation may green up earlier in burned areas than in other areas, and may be richer in nutrients for a few years (Viereck and Schandelmeier 1980). Fire may also result in an increase in the number of caribou predators, making caribou populations more subject to predator-dependent mortality.

Mechanical Treatment Methods

Indirect Effects. Although removal of fuels would likely benefit caribou habitat, mechanical treatments, depending on the method and amount of thinning and debris removal, could also have adverse effects on caribou habitat. Caribou require dense canopies in their early winter habitats to intercept snow (USFWS 1994). In addition, windthrown trees are a good source of lichen forage. Because vegetation also provides escape cover, it is very important that migration corridors be left intact.

Manual Treatment Methods

Indirect Effects. No major adverse effects to caribou or their habitat are expected from manual treatment methods.

Biological Control Treatments

Domestic Animals

<u>Indirect Effects</u>. As a species that forages on green vegetation in the summer and lichens in the winter, woodland caribou are dependent on the availability of food for survival. The introduction of domestic animals into areas utilized by woodland caribou could lead to competition for resources. Thus, this type of biological control would be expected to have a negative effect on caribou habitat.

Other Biological Control Treatments

<u>Direct and Indirect Effects</u>. No major adverse effects to caribou or their habitat are expected from the use of biological control agents. However, there is limited knowledge about the long-term effects of these agents, and it is possible that unanticipated impacts to the ecosystem (and therefore caribou or their habitat) could occur.

Herbicides

Direct Effects. Although caribou would readily flee areas in which herbicide applications were occurring, it is possible that some animals could be sprayed inadvertently. Based on the results of the ERAs, adverse effects to woodland caribou could potentially occur as a result of direct spray by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate (see Table 6-2). Furthermore, if caribou were to come into contact with foliage that had been sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate, adverse health effects could potentially occur.

Results of the ERAs predicted that if woodland caribou were to ingest plant materials treated with 2,4-D, bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, Overdrive®, picloram, or triclopyr at the typical application rate, or with imazapyr, metsulfuron methyl, or tebuthiuron at the maximum application rate, adverse health effects could potentially occur. Such a scenario would be possible if the BLM were to treat caribou habitat during the period in which caribou forage in the area, although it is unlikely that 100% of the animal's diet would come from contaminated forage, as assumed in ERAs.

Indirect Effects. Herbicide treatments in woodland caribou habitat could temporarily reduce the cover of available forage. Effects would be greatest if treatments occurred just before or during a period when forage was scarce. Over the long term, herbicide treatments could potentially improve the quality of forage in caribou habitat by reducing the cover of non-native species.

Conservation Measures

In order to minimize or avoid impacts to the woodland caribou, the BLM would be required to follow, at a minimum, the programmatic-level conservation measures listed below.

- At the local level, prepare a management plan for all proposed treatment activities that could potentially occur on land utilized by woodland caribou. This management plan must be completed with the assistance of a wildlife biologist and a forest ecologist, and must specifically address caribou and caribou habitat.
- Design prescribed burns and mechanical treatments so that no more than 10% of caribou habitat is affected at any one time.
- Time major herbicide treatments in woodland caribou habitats such that they do not occur during the season when caribou rely on the treatment area for forage.
- Do not use 2,4-D in woodland caribou habitats; do not broadcast spray 2,4-D within ¼ mile of woodland caribou habitat.
- Where feasible, avoid use of the following herbicides in woodland caribou habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, Overdrive®, picloram, and tebuthiuron, and triclopyr.
- Do not broadcast spray bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, Overdrive®, picloram, or triclopyr in woodland caribou habitat; do not broadcast spray these herbicides in areas adjacent to woodland caribou habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying imazapyr, metsulfuron methyl, or tebuthiuron in or near woodland caribou habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, imazapyr, metsulfuron methyl, tebuthiuron, or triclopyr to vegetation in woodland caribou habitat, utilize the typical, rather than the maximum, application rate.

Local offices would also be required to develop and implement any additional project- and site-specific conservation measures deemed necessary during the preparation of project-level NEPA documentation.

Determination of Effects

Assuming that any of the vegetation treatments could occur anywhere on public land, the proposed treatment program would be **likely to adversely affect** mountain caribou and/or their habitat. However, with the implementation of programmatic-level and project specific conservation measures, as discussed in the previous section, the proposed vegetation treatments would be **not likely to adversely affect** the woodland caribou.

Grizzly Bear

The primary reference for this section is:
USFWS. 1993n. Grizzly Bear Recovery Plan. Missoula, Montana.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The grizzly bear (*Ursus arctos horriblis*) was originally distributed in various habitats throughout western North America from Central Mexico to the Arctic Ocean. Its current distribution is reduced to less than 2% of its former range south of Canada in five, and perhaps six, small populations. There are four regions in the contiguous United States that accommodate grizzly populations: the Northern Continental Divide and Cabinet/Yaak in Montana, the Selkirks of Idaho and Washington, and the North Cascades of Washington. There is also a population in the Yellowstone ecosystem, and a possible sixth population in the Bitterroot ecosystem in Idaho. In Alaska, where the grizzly bear is more commonly called brown bears, populations are healthy (over 30,000 animals), and the species is classified as a game species. In the lower 48 states, it is estimated that there are a total of approximately 800 to 1,020 grizzly bears (Defenders of Wildlife 2003).

The grizzly bear has a broad range of habitat tolerance. Most areas in which the species remains are characterized by contiguous, relatively undisturbed mountainous habitat with a high level of topographic and vegetative diversity. Grizzly bears prefer areas of dense forest cover. In the winter, when there is deep snow, low ambient air temperatures, and an unavailability of food, bears hibernate in den sites. Excavation of dens starts as early as September, though it may occur just prior to entry in late November. Dens are usually dug on steep slopes where wind and topography cause an accumulation of deep snow, but where the snow is unlikely to melt during warm periods. Bears exhibit no overt defense of their dens, and several have been reported to abandon them because of human disturbance.

Seven essential characteristics of grizzly bear habitat have been defined: space, isolation, sanitation, denning, safety, vegetation types, and food (Craighead et al. 1982). Each of these characteristics contributes to the overall suitability of an area to provide habitat for grizzly bears. If one characteristic is absent from an area, or severely depleted, the ability of the entire ecosystem to sustain a grizzly bear population is much reduced.

Grizzly bears have an adaptive flexibility in food habits. Although the digestive system is essentially that of a carnivore, bears are successful omnivores, and in some areas may be almost entirely herbivorous. Bears feed on animal matter or vegetable matter that is highly digestible and high in starch, sugars, protein, and stored fat (Stebler 1972; Mealey 1975; Hamer et al. 1977). Grizzly bears must avail themselves of foods rich in protein or carbohydrates in excess of maintenance requirements in order to survive denning and post-denning periods. Herbaceous plants are eaten as they emerge from the soil, when crude protein levels are highest. Grizzly bears are opportunistic feeders and will prey or scavenge on almost any available food, including ground squirrels, ungulates, carrion, and garbage (Murie 1944, Hamer 1974). In areas where animal matter is less available, roots, bulbs, tubers, fungi, and tree cambium may be important in meeting protein requirements (Hamer 1974, Pearson 1975, Singer 1978).

The search for food has a prime influence on grizzly bear movements. Upon emergence from the den they seek the lower elevations, drainage bottoms, avalanche chutes, and ungulate winter ranges where their food requirements can be met. Throughout the late spring and early summer they follow plant phenology back to higher elevations. In late summer and fall, there is a transition to fruit and nut sources, as well as herbaceous materials.

Mating in grizzly bears appears to occur from late May through mid-July, with a peak in mid-June and estrus lasting from a few days to over a month (Craighead et al. 1969; Herrero and Hamer 1977). Females in estrus are receptive to practically all adult males (Hornocker 1962). Age of first reproduction and litter size varies, and may be related to nutritional state (Herrero 1978; Russell et al. 1978). Age at first reproduction varies from 3.5 to 8.5 years, with an average of 5.5 years. Litter size varies from one to four cubs, with an average of approximately two throughout much of the range of the species. Reproductive intervals for females average 3 years. The time lapse from conception to birth of cubs is between 229 and 266 days (Banfield 1974).

The grizzly bear was federally listed as threatened on July 28, 1975. Critical habitat has not been designated. The decline in numbers of this species is attributable to habitat loss and indirect human-caused mortality. Any bear-human interaction is a potential threat to either the bear or the human. The rate of grizzly bear mortality resulting from such interactions often exceeds birth rates (Craighead and Mitchell 1982). Factors that threaten the continued survival of this species include habitat alteration, loss, and fragmentation, hunting, and increased access by humans to wilderness. In addition, there has been some displacement of food sources by disease and invasive species.

Effects of Vegetation Treatments on the Grizzly Bear

Grizzly bears occurring in the project area could be negatively affected by vegetation treatments that increase mortality, or that degrade habitat elements used for denning and foraging.

Effects Common to All Treatment Methods

Indirect Effects. Any treatment activity that reduces fuels would be expected to have a positive effect on grizzly bear habitat by reducing the likelihood of a future catastrophic fire. Although grizzly bears generally benefit from periodic burns, a very large burn could destroy a large percentage of available habitat, and result in fragmentation of habitat. There is also some indication that invasive species have displaced some food plants utilized by grizzly bears. Therefore, any activity that reduces the cover of non-native species would be likely to have an indirect positive effect on habitat over the long term.

Grizzly bears typically occur in remote areas, away from human disturbance. Creation of access routes for vegetation treatments in remote areas would increase the likelihood of access to those areas in the future. The presence of human food in grizzly bear habitat attracts bears and results in the loss of natural fear and avoidance of humans (USFWS 1993n). Such habituation may result in “problem” bears that could eventually become a threat to humans and that often must be destroyed. Thus, any treatment activity involving the presence of humans in grizzly bear habitat, or the creation of access routes into habitat would be expected to have a negative effect on bears.

Prescribed Fire Treatments

Direct Effects. Although it is likely that some fire-related mortality of grizzly bears occurs, it is thought to be rare and unlikely to have a substantial impact on the grizzly bear population as a whole (Blanchard and Knight 1990). Denning sites could be burned by fire, possibly resulting in mortality during the hibernation period.

Indirect Effects. In general, fire is thought to have a positive effect on grizzly bear habitat, and fire suppression has been blamed for the decline of grizzly bear populations (Willard and Herman 1977, Tirmenstein 1983, Contreras and Evans 1986). Grizzly bears are opportunistic species with large home ranges, and their populations change little in response to fire (Smith 2000). They tend to thrive in areas where their preferred prey or forage is most plentiful—often in recent burns. Fires promote and maintain many important berry-producing shrubs and forbs, and provide a medium for insects, as well as carrion (primarily in the instance of very large fires). However, fire can also adversely affect other food sources, such as whitebark pine nuts.

Grizzly bears occupy a large area of suitable habitat, with requirements of abundant and concentrated food, the presence of denning areas, and wooded areas for hiding and thermal cover. Shrub and grass communities interspersed within the wooded areas provide a large portion of the food sources. Therefore, prescribed burning is unlikely to negatively affect grizzly bear habitat, unless enormous tracts of wooded areas are burned. Reduction of accumulated fuels would reduce the likelihood of such a catastrophic fire occurring in the future. Prescribed fire can create and maintain seral shrub communities by rejuvenating shrubs, releasing nutrients, and discouraging conifer dominance (Moss and Le Franc 1987, Zager 1980).

Mechanical Treatment Methods

Indirect Effects. Use of heavy equipment to control weeds and/or reduce fuels over a portion of grizzly bear habitat would be unlikely to have lasting negative effects. However, large-scale removal of vegetation could reduce the amount of forage food available to bears. The significance of this impact would depend on the percentage of the habitat area disturbed. The loud noises and human activities associated with mechanical control would be likely to temporarily disturb denning bears. In addition, heavy equipment could destroy some denning areas, such as those associated with downfall timber.

Manual Treatment Methods

Direct and Indirect Effects. Manual methods of vegetation treatment would be unlikely to affect grizzly bears or their habitat. Human activity associated with treatment activities could disturb denning bears, but these effects would be minor and temporary.

Biological Control Treatments

Domestic Animals
Indirect Effects. Moderate levels of grazing are unlikely to affect grizzly bears or their habitat. Given that grizzly bears utilize a wide range of habitat types and food resources, the use of domestic animals to contain undesirable species would be unlikely to substantially affect bear habitat or food resources. Intense levels of grazing, however, could have an effect on the diversity of plant resources available to grizzly bears for consumption.

Other Biological Control Agents
Direct and Indirect Effects. The use of biological control agents in or near grizzly bear habitats is unlikely to have negative effects on bears. There would be some human activity associated with the release of the agents, which could disturb denning bears, but it would be minor and temporary. Since there is limited knowledge about the long-term effects of these agents, there is a chance that their release could result in unanticipated impacts to the ecosystem, which could affect grizzly bears or their habitats. However, these effects are not reasonably foreseeable.

Herbicides

Direct Effects. During treatments, human activity and use of vehicles could disturb any denning bears nearby. The herbicides themselves are unlikely to directly affect grizzly bears. A scenario in which a grizzly bear would be sprayed inadvertently during herbicide application would be unlikely, since grizzly bears would avoid these sites during treatments, and such a large animal is not likely to be overlooked by operators of herbicide application equipment. If an inadvertent spray of one or more grizzly bears by herbicides did occur, chemicals with the potential to cause adverse health effects (based on the ERA results for small mammals) would be 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or imazapyr or metsulfuron methyl at the maximum application rate (see Table 6-2). After a treatment, dermal contact with foliage sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate, could also result in adverse health effects to grizzly bears.

Since grizzly bears are omnivores, exposure to both contaminated animals and plants via ingestion would be possible, should a bear enter a recently-treated area. According to the ERAs, ingestion of plant materials sprayed with 2,4-D, bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, Overdrive®, picloram, or triclopyr at the typical application rate, or with imazapyr, metsulfuron methyl, or tebuthiuron at the maximum application rate, could result in adverse health effects to grizzly bears (see Table 6-5). It is also assumed that adverse health effects could potentially occur as a result of ingesting prey items sprayed by 2,4-D or diuron at the typical application rate, or by bromacil, diquat, or triclopyr at the maximum application rate. However, these risk scenarios assume that 100% of the animal's diet would consist of either contaminated prey or contaminated vegetation, which would be unlikely to occur. Exposure of carnivorous mammals to herbicides through consumption of contaminated prey was not assessed in the ERA for hexazinone. Therefore, the potential for adverse health effects to grizzly bears as a result of exposure to hexazinone via this exposure pathway cannot be determined.

Indirect Effects. Herbicide treatments would be unlikely to have a substantial effect on grizzly bear habitat, unless a key food source (such as a berry patch) was eliminated.

Conservation Measures

Potential effects to grizzly bears from vegetation treatments could be avoided or minimized by following a number of conservation measures. To minimize the potential for displacement/mortality risk during treatments:

- Within the Recovery Zone (defined in Grizzly Bear Recovery Plan, USFWS 1993), ensure that any vehicular travel off highway or on restricted roads adheres to access standards/directions as provided in local or regional interagency agreements, biological opinions, or local land use plans.
- Limit all activities requiring overnight stays or establishment of a base camp to less than 20 individuals and less than 5 days within the Grizzly Bear Recovery Zone.
- Limit firewood collection within the Recovery Zone to roadside hazard tree removal, road maintenance, or campground maintenance activities.
- Within the Recovery Zone, do not conduct vegetation treatment activities in riparian meadows and stream corridors between April 1 and July 1, or complete these activities in 1 day.
- Within the Recovery Zone, do not implement vegetative treatments that would substantially change the vegetative community in huckleberry producing sites.

To minimize the potential for habituation/human conflict:
- Within the Recovery Zone, ensure that all treatment activities adhere to interagency grizzly bear guidelines or local interagency grizzly bear standards for sanitation measures and storage of potential attractants.
- Within the Recovery Zone, do not plant or seed highly palatable forage species near roads or facilities used by humans.

To minimize the likelihood that grizzly bears would suffer adverse health effects as a result of exposure to herbicides:
- Do not use 2,4-D in the Recovery Zone; do not broadcast spray 2,4-D within ¼ mile of the Recovery Zone
- Where feasible, avoid use of the following herbicides in the Recovery Zone: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, Overdrive®, picloram, tebuthiuron, and triclopyr.
- Do not broadcast spray bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, Overdrive®, picloram, or triclopyr in the Recovery Zone; do not broadcast spray these herbicides in areas adjacent to the Recovery Zone under conditions when spray drift into the Recovery Zone is likely.
- If broadcast spraying imazapyr, metsulfuron methyl, or tebuthiuron in or near the Recovery Zone, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, imazapyr, metsulfuron methyl, tebuthiuron, or triclopyr to vegetation in the Recovery Zone, utilize the typical, rather than the maximum, application rate.

In addition, analysis of potential site-specific impacts to grizzly bears would occur at the project level, and any additional conservation measures deemed necessary would also need to be applied to ensure that potential effects were minimized or avoided.

Determination of Effects

Assuming that vegetation treatments could occur anywhere on public lands, including remote areas occupied by grizzly bears, the proposed action is **likely to adversely affect** grizzly bears. However adverse effects could be avoided by implementing both programmatic- and project-level conservation measures, as described in the preceding section, thereby resulting in a **not likely to adversely affect** determination.

Canada Lynx

The primary reference for this section is:
USFWS. 2000m. Determination of Threatened Status for the Contiguous U.S. Distinct Population Segment of the Canada Lynx and Related Rule. Federal Register 65(58): 16051-16086.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Montana Field Office, Helena, Montana.

Lynx occur in moist coniferous forests that provide a prey base of snowshoe hare (Quinn and Parker 1987; Koehler and Brittell 1990; Koehler 1990; Mowat et al. 1999). In the contiguous United States, the Canada lynx (*Lynx canadensis*) historically occurred in the Cascades Range of Washington and Oregon; the Rocky Mountain Range in Montana, Wyoming, Idaho, eastern Washington, eastern Oregon, northern Utah, and Colorado; the western Great Lakes Region; and the northeastern United States region from Maine southwest to New York (McCord and Cardoza 1982, Quinn and Parker 1987). This distribution associated with the southern boreal forest, comprising of subalpine coniferous forest in the West and primarily mixed coniferous/deciduous forest in the East (Aubry et al. 1999). In Canada and Alaska, however, lynx inhabit the classic boreal forest ecosystem known as the taiga (McCord and Cardoza 1982; Quinn and Parker 1987; Agee 1999; McKelvey et al. 1999b). Within these general forest types, lynx are most likely to persist in areas that receive deep snow, for which the lynx is highly adapted (Ruggiero et al. 1999b).

The lynx population in the contiguous U.S. is considered by the USFWS to be part of a larger metapopulation whose core is located in the northern boreal forest of central Canada (Buskirk et al. 1999b; McKelvey et al. 1999a, 1999b). The boreal forest extends south into the contiguous United States along the Cascade and Rocky Mountain ranges in the West, the western Great Lakes Region, and along the Appalachian Mountain Range of the northeastern United States. At its southern margins, the boreal forest becomes naturally fragmented into patches of varying size as it transitions into other vegetation types. These southern boreal forest habitat patches are small relative to the extensive northern boreal forest of Canada and Alaska, which constitutes the majority of the lynx range. Many of these southern boreal forest habitat patches within the contiguous U.S. are able to support resident populations of lynx and snowshoe hare. It is likely that some of the habitat patches act as sources of lynx (recruitment is greater than mortality) that are able to disperse and potentially colonize other patches (McKelvey et al. 1999a). Other habitat patches act as “sinks” where lynx mortality is greater than recruitment and lynx are lost from the overall population. The ability of naturally dynamic habitat to support lynx populations may change as the habitat undergoes natural succession following natural or manmade disturbances (i.e., fire, clearcutting). In addition, fluctuations in the prey populations may cause some habitat patches to change from being sinks to sources and vice versa.

It is believed that historic and current lynx densities in the contiguous U.S. are naturally low relative to lynx densities in the northern boreal forest. At present, in the western states, resident populations currently exist only in Montana and Washington, and populations that are no longer self-sustaining occur in Oregon, Idaho, Wyoming, Utah, and Colorado. Because the lynx is a secretive animal, there are no reliable population estimates for this species. However, sightings of lynx throughout the U.S. have continued to decrease over the years.

Lynx are highly specialized predators whose primary prey is the snowshoe hare, a species that has evolved to survive in areas that receive deep snow (Bittner and Rongstad 1982). Snowshoe hares use forests with dense understories that provide forage, cover to escape from predators, and protection during extreme weather (Wolfe et al. 1982; Monthey 1986; Hodges 1999a, 1999b). Generally, earlier successional forest stages have greater understory structure than do mature forests and therefore support higher hare densities (Hodges 1999a, 1999b). However, mature forests can also provide snowshoe hare habitat as openings develop in the canopy of mature forests when trees succumb to disease, fire, wind, ice, or insects, and the understory grows (Buskirk et al. 1999b). Lynx concentrate their hunting activities in areas where hare activity is relatively high (Koehler et al. 1979; Parker 1981; Ward and Krebs 1985; Major 1989; Murray et al. 1994; O’Donoghue et al. 1997, 1998a). Lynx also prey opportunistically on other small mammals and birds, particularly when hare populations decline (Nellis et al. 1972;

O'Donoghue 1997, 1998a). Red squirrels are an important alternate prey (Apps 1999; Aubry et al. 1999). However, a shift to alternate food sources may not compensate for the decrease in hares consumed (Koehler and Aubry 1994). In northern regions, when hare densities decline, the lower quality diet causes sudden decreases in the productivity of adult female lynx and decreased survival of kittens, which causes the numbers of breeding lynx to level off or decrease (Nellis et al. 1972; Brand et al. 1976; Slough and Mowat 1996; O'Donoghue et al. 1997).

The breeding period for Canada lynx is late winter to early spring, with adult females producing one litter every 1 to 2 years. The gestation period typically lasts from 62 to 74 days, and the litter size is 3 to 4 kittens, on average. Females may reach reproductive maturity by as early as 1 year (Brainerd 1985).

Lynx use large woody debris, such as downed logs and windfalls, to provide denning sites with security and thermal cover for kittens (McCord and Cardoza 1982, Koehler 1990, Koehler and Brittell 1990, Squires and Laurion 1999, Organ 1999). For lynx den sites, the age of the forest stand does not seem as important as the amount of downed, woody debris available (Mowat et al. 1999). The size of lynx home ranges varies by the animal's gender, abundance of prey, season, and the density of lynx populations (Hatler 1988; Koehler 1990; Poole 1994; Slough and Mowat 1996; Aubry et al. 1999; Mowat et al. 1999). Documented home ranges vary from 3 to 300 square miles (Saunders 1963; Brand et al. 1976; Mech 1980; Parker et al. 1983; Koehler and Aubry 1994; Apps 1999; Mowat et al. 1999; Squires and Laurion 1999).

The population of the Canada lynx occurring in the contiguous U.S. was federally listed as threatened on March 24, 2000. The designation of critical habitat for the species was deemed prudent, but has not yet occurred. According to the USFWS, the primary factor affecting lynx in the contiguous U.S. is the lack of guidance for conservation of lynx in federal land management plans. People change forests through timber harvest, fire suppression and conversion of forest lands to agriculture. Forest fragmentation may eventually become severe enough to isolate habitat into small patches, thereby reducing the viability of lynx populations, which are dependent on larger areas of forest habitat (Litvaitis and Harrison 1989). In addition, human alteration of forests may facilitate competition by creating habitats that are more suitable to potential lynx competitors (McCord and Cardoza 1982, Quinn and Parker 1987, Buskirk et al. 1999a). Finally, lynx movements may be negatively influenced by high traffic volume on roads that bisect suitable lynx habitat, such as in the Southern Rockies and in some parts of the Northern Rockies/ Cascades Region.

Effects of Vegetation Treatments on the Canada Lynx

Lynx occurring in the project area may be affected by management activities that reduce or degrade essential habitat elements used by lynx for denning, foraging, and recruitment, or that increase habitat fragmentation and lynx mortality.

Effects Common to All Treatment Methods

Indirect Effects. The invasion of non-native species into lynx habitat can be a risk factor to the species if it occurs at a large scale. The associated habitat degradation and the potential changes in understory vegetation can both have indirect effects on the Canada lynx by changing the structure of stands or reducing the availability of food for prey sources. Therefore, any vegetation treatment that reduces the cover of non-natives or thwarts their establishment would have a long-term positive effect on lynx habitat. In addition, fuels reduction activities that reduce the likelihood of a future catastrophic fire would also have long-term positive effects on lynx and their habitat.

The use of vegetation treatments to reduce hazardous fuels would provide a long-term benefit to lynx habitat by minimizing the potential for a large, catastrophic fire, and by maintaining and improving the diversity of habitats for lynx and lynx prey species (USDI BLM 2002b). Fire exclusion in lynx habitats has, over time, altered forest stand composition and structure, making forests more susceptible to severe fires (Quigley et al. 1996). Use of vegetation treatments to return forests to more natural conditions would be expected to benefit lynx over the long term.

Since snowshoe hares are the primary prey item for lynx, their abundance may affect the success of lynx populations. Vegetation treatments can create openings in the forest that favor snowshoe hares and other lynx prey species.

The use of vegetation treatments in lynx habitat could negatively affect the species by creating new access routes for humans and competitors, and potentially fragmenting habitat. Construction of roads has been observed to increase the likelihood of human-lynx interactions, the vulnerability of lynx to legal and illegal harvest, and the amount of lynx harassment (Washington Department of Wildlife 1993).

Prescribed Fire Treatments

Direct Effects. As large, mobile animals, lynx should be able to avoid direct contact with fire, and are unlikely to be injured or killed by a prescribed burn, with the possible exception of newborn kittens.

Indirect Effects. Lynx have been observed hunting along the edges of mature stands within a burned forest matrix. Fire is a natural component of the conifer forests that lynx typically inhabit, and the species may benefit from certain aspects of prescribed burns. In the short term, a severe fire would likely eliminate snowshoe hares from a site with the destruction of brush habitat. Therefore, lynx numbers would also be expected to drop. For the first few years after a burn, there appears to be a negative correlation between lynx use and the amount of area burned (Fox 1978). The reduction in snowshoe hares, the removal of cover, and the possible increase in competition from coyotes resulting from the burn would all negatively affect lynx populations (Stephenson 1984, Koehler and Brittell 1990). However, hare populations could increase dramatically with the return of shrubs to the area (Bradley et al. 1992). After a fire, it generally takes about 15 to 30 years for the hare populations to increase to peak levels, depending on the habitat type and the severity of the fire (Ruediger et al. 2000). This increase in lynx prey would have a positive effect on lynx for several years.

Areas that sustain burns that are less severe than stand-replacing fires may still be used by snowshoe hares. These low to moderate intensity fires can stimulate understory growth in older stands, having an overall positive effect on lynx habitat. Because lynx are dependent on the early successional habitat that hares prefer, suppression of wildfire has been identified as a factor that negatively affects lynx habitat by limiting the availability of foraging habitat. In addition, fire typically results in a variety of tree species and age classes, which provide lynx with open habitats for prey, as well as unburned mature stands for denning females (Washington Department of Wildlife 1993).

Prescribed fire could negatively affect lynx habitat by destroying structural components of the forest upon which lynx depend. Lynx use large woody debris for denning sites; the removal of this material through prescribed fire could affect the survival of lynx kittens. Lynx also avoid large openings—they typically do not cross openings wider than 300 feet (Koehler and Brittell 1990)—so a fire covering an extremely large area of lynx habitat could have a negative effect on populations if there was a lack of suitable habitat available nearby (for emigration).

Mechanical Treatment Methods

Indirect Effects. Thinning of trees and other fuels can alter lynx habitat by reducing cover and converting mature forests to early successional stages. Snowshoe hare and other prey would likely benefit from these early successional stages. However, large-scale removal of trees could have a negative effect on lynx by creating large openings and reducing the availability of cover for denning. Since lynx use large woody debris for denning sites, the removal of this material could affect the survival of lynx kittens. Thinning in forests could also reduce the dense horizontal cover that is necessary for maintaining an adequate snowshoe hare prey base.

Manual Treatment Methods

Direct and Indirect Effects. The limited disturbance cause by manually removing plants and other materials from lynx habitat is not likely to cause adverse effects to the species.

Biological Treatment Methods

Domestic Animals
Indirect Effects. Grazing has been identified as a factor that potentially affects lynx productivity. Because snowshoe hares depend on understory plants for forage, it is possible that domestic animals brought in to control weeds would compete with snowshoe hares for forage resources, thereby indirectly affecting lynx by potentially reducing the availability of prey. In addition, grazing in openings recently created by fire or timber harvest could delay the regeneration of the shrub understory in these areas, also indirectly affecting lynx. Finally, grazing in key lynx corridors could reduce the amount of cover connecting patches of lynx habitat within a home range.

Other Biological Control Agents
Direct and Indirect Effects. Biological control methods that target a particular undesirable species are unlikely to adversely affect lynx or their habitat. However, given the lack of knowledge about long-term effects of biological control agents, unanticipated effects to the ecosystem are always possible. However, these effects are not reasonably foreseeable.

Herbicides

Direct Effects. Herbicide treatments would be unlikely to directly affect Canada lynx that occur on public lands, since lynx would readily avoid areas in which herbicide treatments were occurring. Unintentional spray of lynx would be highly unlikely. Nonetheless, if such an exposure were to occur, animals that were directly exposed could suffer adverse health effects as a result of direct spray by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate (see Table 6-2). Furthermore, dermal contact with foliage sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate, could potentially result in adverse health effects to lynx.

Based on the results of the ERAs, it is possible that a Canada lynx would suffer adverse health effects if it consumed a prey item sprayed by 2,4-D or diuron at the typical application rate, or by bromacil, diquat, or triclopyr at the maximum application rate (see Table 6-5). Exposure of carnivorous mammals to herbicides through ingestion of contaminated prey was not assessed in the ERA for hexazinone. Therefore, the potential for adverse effects from exposure to hexazinone via this exposure pathway cannot be determined.

Indirect Effects. Herbicide treatments could potentially affect snowshoe hares via direct spray or ingestion exposure pathways. However, it is unlikely that populations of snowshoe hares or other prey items would change substantially, since these species would also be able to flee or hide from treatments.

Conservation Measures

In order to minimize or avoid impacts to lynx, the BLM must follow, at a minimum, the conservation measures listed below:

- Prior to vegetation treatments, map lynx habitat within areas in which treatments are proposed to occur. Identify potential denning and foraging habitat, and topographic features that may be important for lynx movement (major ridge systems, prominent saddles, and riparian corridors).
- Design vegetation treatments in lynx habitat to approximate historical landscape patterns and disturbance processes.
- Avoid the construction of permanent firebreaks on ridges or saddles in lynx habitat.
- Where possible, keep linear openings out of mapped potential habitat and away from key habitat components, such as denning areas.
- When planning vegetation treatments, minimize the creation of linear openings (fire lines, access routes, and escape routes) that could result in permanent travel ways for competitors and humans.

- Obliterate any linear openings constructed within lynx habitat in order to deter future uses by humans and competitive species.
- Design burn prescriptions to regenerate or create snowshoe hare habitat (e.g., regeneration of aspen and lodgepole pine).
- Ensure that no more than 30% of lynx habitat within a Lynx Analysis Unit (as defined in Ruediger et al. 2000) would be in an unsuitable condition at any time.
- If deemed necessary, defer livestock grazing following vegetation treatments to ensure the re-establishment of key plant species. Bureau of Land Management personnel should use resource goals and objectives to determine the need for this restriction and the length of deferment on a case by case basis.
- Give particular consideration to amounts of denning habitat, condition of summer foraging and winter foraging habitat, as well as habitat linkages, to ensure that that treatments do not negatively impact lynx.
- Do not use 2,4-D in Canada lynx habitat; do not broadcast spray 2,4-D within ¼ mile of Canada lynx habitat.
- Where feasible, avoid use of the following herbicides in Canada lynx habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr.
- Do not broadcast spray clopyralid, diuron, glyphosate, hexazinone, picloram, or triclopyr in Canada lynx habitat; do not broadcast spray these herbicides in areas adjacent to Canada lynx habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, diquat, imazapyr, or metsulfuron methyl in or near Canada lynx habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in Canada lynx habitat, utilize the typical, rather than the maximum, application rate.

In addition, the BLM must develop and implement additional conservation measures, as necessary, during project-level analysis at the local level.

Determination of Effects

Assuming that any vegetation treatments could occur anywhere on public lands, including lynx habitat, the proposed treatment program would be **likely to adversely affect** lynx and/or their habitat. However, impacts could be minimized or avoided through the implementation of programmatic- and project-level conservation measures, as described in the previous section. By following this guidance, the BLM would be able to reduce the effects determination to **not likely to adversely affect** the Canada lynx or its habitat.

San Joaquin Kit Fox

The primary reference for this section is:
USFWS. 1998h. Recovery Plan for Upland Species of the San Joaquin Valley, California. Region 1. Portland, Oregon.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The San Joaquin kit fox (*Vulpes macrotis mutica*) is endemic to California's San Joaquin Valley and surrounding foothills. Prior to 1930, kit foxes inhabited most of the San Joaquin Valley from southern Kern County north to Tracy, San Joaquin County, on the west side, and near LaGrange, Stanislaus County, on the east side. The habitat of this species has been much reduced as a result of urban development and cultivation for agriculture. The largest remaining extant populations of kit foxes are in western Kern County on and around the Elk Hills and Buena Vista Valley, and in the Carrizo Plain Natural Area, San Luis Obispo County.

Historically, San Joaquin kit foxes occurred in several native plant communities of the San Joaquin Valley, some of which are only represented by small, degraded remnants today. Other habitats in which kit foxes are found have been extensively modified by humans. These habitats include grasslands and scrublands with active oil fields, wind turbines, and an agricultural matrix of row crops, irrigated pasture, orchards, vineyards, and grazed annual grasslands. Other plant communities in the San Joaquin Valley providing habitat for the species include are vernal pools and alkali meadows and playas. In the southernmost portion of its range, the kit fox is associated with Valley sink scrub, Valley saltbush scrub, Upper Sonoran subshrub scrub, and annual grassland. In the central portion of its range, the species is associated with Valley sink scrub, Interior Coast Range saltbush scrub, Upper Sonoran subshrub scrub, annual grassland, and the remaining native grasslands. In the northern portion of its range, the species is associated with annual grassland (Hall 1983) and Valley oak woodland (Bell 1994).

Kit foxes prefer loose-textured soils (Grinnell et al. 1937; Hall 1946; Egoscue 1962; Morrell 1972), but are found on virtually every soil type. Dens appear to be scarce in areas with shallow soils because of the proximity to bedrock (O'Farrell and Gilbertson 1979; O'Farrell et al. 1980), high water tables (McCue et al. 1981), or impenetrable hardpan layers (Morrell 1972). However, kit foxes will occupy soils with high clay content, where they modify burrows dug by other animals (Orloff et al. 1986).

The diet of kit foxes varies geographically, seasonally, and annually, based on variation in abundance of potential prey. In the southern portion of their range, kangaroo rats, pocket mice, white-footed mice, and other nocturnal rodents comprise about one-third or more of their diets. Kit foxes there also prey on California ground squirrels, black-tailed hares, San Joaquin antelope squirrels, desert cottontails, ground-nesting birds, and insects (Scrivner et al. 1987). Vegetation and insects are also eaten, with grass being the most commonly ingested plant material (Morrell 1971). In the central portion of their range, known prey species include white-footed mice, insects, California ground squirrels, black-tailed hares, and chukar (Jensen 1972, Archon 1992). In the northern part of their range, kit foxes consume California ground squirrels most frequently (Orloff et al. 1986), and also prey upon black-tailed hares, pocket mice, and kangaroo rats (Hall 1983).

Kit foxes can breed when 1 year old, but may not breed during their first year of adulthood (Morrell 1972). Adult pairs remain together all year, sharing the home range, but not necessarily the same den. During September and October, adult females begin to clean and enlarge natal or pupping dens. Mating and conception take place between late December and March (Egoscue 1956; Morrell 1972; Zoellick et al. 1987). The median gestation period is estimated to range from 48 to 52 days, and litters of between two and six pups are born sometime between February and late March. The pups emerge above ground at slightly more than 1 month of age. After 4 to 5 months, usually in August or September, the family bonds begin to dissolve and the young begin dispersing.

The San Joaquin kit fox was federally listed as endangered on March 11, 1967. Critical habitat has not been designated. The primary factors associated with the decline of this species were loss, degradation, and fragmentation of habitats as a result of agricultural, industrial, and urban developments in the San Joaquin Valley (Laughrin 1970, Morrell 1971, Jensen 1972, Knapp 1978). The primary threats to this species continue to be loss and degradation of habitat, which decrease the carrying capacity of the remaining habitat. Livestock grazing may affect habitat by altering the number of prey species, and destroying shrub cover although some amount of grazing may benefit kit foxes in some areas (Laughrin 1970, Balestreri 1981). The use of pesticides and rodenticides also poses a threat to kit foxes, either directly, secondarily, or indirectly by reducing prey. At present, the status of the kit fox throughout much of its current range is poorly known. It is estimated that fewer than 7,000 kit foxes remain.

Effects of Vegetation Treatments on the San Joaquin Kit Fox

Effects Common to All Treatment Methods

Indirect Effects. Fuels reduction treatments would potentially have a positive effect on kit fox habitat. Given the current fragmented nature of the species' habitat, an uncontrolled wildfire burning through a large tract of remaining suitable habitat could have a severe effect on fox populations. Such a fire could also increase the amount of habitat fragmentation. Thus, any treatment method that reduced the amount of fuels would be expected to have a long-term positive effect on kit foxes. Activities that reduce the cover of non-native species can also have a beneficial effect by helping to restore the native conditions that historically supported the fox. However, given the ability of the fox to adapt to altered landscapes, these benefits may be minimal.

Prescribed Fire Treatments

Direct Effects. A prescribed fire could cause some direct mortality to kit foxes, depending on its location and intensity. Many foxes, however, would be able to escape the burn by fleeing into underground dens and other sheltered places.

Indirect Effects. A prescribed fire would be expected to injure, kill, or reduce the suitability of habitat for small mammals and ground nesting birds, which are among the most important prey items of kit foxes. Immediately after the fire, foxes could be forced into other habitats to feed, which could temporarily reduce the success of populations. Shortly after a prescribed burn, however, the number of small mammals in the area typically increases, provided food and shelter are available (Smith 2000). This increase would be expected to benefit kit foxes.

Mechanical Treatment Methods

Direct Effects. Direct effects to kit foxes from mechanical treatment methods are unlikely, since foxes can escape equipment by running into underground burrows. It is possible that heavy equipment could cause some amount of damage to burrows.

Indirect Effects. Removal of shrub cover and tall vegetation would make the habitat more suitable for prey species, but would also reduce the hiding places for foxes, making it harder for them to hunt prey. Wide-scale removal of vegetation could also have negative effects on kit foxes by reducing the availability of food for prey species, which typically eat plant materials. However, these effects would be short-term in nature, and the vegetation removal could actually stimulate the growth of more desirable species.

Manual Treatment Methods

Direct and Indirect Effects. Hand pulling and other manual treatment methods would be unlikely to substantially affect kit foxes or their habitat. Disturbances and habitat alterations associated with these activities would be minimal.

Biological Control Treatments

Domestic Animals

Foxes occur in grazed grasslands, among other habitats (USFWS 1998h), indicating that some level of grazing is unlikely to be detrimental to kit foxes (Morrell 1975; Orloff et al. 1986). Presumably, the effects of weed containment by domestic animals on fox habitat are dependent on the intensity of the treatment, as well as its timing and duration. Some amount of grazing can be beneficial to prey species in some areas (see kangaroo rats section following this kit fox analysis), and would therefore benefit kit foxes in turn (Laughrin 1970, Balestreri 1981). However, higher levels of grazing may reduce the number of certain prey species by reducing the amount of forage available to these species. In addition, more intense grazing can destroy shrub cover, reducing the predatory advantage of kit foxes.

Other Biological Control Agents

Direct and Indirect Effects. The use of biological control agents to control undesirable vegetation would be unlikely to substantially affect kit foxes or their habitat. These agents target specific, undesirable plant species, and have a gradual effect on vegetation. However, given the limited knowledge about the long-term effects of biological control, it is possible that unanticipated impacts to the ecosystem (and therefore kit foxes and/or their habitat) could occur. These impacts are not reasonably foreseeable, however.

Herbicides

Direct Effects. It is unlikely that kit foxes would be inadvertently sprayed by herbicides during chemical treatments. Foxes would readily flee the treatment area or run into underground burrows. Nonetheless, inadvertent direct spray by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, could conceivably result in adverse health effects to kit foxes (see Table 6-2). In addition, dermal contact with foliage sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate could potentially result in adverse health effects to foxes.

Kit foxes would also be exposed to herbicides by consuming prey items that were directly exposed to herbicides. Since kit foxes also are known to ingest vegetation such as grasses occasionally, it is possible that indirect exposure to herbicides via plant material could also occur. According to the ERAs, ingestion of prey items recently exposed to 2,4-D or diuron at the typical application rate, or to bromacil, diquat, or triclopyr at the maximum application rate, could result in adverse health effects to San Joaquin kit foxes (see Table 6-5). Since the ERA for hexazinone did not assess the potential risks to carnivorous species through ingestion of contaminated prey, the potential for adverse effects to San Joaquin kit foxes from exposure to hexazinone via this exposure pathway cannot be determined. Finally, consumption of plant materials exposed to 2,4-D, diquat, or diuron at the typical application rate, or to bromacil, fluridone, glyphosate, hexazinone, or tebuthiuron at the maximum application rate, could result in adverse health effects. However, these effects were predicted based on the assumption that 100% of the diet would consist of contaminated food items, which is not a reasonably foreseeable scenario.

Indirect Effects. Herbicide treatments would have minimal effects on kit fox habitat over the short term. A temporary reduction in vegetative cover could benefit foxes by increasing their ability to locate prey items. Over the long term, use of herbicides to return kit fox habitats to more native conditions would likely benefit the species.

Conservation Measures

In order to minimize or avoid impacts to San Joaquin kit foxes from herbicide treatments, the BLM must follow, at a minimum, the programmatic conservation measures listed below:

- Do not use 2,4-D in San Joaquin kit fox habitat; do not broadcast spray 2,4-D within ¼ mile of San Joaquin fox habitat.
- Where feasible, avoid use of the following herbicides in San Joaquin kit fox habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, tebuthiuron, and triclopyr.
- Do not broadcast spray clopyralid, diuron, glyphosate, hexazinone, picloram, or triclopyr in San Joaquin kit fox habitat; do not broadcast spray these herbicides in areas adjacent to San Joaquin kit fox habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, diquat, imazapyr, metsulfuron methyl, or tebuthiuron in or near northern San Joaquin kit fox habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of diuron, glyphosate, hexazinone, tebuthiuron, or triclopyr to vegetation in San Joaquin kit fox habitat, utilize the typical, rather than the maximum, application rate.

In addition, the BLM must develop and implement additional conservation measures, as necessary, during project-level analysis at the local level.

Summary of Effects

The long-term effects of the proposed action would likely be beneficial to kit foxes. However, assuming that herbicide applications with all chemicals could occur anywhere on public lands, including areas that support kit foxes, the proposed action would be **likely to adversely affect** San Joaquin kit foxes, based on the ERAs. Potential impacts to the species could be minimized or avoided through the implementation of programmatic- and project-level conservation measures, as described in the previous section. By following this guidance, the BLM would be able to reduce the effects determination to **not likely to adversely affect** the San Joaquin kit fox or its habitat.

Grassland Ground-Burrowing Mammals: Kangaroo Rats, Utah Prairie Dog, and Black-Footed Ferret

Kangaroo Rats
The primary reference for the next three sections is:
USFWS. 1998h. Recovery Plan for Upland Species of the San Joaquin Valley, California. Region 1. Portland, Oregon.

References cited in these sections are internal to the above-referenced document. Full citations have been included in the Bibliography.

Giant Kangaroo Rat

The giant kangaroo rat (*Dipodomys ingens*) is found in grassland and shrubland communities in the southern San Joaquin Valley of California. Below about 1,300 feet, the species occurs in annual grassland and saltbush scrub. At higher elevations, it is found in Upper Sonoran subshrub scrub associations. Giant kangaroo rats are most numerous where annual grasses and forbs predominate. The species population is currently fragmented into six major geographic units. These major units are in turn fragmented into more than 100 smaller populations, many of which are isolated by steep terrain with plant communities unsuitable as habitat, or by agricultural, industrial, or urban land that provides poor habitat for this species.

Giant kangaroo rats are primarily seed eaters, but also eat green plants and insects. They forage on the surface from around sunset to near sunrise, though most activity takes place in the first 2 hours after dark. Foraging activity is greatest in the spring as seeds of annual plants ripen. The ability to transport large quantities of seeds and other foods in their cheek pouches, and highly developed caching behaviors, coupled with relatively high longevity of adults with established burrow systems, probably allow giant kangaroo rats to endure severe drought for 1 or 2 years without great risk of population extinction (Williams et al. 1993b).

Giant kangaroo rats have an adaptable reproductive pattern that is affected by both population density and availability of food. During times of relatively high density, females have a short, winter reproductive season with only one litter produced, and there is no breeding by young-of-the-year. However, if there is sufficient food and space, females can breed the year of their birth, and some may have two to three litters per year. In most years, females are reproductive between December and March or April, but in colonies with low densities, reproduction can extend into August or September (Williams et al. 1993b). Young disperse at about 11 to 12 weeks after birth. However, in years of high density, when most or all burrow systems are occupied, most young appear to remain in their natal burrows until the opportunity to disperse arises or they are finally driven off by the mother or one of the siblings.

The giant kangaroo rat was federally listed as endangered on January 5, 1987. Critical habitat has not been designated. Since the time of listing, conversion of habitat for agricultural purposes has slowed substantially, as most tillable land has already been cultivated. However, urban and industrial developments, petroleum and mineral exploration and extraction, and other activities continue to destroy habitat and increase threats to the species by reducing and further fragmenting populations. In addition, populations are small and vulnerable to extinction from demographic and random catastrophic events (drought, flooding, fire), and inappropriate land uses that can degrade or destroy habitat.

Fresno Kangaroo Rat

The Fresno kangaroo rat (*D. nitratoides exilis*), a subspecies of the San Joaquin kangaroo rat, occupies sands and saline sandy soils in chenopod scrub and annual grassland communities on the San Joaquin Valley floor. The current distribution of this subspecies is unknown. Recently, they have been found only in alkali sink communities located between 200 and 300 feet in elevation. Like other species of kangaroo rat, Fresno kangaroo rats collect and carry seeds in fur-lined cheek pouches. Seeds are a staple in their diet, but they also eat some types of green, herbaceous vegetation, and insects. Seeds are gathered when they are available and then cached in small pits for future consumption. Fresno kangaroo rats shelter in ground burrows, which are usually found in relatively light, crumbly soils in raised areas. In all species of San Joaquin kangaroo rats, each burrow system is typically occupied by a single adult individual.

Little is known about the mating behavior of Fresno kangaroo rats in the wild, although breeding is probably initiated in the winter after the onset of the rainy season. In captivity, gestation is 32 days, and young are weaned at 21 to 24 days. Average litter size is two (Culbertson 1946, Eisenberg and Isaac 1963). Young are born in the burrow, and remain there until they are fully furred and able to move about easily. Foraging is believed to start at about 6 weeks (Culbertson 1946).

The Fresno kangaroo rat was federally listed as endangered on January 30, 1985. On the same date, approximately 857 acres were designated as critical habitat for the species. Critical habitat is located in the Mendota Wildlife Area, the Alkali Sink Ecological Reserve, and on privately-owned land. Loss of habitat to cultivation, year-round grazing, and conversion of land to other uses continue to diminish the size and quality of extant, historical habitat. Coupled with the resulting fragmentation and isolation of habitat, these developments increase the probability of extinction. Flooding poses a high risk to protected habitat in Fresno County because of its proximity to the San Joaquin River. Other potential threats are the illegal use of rodenticides, competition with Heerman's kangaroo rats, and disease and predation, any of which could eliminate small, isolated populations (Williams and Germano 1993).

Tipton Kangaroo Rat

The Tipton kangaroo rat (*D. nitratoides nitratoides*), another subspecies of the San Joaquin kangaroo rat, is limited to arid-land communities occupying the valley floor of the Tulare Basin in level or nearly level terrain. The subspecies occupies alluvial fan and floodplain soils ranging from fine sands to clay-sized particles with high salinity. Today, much of the occupied remnants of the subspecies' range contain one or more species of sparsely scattered woody shrubs and a ground cover of mostly introduced and native annual grasses and forbs. Current occurrences are limited to scattered, isolated areas in Tulare and Kern counties.

Burrows of Tipton kangaroo rats are commonly located in slightly elevated mounds, the berms of roads, canal embankments, railroad beds, and at the bases of shrubs and fences where windblown soils accumulate above the level of surrounding terrain. Most aspects of food and foraging are identical to those of Fresno kangaroo rats.

Reproduction in Tipton kangaroo rats is similar to that of the Fresno kangaroo rat. Reproduction commences in winter and peaks in late March and early April. Most females appear to have only a single litter, though some adult females have two or more, and females born early in the year also may breed.

The Tipton kangaroo rat was federally listed as endangered on July 8, 1988. Critical habitat has not been designated. The principle reason for the decline of this subspecies was the loss of habitat as a result of agricultural conversion. Current threats come from industrial and agricultural-related developments, cultivation, the formation of heavy thatch by exotic grasses, urbanization, and flooding. The 1999 population estimate for this species was 190,200 individuals, down from a historic estimate of 17.2 million individuals (California Department of Fish and Game 2000f). In the mid-1990s, the population declined to all-time lows, and then began to slowly increase again. Overall, however, the species is still declining in numbers.

Stephens' Kangaroo Rat
The primary reference for this section is:
Massicot, P. 2002. Animal Info – Stephens' Kangaroo Rat. Available at http://animalinfo.org/species/.

References cited in this sections are internal to the above-referenced document. Full citations have been included in the Bibliography.

The Stephens' kangaroo rat (*Dipodomys stephensi*) occurs at elevations below about 2,000 feet in flat or gently rolling, often degraded, annual grassland. The entire geographic range of this species, which is estimated at approximately 1,000 square miles, is centered in the San Jacinto and Perris valleys of western Riverside County, California, with minor extensions south into San Diego County and north into San Bernardino County. Stephens' kangaroo rats are associated with locations where grass cover and bare ground are abundant, but where bush and rock are uncommon. Rainfall is an important factor in the species' ecology: Stephens' kangaroo rats show 10-fold fluctuations in population density related to regional rainfall.

Stephens' kangaroo rats are nocturnal and have a diet consisting of seeds. They seldom drink water because they are able to use water resulting from the chemical breakdown of their food. They also conserve moisture by coming out of their burrows at night when the humidity is highest. Reproductively active individuals have been found in every month of the year, although onset of estrus in females appears to be triggered by the start of winter, and estrous cycling ceases after plants disperse seeds (Price and Kelly 1994). The number of young per litter averages about 2.5. Weaning takes place at 18 to 22 days.

The Stephens' kangaroo rat was federally listed as endangered on September 30, 1988. Critical habitat has not been designated for this species. By 1938, only about 37% of the species' original habitat was estimated to have remained, and its range had become greatly fragmented. Accelerating urban development has led to a further degradation in available habitat. Currently, the species' remaining habitat occurs as small isolated patches embedded in rocky outcrops unsuitable for cultivation or as relatively extensive patches in protected watersheds (Price and Kelly 1994). Only 5% of its original habitat remains. The Stephens' kangaroo rat is currently threatened by urban development.

Morro Bay Kangaroo Rat
The Morro Bay kangaroo rat (*Dipodomys heermanni morroensis*), one of nine subspecies of Heermann's kangaroo rat, occurs in stabilized sand dune areas south of Morro Bay, California. Morro Bay kangaroo rats are essentially found only in disturbed areas; optimum habitat consists of the earlier successional stages of the coastal sagebrush community which occur on the old, stabilized dune terraces on the south and southeast sides of Morro Bay. Typical vegetation in this habitat is herbaceous annuals, with scattered woody perennial shrubs (coastal sagebrush, coyotebrush, yellow bush lupine and chamisso bush lupine, and buckwheat) no more than 2 feet in height. Shrub cover may be totally absent, or range as high as 60%; ground cover may vary from practically zero to 100% (Stewart 1958; Stewart and Roest 1960; Condon 1971, 1975; Roest 1973; Toyoshima 1978, 1979).

Early successional stages inhabited by kangaroo rats exist until about 15 to 30 years after an area has been cleared of vegetation, depending on the specific site. Succession involves a gradual increase in size and coverage of brushy species, and after 20 to 30 years the brush is too tall and dense for kangaroo rats. In earlier times, vegetation was cleared and succession restarted as a result of fires intentionally set by Native Americans; more recently brushy areas have been cleared by bulldozers for either development or cultivation. The animals quickly move into such areas, usually within the first year after clearing. If the area is cultivated, they move in after the first harvest of oats or other grain, or within the first year, if the land is allowed to lie fallow (Stewart 1958; Stewart and Roest 1960; Roest 1973; Toyoshima 1978, 1979). Large scale development efforts and to a lesser extent cultivation (oats/pasture) surround the known occupied habitat. Several roads surround the known occupied habitat and provide access to homes, schools, and shopping centers. Soil is essentially raw wind-blown sand (but not active dunes), anchored by the roots of the vegetation it supports. Burrows can readily be dug in this soft substrate by the animals. Kangaroo rats are not found on steeper slopes (over about 10 to 15%). They are known to occur in areas just above the highest tide level to an elevation of about 1,000 feet, but only in areas with sandy soil. Burrows cannot be dug in the heavy clay soils found elsewhere in the region.

Morro bay kangaroo rats feed on vegetation, obtaining sustenance primarily from seeds, but also from the leaves, stems, and fruits of plants (Stewart 1958; Stewart and Roest 1960). The subspecies is strictly nocturnal, and is active early in the evening, sometimes with another active period before dawn (Toyoshima 1979, Roest 1985)

Morro Bay kangaroo rats construct burrows, which usually include 2 to 3 rooms and numerous dead-end side pockets that are often filled with seed caches (Stewart 1958; Stewart and Roest 1960). Each kangaroo rat adult maintains and defends its own burrow system. Home ranges may overlap, although the animals are not truly social. In optimum habitats population densities vary from 1 animal per acre to over 30 per acre in optimum habitats (Stewart 1958; Stewart and Roest 1960; Condon 1971; Roest 1977, 1984; Toyoshima 1979).

Morro Bay kangaroo rats apparently have at least two breeding periods per year, and litter size varies from 2 to 4 young. Young Morro Bay kangaroo rats remain with their mother in her burrow until the age of about 5 to 6 weeks. The species is nonmigratory (Roest 1985).

The Morro Bay kangaroo rat was federally listed as endangered in 1970. On August 11, 1977, the USFWS designated critical habitat for the taxon in San Luis Obispo County. Threats to the Morro Bay kangaroo rat include the continued loss and fragmentation of habitat, predation from domestic animals, and destruction of burrows by vehicles and pedestrians. In addition, the small population size of the taxon makes it vulnerable to extinction from naturally occurring events such as drought and disease.

Utah Prairie Dog
The primary reference for this section is:
USFWS. 1991e. Utah Prairie Dog Recovery Plan. Denver, Colorado.

The Utah prairie dog (*Cynomys parvidens*) inhabits arid grassland in southwest Utah, and has the most restricted range of all prairie dog species in the United States. It is thought that the species’ range once extended across the desert almost to the Nevada-Utah state line. The Utah prairie dog presently occurs in principal concentrations in only three areas: the Awapa Plateau; the Paunsaugunt region along the East Fork and main stem of the Sevier River; and the West Desert region of eastern Iron County.

Because prairie dogs get most of their water from plants, there is a positive correlation between available moisture and prairie dog abundance and density. Prairie dogs appear to prefer swale type formations where moist herbage is available even during drought periods. Soil characteristics are an important factor in the location of Utah prairie dog colonies. A well-drained area is necessary for home burrows. The soil should be deep enough to allow burrowing to depths sufficient to provide protection from predators and insulation from environmental and temperature extremes. Prairie dogs must be able to inhabit a burrow system 3.3 feet underground without becoming wet. The vegetation height within the prairie dog colony must be low enough to allow standing prairie dogs to scan their environment for predators. For this reason, controlled grazing is compatible with prairie dog colonies (Crocker-Bedford 1975).

Prairie dogs are predominantly herbivores, with grasses the preferred food items during all seasons. The flowers and seeds of forbs such as alfalfa also are preferred. Although forbs other than alfalfa are not always highly preferred items, they may be critical to a prairie dog colony’s survival during drought. Prairie dogs have also been observed eating the flowering parts of shrubs, especially during the fall.

Because of the high mortality rate for juvenile males resulting from conflicts with other males, approximately two-thirds of the adult population is female. Female Utah prairie dogs are capable of giving birth annually to litters that average three to four young. The young are usually born in April, after a gestation period of about 30 days. Juvenile prairie dogs appear above ground at an age of 5 to 7 weeks. They attain adult size by October and reach sexual maturity at the age of 1 year. Adult males cease surface activity during August and September, and females follow suit several weeks later. Juvenile prairie dogs remain above ground 1 to 2 months longer than adults. Few prairie dogs are above ground from the first of November through mid-February, although they are not completely dormant in the winter.

The Utah prairie dog was federally listed as endangered on June 4, 1973, and was reclassified as threatened on May 29, 1984 because of increases in population numbers. Critical habitat has not been designated. The Utah prairie dog once maintained an ecological relationship with bison, which maintained shortgrass habitat, interspaced with patches of forbs and bare ground. However, the replacement of bison with cattle resulted in long-term overgrazing on prairie dog habitat, which has resulted in a reduction in habitat quality and a reduction in moisture availability in the vegetation. Past control programs targeting prairie dogs also contributed to the decline of the species. Habitat loss and poor habitat quality are immediate concerns for the remaining Utah prairie dogs. Most of the species' distribution occurs on privately-owned lands that are or will be largely developed for agricultural production and housing. Population numbers declined between 1989 and 1995, and then increased in 1996 and 1997 (USDI BLM 2003). Current populations are estimated at between 4,000 and 5,000 individuals.

Black-footed Ferret
The primary reference for this section is:
USFWS. 2000n. Establishment of a Nonessential Experimental Population of Black-footed Ferrets in North-Central South Dakota.

References cited in this section are internal to the above-referenced document. Full citations have been included in the Bibliography.

The black-footed ferret (*Mustela nigripes*) is the only ferret species native to North America. Its historical range extended from southern Canada south through the Great Plains, mountain basins, and semi-arid grasslands of the western United States. Black-footed ferrets depend almost exclusively on prairie dogs for food, shelter, and denning (Henderson et al. 1969, Forrest et al. 1985). The range of the ferret coincides with that of three prairie dog species (Anderson et al. 1986), and ferrets with young have been documented only in the vicinity of active prairie dog colonies. Historically, black-footed ferrets have been reported in association with black-tailed prairie dog, white-tailed prairie dog, and Gunnison's prairie dog towns.

Prairie dogs make up the vast majority of the black-footed ferret's diet, and ferrets occupy underground prairie dog burrows during periods of inactivity. Other food sources are mice, rabbits, rats, birds, reptiles, insects, and carrion. Breeding occurs during March and April. The gestation period is 41 to 45 days, after which a litter of three to four young is produced. Ferrets develop quickly, reaching sexual maturity by September.

Substantial reductions in both prairie dog numbers and distribution occurred during the last century as a result of widespread poisoning of prairie dogs, the conversion of native prairie to farmland, and outbreaks of sylvatic plague, particularly in the southern portions of prairie dog ranges in North America. Sylvatic plague, which arrived from Asia in approximately 1900, is an exotic disease foreign to the evolutionary history of prairie dogs, which have little or no immunity to it. Black-footed ferrets also are highly susceptible to sylvatic plague. This severe reduction in the availability of the ferret's principal prey, in combination with other factors such as secondary poisoning from prairie dog toxicants, resulted in the near extinction of the black-footed ferret in the wild by 1980. In 1974, a remnant wild population of ferrets in South Dakota, originally discovered in 1964, abruptly disappeared. Afterwards, the species was believed to be extinct; however, in 1981 a small population of ferrets was discovered near Meeteetse, Wyoming. In 1985 to 1986, the Meeteetse population declined to only 18 animals due to outbreaks of sylvatic plague and canine distemper. Following this critical decline, the remaining individuals were taken into captivity in 1986 to 1987 to serve as founders for a captive-propagation program. Since that time, captive-breeding efforts have been highly successful and have facilitated ferret reintroductions in several areas of formerly occupied range. These reintroductions, however, have met with limited success.

The black-footed ferret was federally listed as endangered on March 11, 1967. This endangered status applies to the entire range of the species, except where reintroduced and designated as a non-essential experimental population. Such populations occur in Arizona, Colorado, Montana, South Dakota, Utah, and Wyoming. Critical habitat has not been designated for the species.

Effects of Vegetation Treatments on Kangaroo Rats, the Utah Prairie Dog, and the Black-footed Ferret

Effects Common to All Treatment Methods

Indirect Effects. Fuels reduction treatments would have a long-term positive effect on these mammal species by reducing the likelihood of a future wildfire that could drive one or more species to extinction. In addition, removal of non-native vegetation is likely to have either a positive effect or no effect, depending on the species. Kangaroo rats can thrive in annual grasslands that have a large component of non-native species, including red brome. However, excessive amounts of non-native species, especially shrubby species, can degrade habitat used by these small mammals. The invasion of vegetation onto Fresno kangaroo rat habitat has been linked to declines in species numbers (Morrison et al. 1996). In desert shrubland and grassland habitats, the invasion of woody species as a result of fire suppression has eliminated suitable prairie dog and black-footed ferret habitat (USDI BLM 2002b). For all of the species considered in this section, the control of shrubs and other weedy species that invade grassland habitats and reduce the degree of openness would have a long-term positive effect on habitat.

Prescribed Fire Treatments

Direct Effects. Small mammals tend to seek refuge underground or in sheltered places within a burn (Smith 2000). Therefore, depending on the intensity of the fire and the availability of burrows and hiding places, there would likely be some direct mortality to the mammal species considered in this section.

Indirect Effects. Because all of these species have small populations and occur on very fragmented patches of habitat, a random catastrophic event, such as a severe fire, could lead to extinction of the species. Because kangaroo rats and prairie dogs predominantly eat plant materials, prescribed fire could temporarily reduce the availability of food for these species. These effects would be most severe during a drought or any other period of low food availability.

Lack of fire (or another suitable disturbance) to control the density of vegetation on sites that support these species has been identified as a possible threat (USFWS 1991d, 1998v). Therefore, prescribed fire can be used as a management tool to maintain the open, grassy habitat conditions favored by these species.

Mechanical Treatment Methods

Direct Effects. Given that these mammal species can escape into underground burrows during mechanical control activities, direct effects to these TEP species by this treatment method are unlikely. It is possible that heavy equipment could collapse some shallow burrows, but major effects to the extensive burrow systems utilized by these species are unlikely.

Indirect Effects. Widescale removal of vegetation in suitable remaining habitat could reduce the availability of food for the kangaroo rats and the prairie dog. During a period of reduced food availability, such as a drought, such vegetation removal could have negative effects on populations of these species. However, removal of tall invasives or shrub species would have long term positive effects on habitat.

Manual Treatment Methods

Direct and Indirect Effects. Removal of vegetation and other materials by hand is unlikely to cause a major disturbance to kangaroo rat, prairie dog, or black-footed ferret habitat.

Biological Control Treatments

Domestic Animals

Indirect Effects. One of the species considered in this section, the giant kangaroo rat, can survive in areas that have been grazed to the point where almost no plant material remains (USFWS 1998v). However, it is not known whether they could survive indefinitely if those grazing intensities were sustained. In one area, moderate levels of grazing by domestic animals have maintained nearly optimum conditions for giant kangaroo rats in what is among the better quality habitat remaining.

For all of the species considered in this section, some amount of grazing either appears to have no effect or a beneficial effect on habitat by keeping the vegetation sparse and low. These mammals require low grass conditions to detect predators and quickly escape into burrows in dangerous situations. In addition, grazing helps prevent an excessive accumulation of mulch. The Utah prairie dog historically had an ecological relationship with bison, which maintained short grass habitat, but which moved constantly and seldom overgrazed (USFWS 1991d). Reestablishing controlled, moderate levels of grazing has been recommended for many of these mammals by the USFWS. For all of these species, however, excessive grazing is likely to be a threat because it would lead to degradation of the habitat.

Other Biological Control Agents

Direct and Indirect Effects. Biological control methods are unlikely to have substantial effects on these TEP species or their habitat. These agents target specific, undesirable plant species, and have a gradual effect on vegetation. However, since there is limited knowledge about the long-term effects of these agents, it is possible that unanticipated impacts to the ecosystems these species inhabit could occur.

Herbicides

Direct Effects. Direct spray of listed kangaroo rats, prairie dogs, or black-footed ferrets would be unlikely during herbicide applications, since these animals would be able to flee the site or run into underground burrows. Nonetheless, an inadvertent direct spray of these species by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate, could potentially result in adverse health effects (see Table 6-2). In addition, if these TEP kangaroo rats, prairie dogs, or ferrets were to come into contact with vegetation that had been sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate, adverse health effects would be possible, according to the ERAs.

The listed kangaroo rats and the Utah prairie dog are herbivores. Therefore, they could be indirectly affected by herbicide treatments by ingesting plant materials that have been directly contaminated during spray applications. According to the ERAs, ingestion of plant materials sprayed by 2,4-D, diquat, or diuron at the typical application rate, or by bromacil, fluridone, glyphosate, hexazinone, or tebuthiuron at the maximum application rate, could result in adverse health effects (see Table 6-5).

The black-footed ferret is a carnivore, feeding almost exclusively on prairie dogs. Therefore, ferrets could be indirectly exposed to herbicide chemicals by consuming prey items that have been directly exposed to herbicides. Since most prey items would avoid this exposure, such a scenario is improbable. Nonetheless, the ERAs predicted that adverse health effects to black-footed ferrets could occur if they ingested prey sprayed by 2,4-D or diuron at the typical application rate, or by bromacil, diquat, or triclopyr at the maximum application rate. Since the ERA for hexazinone did not assess the potential risks to carnivorous species through ingestion of contaminated prey, the potential for adverse effects to prairie dogs from exposure to hexazinone via this exposure pathway cannot be determined.

Indirect Effects. Herbicide treatments could affect the habitat of kangaroo rats and prairie dogs by temporarily reducing the amount of forage available to these species. Over the long term, effects to kangaroo rats would be minimal, since they commonly inhabit grasslands with a large component of non-native species. However, all of these mammal species could benefit over the long term if weedy species that alter the structure of grassland habitat were controlled, maintaining or improving open conditions.

Conservation Measures

The following programmatic-level conservation measures would be required to ensure that the proposed vegetation treatments did not adversely affect listed kangaroo rat species, the Utah prairie dog, or the black-footed ferret, or their habitats:

- Prior to conducting vegetation treatments, survey areas scheduled to receive treatments for listed kangaroo rats, Utah prairie dogs, and black-footed ferrets.
- Incorporate these species and their habitat into management plans developed for treatment activities.
- Avoid vegetation treatments during drought conditions.
- Where possible, perform treatments during the hibernation period.
- Do not use 2,4-D in listed kangaroo rat, Utah prairie dog, or black-footed ferret habitats; do not broadcast spray 2,4-D within ¼ mile of listed kangaroo rat, Utah prairie dog, or black-footed ferret habitat.
- Do not use diquat or diuron in listed kangaroo rat or Utah prairie dog habitats; do not broadcast spray these herbicides within ¼ mile of listed kangaroo rat or Utah prairie dog habitat.

Additional conservation measures for kangaroo rats and the Utah prairie dog:

- Where feasible, avoid use of the following herbicides in listed kangaroo rat and Utah prairie dog habitat: bromacil, clopyralid, fluridone, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, tebuthiuron, and triclopyr.
- Do not broadcast spray clopyralid, glyphosate, hexazinone, picloram, or triclopyr in listed kangaroo rat or Utah prairie dog habitat; do not broadcast spray these herbicides in areas adjacent to listed kangaroo rat or Utah prairie dog habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, imazapyr, fluridone, metsulfuron methyl, or tebuthiuron in or near listed kangaroo rat or Utah prairie dog habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of bromacil, glyphosate, hexazinone, tebuthiuron, or triclopyr to vegetation in listed kangaroo rat or Utah prairie dog habitat, utilize the typical, rather than the maximum, application rate.

Additional conservation measures for the black-footed ferret:

- Where feasible, avoid use of the following herbicides in black-footed ferret habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr.
- Do not broadcast spray clopyralid, diuron, glyphosate, hexazinone, picloram, or triclopyr in black-footed ferret habitat; do not broadcast spray these herbicides in areas adjacent to black-footed ferret habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, diquat, imazapyr, or metsulfuron methyl in or near black-footed ferret habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in black-footed ferret habitat, utilize the typical, rather than the maximum, application rate.

Individual projects would be subject to review at the local level, during which additional conservation measures could be identified as necessary to protect these species.

Determination of Effects

Assuming that any type of vegetation treatment method could occur anywhere on public lands, the proposed action is **likely to adversely affect** listed kangaroo rat species, the Utah prairie dog, and the black-footed ferret. However, with the implementation of both programmatic- and project-level mitigation (as discuss in the previous section, Conservation Measures) to avoid or minimize these effects, vegetation treatments would likely have long-term benefits for these species and their habitats. Therefore, implementation of these measures would likely reduce the effects determination to **not likely to adversely affect** listed kangaroo rat species, the Utah prairie dog, or the black-footed ferret.

Bighorn Sheep

The primary references for this section are:
USFWS. 2000o. Final Rule To List the Sierra Nevada Distinct Population Segment of the California Bighorn Sheep as Endangered. Federal Register 65(1): 20-30;

and

USFWS. 1998x. Endangered Status for the Peninsular Ranges Population Segment of the Desert Bighorn Sheep in Southern California. Federal Register 63(52): 13134-13150.

References cited in this section are internal to the above-referenced documents. A complete list of these references is available from the USFWS, Ventura Field Office, Ventura, California; and the Carlsbad Field Office, Carlsbad, California.

Two populations of bighorn sheep (*Ovis canadensis* and *O. c. californiana*) occur in the project area: Peninsular Ranges bighorn sheep (*O. canadensis*), which inhabit the Peninsular Mountain Ranges of southern California and Mexico, and Sierra Nevada bighorn sheep (*O. c. californiana*), which occupy the Sierra Nevada mountain range located along the eastern boundary of California.

The Peninsular Ranges bighorn sheep occurs on open slopes in hot and dry desert regions where the land is rough, rocky, sparsely vegetated, and characterized by steep slopes, canyons, and washes. In general, sheep inhabit elevations ranging between 300 and 4,000 feet, in areas where annual precipitation averages less than 4 inches and daily high temperatures in the summer average 104 °F. Sheep use caves and rock outcrops as shelters during inclement weather, and ridge benches or canyon rims adjacent to steep slopes or escarpments as lambing areas. From May through October, populations aggregate near water sources and engage in breeding activities.

Bighorn sheep are diurnal, with a daily activity pattern that consists of both feeding and resting periods. The primary source of food for desert-associated bighorn sheep is browse, which includes such species as brittlebrush, mountain mahogany, Russian thistle, bursage, mesquite, and palo verde. Sheep may also eat the pulp and fruits of various cactus species and graze on native grasses.

Bighorn sheep have a gestation period of 5 to 6 months, and produce one lamb per year. Lambing occurs between January and June, peaking between February and May. Ewes with their lambs frequently inhabit areas where there are a diversity of slopes and exposures for escape cover and shelter from heat. Lambs are weaned between 1 and 7 months of age, and are independent of ewes by their second spring (Cowan and Geist 1971).

The Sierra Nevada bighorn sheep is found in alpine and subalpine zones during the summer months, and on high, windswept ridges or lower elevation sagebrush-steppe habitat in the winter. Summer habitat is primarily open areas that are rough, rocky, sparsely vegetated, and characterized by steep slopes and canyons (Wehausen 1980, Sierra Nevada Bighorn Sheep Interagency Advisory Group 1997), at elevations between 10,000 and 14,000 feet. In the winter, the sheep exhibit a preference for south-facing slopes. The steep, rugged terrain is necessary for escape, lambing, and bedding, and adjacent areas of low growing vegetation are required for food. An adequate supply of water is also necessary, as are travel routes that link all of the habitat areas. The sheep also require areas that are free of competition from other grazing ungulates, in particular domestic sheep. Sierra Nevada bighorn sheep are primarily grazers, but may browse woody vegetation as well, and their diet commonly includes a mixture of grasses and other herbaceous plants, as well as shoots, twigs, and leaves of trees and shrubs.

The general reproductive cycle is the same as that of Peninsular Range populations; however, in Sierra Nevada populations, breeding takes place in November, and lambing occurs from late April to early July, peaking in May or June.

The Peninsular Ranges population segment was federally listed as endangered on March 18, 1998, and the Sierra Nevada population segment was federally listed as endangered on January 3, 2000. A total of approximately 844,897 acres in Riverside, San Diego, and Imperial counties, California, were designated as critical habitat for Peninsular Ranges bighorn sheep on February 1, 2001. Critical habitat has not yet been designated for the Sierra Nevada population segment. Threats to the Peninsular population include the synergistic effects of disease; low recruitment; habitat loss, degradation, and fragmentation; non-adaptive behavioral responses associated with residential and commercial development; and high predation rates coinciding with low bighorn sheep population numbers. Threats to the Sierra Nevada population are similar, with small population size, mountain lion predation, disease, naturally-occurring environmental events, and genetic problems associated with small population size.

The Sierra Nevada population declined in the 1980s and early 1990s, hitting a low of about 100 individuals in 1995 (Sierra Nevada Bighorn Sheep Foundation, no date). Over the past few years, however, populations have increased, and were estimated at 150 adults in 2000, and 250 individuals in 2001. The population of the Peninsular Ranges bighorn sheep has decreased over the past 20 years from about 1,100 individuals (in 1974) to as few as 300 individuals (USDI BLM 2002b). In 2000, the population estimate was 400 adults.

Effects of Vegetation Treatments on Bighorn Sheep

Effects Common to All Methods

Indirect Effects. All treatments that reduce accumulated fuels in bighorn sheep habitats would be expected to have a positive effect on sheep habitat by reducing the threat of future catastrophic wildfire. In addition, the removal of old, decadent vegetation can stimulate more nutritious bighorn sheep forage, improving habitat for a number of years following treatments (USDA Forest Service 2002). Treatment methods that reduce cover would also have a positive effect on the species' habitat. Bighorn sheep prefer the high visibility of open habitats, which make it easier for them to detect predators, and to communicate with one another (i.e., alarm postures) (Geist 1971, Risenhoover and Bailey 1985). Bighorn sheep have been observed to shift their habitat use to logged and burned areas after treatments (Smith et al. 1999).

Bighorn sheep rely on forested areas for temporary refuge from adverse weather or cover from predators. Removal of vegetation could reduce the availability of these habitats. However, sheep should not be affected as long as some cover was available in the area. Because bighorn sheep inhabit remote areas, vegetation treatments could affect habitat suitability by increasing the accessibility of habitat areas by humans. Creation of roads to access remote areas for treatments would decrease their remoteness and could result in an increased human intrusion into sheep habitat.

Prescribed Fire Treatments

Direct Effects. Direct effects to bighorn sheep from fire would be unlikely, since most animals would be able to move out of the burn area during the fire. Newborn animals would be the most susceptible to harm. In addition, some injury could occur through smoke inhalation.

Indirect Effects. Prescribed fire in forests and woodland that resulted in canopy openings, could yield increased productivity in the shrub and herb layers, increasing the amount of forage available to bighorn sheep (Bradley et al. 1992). The amount of habitat available to the species would also be increased. Historically, fire was an important factor in sheep habitats. Fire can slow/prevent the succession of shrubs and trees onto alpine grasslands, increase the palatability and productivity of important forage species, and possibly aid in parasite control. In mountain shrublands, fire can improve nutrition by increasing the availability of green grass species. Past experiments conducted in British Columbia enhanced range for a related species of sheep, and produced faster growing, larger sheep than an unburned control area (Elliott 1978).

Burning could also help reduce disease rates in bighorn sheep populations, as animals tend to disperse after a fire, which would reduce animal densities and rates of infection (Peek et al. 1985).

Mechanical Treatment Methods

Direct Effects. Thinning and fuels reduction by mechanical methods would be unlikely to have negative direct effects on bighorn sheep, as animals would be able to avoid the areas where work was taking place. There could be some disturbances associated with noise and the presence of humans. However, these effects would likely be minor.

Indirect Effects. Removal of invading trees and opening up forested or wooded areas would have positive effect on bighorn sheep habitat, for reasons described above. However large-scale removal of vegetation from an area used by sheep would have negative effects if the coverage of shrubs and herbs used for forage decreased, and if temporary hiding and thermal cover refuges were eliminated.

Manual Treatment Methods

Direct and Indirect Effects. Removal of vegetation and fuels by hand would not have substantial effects on bighorn sheep habitat. It is expected that the amounts of vegetation removed by this method would be small. Some forage might be removed, but would have minor effects.

Biological Control Treatments

Domestic Animals
<u>Direct Effects</u>. Use of domestic sheep to contain weeds could have adverse effects on bighorn sheep populations. Chance encounters between wild and domestic sheep may result in the transfer of viruses, parasites, and bacteria from domestic sheep to bighorn sheep. In particular, there is a respiratory pathogen that leads to pneumonia and has been observed to cause lamb mortality for 3 to 5 years (USDA Forest Service 2002). Domestic sheep should not be brought in to treat undesirable vegetation in areas where bighorn sheep occur.

<u>Indirect Effects</u>. All types of grazing ungulates brought into bighorn sheep habitat would also be expected to have a negative effect by competing with bighorn sheep for preferred forage plants. These effects would be cumulative with those of competition that is already occurring from non-domestic ungulates, including elk, deer, and wild horses and burros. Bighorn sheep are poor competitors with both wild and domestic ungulates, and some social intolerance of these species by sheep also occurs (Chapman and Feldhamer 1982).

Other Biological Control Agents
<u>Direct and Indirect Effects</u>. Release of biological control agents into bighorn sheep habitat would be unlikely to affect populations of bighorn sheep. Biological control agents target weed species, and their effects are gradual. There is the chance, however, that unanticipated impacts to the ecosystem (and therefore bighorn sheep or their habitat) could occur as a result of these agents. Such an occurrence is not reasonably foreseeable.

Herbicides

Direct Effects. Because bighorn sheep are large, mobile animals, it is unlikely that they would be sprayed inadvertently during herbicide treatments. However, if a direct spray of 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or of imazapyr or metsulfuron methyl at the maximum application rate, were to occur, adverse health effects to sheep could potentially occur (see Table 6-2). Bighorn sheep could also come into contact with sprayed foliage after the application. Via this exposure pathway, adverse health effects to sheep could occur if vegetation was sprayed by 2,4-D at the typical application rate, or by glyphosate, hexazinone, or triclopyr at the maximum application rate.

Bighorn sheep could conceivably ingest plant materials at a treatment site shortly following the herbicide application. Under such a scenario, ingestion of plants materials sprayed by 2,4-D, bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, Overdrive®, picloram, or triclopyr at the typical application rate, or by imazapyr, metsulfuron methyl, or tebuthiuron at the maximum application rate, would potentially result in adverse health effects to sheep (see Table 6-5).

Indirect Effects. Because bighorn sheep occur in sparsely vegetated habitats, herbicide treatments could adversely affect bighorn sheep over the short term by further reducing the amount of forage available. These effects would be temporary, but during a particularly sparse year could affect sheep populations. Over the long term, however, control of non-native species should improve the quality of forage in these habitats.

Conservation Measures

The following programmatic-level conservation measures are the minimum steps required of the BLM to ensure that bighorn sheep and their habitats would not be impacted by vegetation treatment activities. Additional project-specific conservation measures would be identified at the local level, as appropriate.

- Prior to treatment activities, survey suitable habitat for evidence of use by bighorn sheep.
- Do not use domestic animals as a vegetation treatment in bighorn sheep habitat.
- When planning vegetation treatments, minimize the creation of linear openings that could result in permanent travel ways for competitors and humans.
- Obliterate any linear openings constructed within bighorn sheep habitat in order to deter future uses by humans and competitive species.
- Where feasible, time vegetation treatments such that they do not coincide with seasonal use of the treatment area by bighorn sheep.
- Do not broadcast spray herbicides in key bighorn sheep foraging habitats.
- Do not use 2,4-D in bighorn sheep habitat; do not broadcast spray 2,4-D within ¼ mile of bighorn sheep habitat.
- Where feasible, avoid use of the following herbicides in bighorn sheep habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, Overdrive®, picloram, and tebuthiuron, and triclopyr.
- Do not broadcast spray bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, Overdrive®, picloram, or triclopyr in bighorn sheep habitat; do not broadcast spray these herbicides in areas adjacent to bighorn sheep habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying imazapyr, metsulfuron methyl, or tebuthiuron in or near bighorn sheep habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, imazapyr, metsulfuron methyl, tebuthiuron, or triclopyr to vegetation in bighorn sheep habitat, utilize the typical, rather than the maximum, application rate.

Determination of Effects

Assuming that any type of proposed vegetation treatment could occur anywhere on public lands, including habitats that support listed populations of bighorn sheep, the proposed action would be **likely to adversely affect** the Peninsular Range and/or its critical habitat as listed in Table 1-1, and Sierra Nevada populations of bighorn sheep. However, by implementing programmatic- and project-level conservation measures, as discussed in the previous section, the BLM would be able to minimize or avoid these effects, and the treatments would likely benefit bighorn sheep populations. Thus, the effects determination at the local level would be reduced to **not likely to adversely affect** bighorn sheep or their habitats.

Wolves

The primary reference for this section is:

USFWS. 2000p. Proposal to Reclassify and Remove the Gray Wolf From the List of Endangered and Threatened Wildlife in Portions of the Coterminous United States; Proposal To Establish Three Special Regulations for Threatened Gray Wolves; Proposed Rule. Federal Register Volume 65(135): 43449-43496.

References cited in this section are internal to the above-referenced document. A complete list of these references is available from the USFWS Region 3 Office, Fort Snelling, Minnesota.

Gray wolves (*Canis lupus*) are the largest wild members of the dog family. The species historically occurred across most of North America, Europe, and Asia. In North America, wolves occurred from the northern reaches of Alaska, Canada, and Greenland to the central mountains and the high interior plateau of southern Mexico. The only areas of the contiguous U.S. that apparently lacked gray wolves are much of California and the Gulf and Atlantic coastal plain south of Virginia. In addition, wolves were generally absent from the extremely arid deserts and the mountaintops of the western United States (Goldman 1944, Hall 1959, Mech 1974). The cultural attitudes of European settlers, coupled with perceived and real conflicts between wolves and human activities along the frontier, led to widespread persecution of wolves. Poisons, trapping, and shooting—spurred by federal, state, and local government bounties—resulted in extirpation of the species from more than 95% of its range in the 48 coterminous states.

Wolves are predators of large animals. Wild prey species in North America include white-tailed deer, mule deer, moose, elk, woodland caribou, barren ground caribou, bison, muskox, bighorn sheep, Dall sheep, mountain goat, beaver, and snowshoe hare, with small mammals, birds and large invertebrates sometimes being taken (Mech 1974, Stebler 1944, Wisconsin Department of Natural Resources 1999a). Wolves may also feed on domestic animals (Paul 1999).

Wolves are social animals, normally living in packs of 2 to 10 members. Packs are primarily family groups consisting of a breeding pair, their pups from the current year, offspring from the previous year, and occasionally an unrelated wolf. Packs occupy, and defend from other packs and individual wolves, a territory of 20 to 214 square miles (though typically larger in the Rocky Mountains). Normally, only the top-ranking male and female in each pack breed and produce pups. Litters are born from early April into May; they can range from 1 to 11 pups, but generally contain 4 to 6 pups (USFWS 1992a, Michigan Department of Natural Resources 1997). Yearling wolves frequently disperse from their natal packs, although some remain with their pack. Dispersers may become nomadic and cover large areas as lone animals, or they may locate suitable unoccupied habitat and a member of the opposite sex and begin their own territorial pack. Dispersal movements of over 500 miles have been documented (Fritts 1983).

As many as 24 distinct subspecies of gray wolf have been recognized, and federal listings were originally at the subspecies level. On March 9, 1978, the gray wolf was relisted as endangered throughout the conterminous 48 States and Mexico. In Minnesota, however, the gray wolf was reclassified to threatened. In addition, critical habitat was designated in Isle Royale National Park, Michigan, and Minnesota. On November 22, 1994, areas in Idaho, Montana, and Wyoming were designated as nonessential experimental populations in order to initiate gray wolf reintroduction projects in central Idaho and the Greater Yellowstone area. On January 12, 1998, a nonessential experimental population was established for the Mexican gray wolf in portions of Arizona, New Mexico, and Texas.

On July 13, 2000, the USFWS proposed the establishment of four distinct population segments (DPSs) for the gray wolf in the United States and Mexico. Under this proposal, gray wolves in the Western Great Lakes DPS (North Dakota, South Dakota, Minnesota, Wisconsin, and Michigan), the Western DPS (Washington, Oregon, Idaho, Montana, Wyoming, Utah, Colorado, and parts of Arizona and New Mexico), and the Northeastern DPS (New York, Vermont, New Hampshire, and Maine) would be reclassified from endangered to threatened, except where already classified as an experimental population or as threatened. Gray wolves in the Southwestern (Mexican) DPS (portions of Arizona and New Mexico) would retain their endangered status. All three existing gray wolf experimental population designations would be retained. In all other areas of the 48 conterminous states, gray wolves would be removed from the protections of the ESA. Gray wolf populations in all DPSs, except the Southwestern DPS, have shown steady increases from the late 1970s to the present. As of the 1998/1999 census, there were a total of 22 gray wolves in the Southwestern DPS. Gray wolves are still threatened by direct human-caused mortality, and potentially by habitat loss.

Effects of Vegetation Treatments on Wolves

Effects Common to All Treatment Methods

Indirect Effects. Habitat preferences by wolves appear to be more dependent on the availability of desired prey than on cover type. Although most treatment methods would result in some modification of wolf habitat, it is the changes in the habitats of prey species that would have the most effect on wolves. Since some prey species prefer open habitat and others prefer dense habitat, fuels reduction treatments would benefit some species while adversely affecting others (Agyagos et al. 2001). Treatments that reduce fuels would also reduce the risk of a future catastrophic wildfire, which would probably have some long-term benefits for wolves and their prey. In addition, treatments that reduce the cover of non-native species should have some long-term benefits by helping to restore native plant communities to wolf habitats, possibly increasing the diversity of food sources.

Prescribed Fire Treatments

Direct Effects. Because wolves are highly mobile animals, direct effects resulting from prescribed fire are unlikely.

Indirect Effects. Wolves are able to disperse long distances, opportunistically forage on a variety of prey species, and occupy a variety of habitat types (Agyagos et al. 2001). Many fire-dependent species such as beaver, elk, moose, and deer are also wolf prey species (Hansen et al. 1973; Kramp et al. 1983). Therefore, increases in populations of these species, as often occurs shortly after a prescribed fire, are often linked to an increase in wolves. One study indicated that enough early successional plant communities must exist within a gray wolf pack's territory to support a surplus of deer, moose, and beaver for prey (Heinselman 1973).

Mechanical Treatment Methods

Direct Effects. Mechanical treatments would be unlikely to directly affect wolves, which are large and very mobile animals.

Indirect Effects. There could be some mortality of small mammals and other animals on which wolves feed, but these effects would be short-term in nature. In addition, large-scale removal of vegetation might eliminate habitat or food for certain prey species, but may favor other prey species. The noise associated with the use of heavy equipment would be likely to disturb wolves and their prey species, but overall, these impacts should be minor, and of short duration.

Manual Treatment Methods

Direct and Indirect Effects. Removal of vegetation and fuels using manual control would be unlikely to affect wolves or their habitat. Any disturbances to wolves or prey species by humans would be minor and of short duration.

Biological Control Treatments

Domestic Animals
Indirect Effects. Use of domestic animals to contain weeds within wolf habitats would be unlikely to negatively affect wolf populations. In fact, wolves are often attracted to grazed lands and other open areas because of the presence of ungulates in these habitats. Ungulates, including any animals brought in to graze weeds, are a source of prey to wolves, which often follow herds. Therefore, though cultural control could benefit wolves by providing them with a source of prey, this treatment method is obviously not suitable for use in wolf habitats because of risks to the domestic animals that would be used.

Other Biological Control Agents
Direct and Indirect Effects. The use of biological control agents would be unlikely to affect wolves or their habitat. These agents target a particular invasive plant species, and have a gradual effect. Given the unknown long-term effects of biological control agents, there is always a chance that their release could result in unanticipated impacts to the ecosystem, which could in turn affect wolves, their prey, or their habitats. However, these effects are not reasonably foreseeable.

Herbicides

Direct Effects. It is unlikely that wolves would be directly exposed to herbicides, since animals would avoid treatment sites, and are large enough that herbicide applicators should be able to see them. Nonetheless, adverse health effects could occur if one or more wolves were sprayed unintentionally by 2,4-D, clopyralid, glyphosate, hexazinone, picloram, or triclopyr at the typical application rate, or by imazapyr or metsulfuron methyl at the maximum application rate (see Table 6-2). Following an herbicide treatment, wolves could potentially suffer adverse effects from dermal contact with foliage that was treated with 2,4-D at the typical application rate, or with glyphosate, hexazinone, or triclopyr at the maximum application rate.

As a carnivore that feeds on large animals, it is unlikely that the prey items of wolves would themselves be directly exposed to herbicides during chemical treatments by the BLM. However, the ERAs did indicate the potential for adverse health effects to occur if a wolf consumed a prey item that had been sprayed by 2,4-D or diuron at the typical application rate, or by bromacil, diquat, or triclopyr at the maximum application rate (see Table 6-5). The potential for adverse effects to wolves from exposure to hexazinone via ingestion of contaminated prey cannot be determined.

Indirect Effects. Herbicide treatments would have few effects on wolf habitat or prey. Over the long term, treatments that reduce the cover of non-native species could benefit habitat by helping to restore native plant communities and possibly increasing the diversity of food sources.

Conservation Measures

Although the proposed vegetation treatments would not be likely to have negative effects on wolves or their habitat, the following programmatic-level conservation measures are recommended to ensure protection of the species. Additional or more specific guidance would also be provided at the project level, as appropriate.

- Avoid human disturbance and/or associated activities within 1 mile of a den site during the breeding period (as determined by a qualified biologist).
- Avoid human disturbance and/or associated activities within 1 mile of a rendezvous site during the breeding period (as determined by a qualified biologist).
- Do not use 2,4-D in areas where gray wolves are known to occur; do not broadcast spray within ¼ mile of areas where gray wolves are known to occur.
- Where feasible, avoid use of the following herbicides in gray wolf habitat: bromacil, clopyralid, diquat, diuron, glyphosate, hexazinone, imazapyr, metsulfuron methyl, picloram, and triclopyr.

- Do not broadcast spray clopyralid, diuron, glyphosate, hexazinone, picloram, or triclopyr in gray wolf habitat; do not broadcast spray these herbicides in areas adjacent to gray wolf habitat under conditions when spray drift onto the habitat is likely.
- If broadcast spraying bromacil, diquat, imazapyr, or metsulfuron methyl in or near gray wolf habitat, apply at the typical, rather than the maximum, application rate.
- If conducting manual spot applications of glyphosate, hexazinone, or triclopyr to vegetation in gray wolf habitat, utilize the typical, rather than the maximum, application rate.

Determination of Effects

Because of the size and mobility of the gray wolf, and its lack of dependence on a specific vegetation type, most of the proposed vegetation treatments should not affect the gray wolf. However, since some of the herbicides proposed for use by the BLM could cause health effects to wolves under direct spray or ingestion of contaminated prey scenarios, the proposed treatments would be **likely to adversely affect** the gray wolf, without precautionary measures in place. By implementing programmatic- and project-level conservation measures, as discussed in the previous section, adverse effects could be avoided, reducing the determination to **not likely to adversely affect** gray wolves or their habitats.

Cumulative Effects for Terrestrial Animals

Private, tribal, and non-federal agency actions occurring on or near public lands that could affect terrestrial animals discussed in this BA. Public activities, including recreation, OHV use, collecting, and hunting could impact listed species and species proposed for listing. Direct effects include removal of terrestrial animals by collectors, hunters, or other recreationists, and the harming of terrestrial animals by OHVs, pack horses and mules, or other recreationists and public land users. Indirect effects include loss or degradation of habitat, and disturbance, associated with these activities that affects wildlife behavior and productivity.

Livestock grazing on public lands could impact TEP terrestrial animals. Livestock could directly affect TEP organisms by trampling them. Indirect effects would include erosion and degradation of water quality, loss of forage and cover, and removal of water in areas of heavy livestock use that could affect TEP species.

TEP terrestrial animals are at risk from private, industrial activities occurring on public lands, including mining, oil and gas and ROW development, and timber harvest activities that would potentially disturb large areas of habitat. Direct impacts would include injury or mortality. Indirect impacts of clearing and for construction include loss of habitat, water pollution, and introduction of noxious weeds and other invasive vegetation. If herbicides were used to maintain vegetation on ROW or at facilities, nearby TEP terrestrial animals could be exposed to these chemicals.

Tribal actions that could harm TEP terrestrial animals include hunting and collecting of animals for traditional lifeway uses. Indirect effects from tribal actions would be similar to those associated with recreation.

TEP terrestrial animals could be indirectly harmed by activities occurring on non-federal lands adjacent to public lands. For example, herbicides applied to nearby agricultural lands or rangelands could drift onto public lands and harm TEP terrestrial animals. In addition, there could be impacts to air and water quality from the spread of weeds or from wildfire associated with activities occurring off public lands.

Conservation measures (see below) and SOPs identified in this BA and in the PEIS and PER would reduce the likelihood of terrestrial animals being impacted by vegetation treatments and non-federal activities on public lands. The BLM would conduct surveys for TEP terrestrial animals, and an analysis of project impacts to these species would be done under NEPA as part of the permitting and siting process for land-disturbing activities conducted by private entities on public lands. The BLM would conduct local level consultation with the Services, as discussed in Chapter 3, for actions that have potential to affect TEP terrestrial animals. The BLM would coordinate with tribes having an interest in TEP terrestrial animals on public lands, or potentially affecting these species, to minimize impacts to these species.

CHAPTER 7

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CHAPTER 8

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APPENDIX A

ESSENTIAL FISH HABITAT ASSESSMENT

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APPENDIX A

ESSENTIAL FISH HABITAT ASSESSMENT

Action Agency

U.S. Department of Interior Bureau of Land Management

Project Name

Vegetation Treatments on Bureau of Land Management Lands in 17 Western States Programmatic EIS.

Introduction

The BLM is seeking to expand its vegetation treatment program from current levels (approximately 2 million acres annually) to approximately 6 million acres annually in order to improve public land health by slowing the rapid spread of weeds, reducing vegetative fuel levels, and restoring fire-adapted ecosystems. The proposed treatment of would occur in 17 western states in the continental U.S. (including Washington, Oregon, California, and Idaho), and Alaska.

Vegetation would be managed using five primary vegetation treatment methods, which include manual and mechanical control, prescribed fire, biological, and chemical (i.e., herbicides) controls. Approximately half of the acres would be treated using prescribed fire, while the remaining areas would be treated using the other methods. A more complete description of the proposed action can be found in Chapter 2 of the Biological Assessment.

In 1976, Congress passed into law what is currently known as the Magnuson-Stevens Fishery Conservation and Management Act (MSA). This law authorized the U.S. to manage its fishery resources out to 200 miles off its coast. This 200-mile area is referred to as the exclusive economic zone. Regional Councils were established by Congress under the MSA, and were charged to prepare Fishery Management Plans (FMPs) for every fishery that required management. In 1996, the Sustainable Fisheries Act (Public Law 104-267) amended the MSA, requiring the identification of Essential Fish Habitat (EFH) for federally-managed fishery species, and the implementation of measures to conserve and enhance the habitat of these species, as described in federal FMPs. All federal agencies are required to consult with the NOAA Fisheries on all actions or proposed actions, permitted, funded, or undertaken by the agency that may adversely affect EFH. Adverse affects may include direct (e.g., contamination or physical disruption), indirect (e.g., loss of prey), site-specific, or habitat-wide impacts. The vegetation treatments proposed by the BLM have the potential to adversely affect EFH.

Congress defined EFH in the interim final rule (62FR 66551) as: "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity." For the purpose of interpreting the definition of EFH habitat, "waters" include aquatic areas and their associated physical, chemical, and biological properties; "substrate" includes sediment underlying the waters; "necessary" means the habitat required to support a sustainable fishery and the managed species contribution to a healthy ecosystem; and "spawning, breeding, feeding, or growth to maturity" covers all habitat types utilized by a species throughout its life cycle.

There are four components of an EFH consultation:

1. **Notification** – the federal agency (i.e., BLM) provides notification of an activity that "may adversely affect" EFH to NOAA Fisheries.

2. **EFH Assessment** – The federal agency provides a description of the proposed action, an analysis, and effects determination to NOAA Fisheries.

3. **Conservation Recommendations** – As dictated under section 305(b)(4) of the MSA, NOAA Fisheries provides EFH conservation and enhancement recommendations to the federal agency for actions that may adversely affect EFH. In turn, NOAA Fisheries discusses EFH conservation recommendations with the Federal agency and provides these recommendations to the federal agency, pursuant to section 305(b)(4)(A) of the MSA.

4. **Federal Agency Response** – the federal agency provides written responses to NOAA Fisheries and the appropriate Council within 30 days of receiving the conservation recommendations.

The objective of this EFH assessment is to describe potential adverse effects of the proposed BLM vegetation treatments in designated EFH for the federally-managed Pacific Coast (including Washington, Oregon, California, and Idaho) and Alaskan salmon fisheries. Five salmon species will be reviewed in this assessment: chinook and coho for the Pacific Coast; and chinook, coho, chum, pink, and sockeye for Alaska. This assessment will also describe proposed conservation measures for avoiding, minimizing, or otherwise offsetting the potential adverse effects to EFH resulting from the BLM's proposed vegetation treatment program.

Species and Regions Involved in this EFH Assessment

In a letter (dated September 13, 2000) addressed to the BLM, the Northwest Region of NOAA Fisheries stated that the BLM's existing environmental review procedures for federal actions meet the requirements for EFH consultation. The Northwest Region administers NOAA Fisheries programs for coastal habitats of Washington and Oregon, as well as the inland watershed habitats of Pacific salmon and steelhead in Washington, Oregon, Idaho, and Montana. The Southwest Region administers NOAA Fisheries programs for coastal habitats of California and islands in the Pacific Ocean, as well as the inland watershed habitats of Pacific salmon and steelhead in California. The Alaskan Region administers NOAA Fisheries programs for coastal habitats of Alaska, the Bering Sea, and the Aleutian Islands, and inland watershed habitats of Pacific salmon and steelhead in Alaska.

For the Pacific Coast (excluding Alaska), the Pacific Fishery Management Council (Pacific Council) manages federal fisheries for Washington, Oregon, Idaho, and California under three FMPs. These FMPs are the Pacific Coast Groundfish Management Plan (82 species), the Coastal Pelagic Species Fishery Management Plan (five species), and the Pacific Coast Salmon Management Plan (three species: chinook, coho, and Puget Sound pink salmon).

For Alaska, the North Pacific Fishery Management Council (Alaskan Council) manages federal fisheries for the Bering Sea/Aleutian Islands area and the Gulf of Alaska, under five FMPs. These FMPs are the Bering Sea/Aleutian Islands King and Tanner Crab Management Plan (eight species), the Alaska Scallop Management Plan (one species), the Groundfish of the Gulf of Alaska Management Plan (18 species), the Bering Sea/Aleutian Islands Groundfish Management Plan (17 species), and the Alaskan Salmon Management Plan (five species - chinook, coho, sockeye, chum, and pink salmon). The primary responsibility of the Alaskan Council is groundfish management in the Gulf of Alaska, Bering Sea, and Aleutian Islands. Although the Alaskan Council oversees the salmon fishery, the State of Alaska is the primary agency responsible for managing the harvesting, escapement numbers (salmon returning), and quota allocation aspect of Alaska's salmon fishery (North Pacific Fisheries Management Council 2002). This EFH assessment will only evaluate Pacific Coast salmon within both regions.

Essential fish habitat for the Pacific Coast salmon fishery refers to those waters and substrates that are necessary for salmon production that is capable of supporting a long-term, sustainable salmon fishery and salmon contributions to a healthy ecosystem. To achieve this level of production, EFH includes all streams, lakes, ponds, wetlands, and other viable water bodies that are accessible to salmon, as well as most of the habitat that was historically accessible (excluding areas upstream of longstanding naturally impassable barriers), in Washington, Oregon, Idaho, and California. In estuarine and marine areas, salmon EFH extends from the nearshore and tidal submerged environments within state territorial waters, out to the full extent of the exclusive economic zone offshore of Washington, Oregon, and California, north of Point Conception (Pacific Fishery Management Council 1999). The description of EFH for the Alaskan salmon fishery is consistent with that of the Pacific Coast, focusing on both the freshwater and marine habitats within the state (North Pacific Fishery Management Council 1998).

Based on completed EFH consultations at the project level for similar activities, the BLM does not anticipate that the proposed action would adversely affect EFH for shellfish, crustaceans, groundfish, and/or pelagic species (i.e., marine stocks) of EFH-identified species. The proposed vegetation treatments would occur inland, and would not impact any nearshore or marine habitats. Although the groundfish EFH includes the upriver extent of saltwater intrusion in river mouths along the Pacific Coast, no public lands are located within these regions. All species of the Alaskan salmon fishery will be included in this EFH assessment. However, only the chinook and coho salmon will be included for the Pacific Coast. The Puget Sound pink salmon population has not been included, as there are no public lands located within the greater Puget Sound area, where Puget Sound pink salmon reside. Pink salmon spawn closer to tidewater than other species of Pacific salmon, generally within 31 miles of the river mouth (Heard 1991). Therefore, no impacts to spawning, rearing, and/or migrating habitats of Puget Sound pink salmon are expected as a result of the proposed vegetation treatments.

Endangered Species Act (ESA)-listed Species and Their Relationship to EFH

The scope and requirements of EFH and ESA consultations differ from one another in that an EFH consultation is required for non-listed, federally-managed fishery species, while an ESA consultation only addresses fishery species within the action area that are federally listed or proposed for listing. Species listed under the ESA within the Pacific Coast region include chinook salmon from Washington and Oregon coastal sub-basins, as well as several populations in the middle and upper Columbia River basins, and the Clearwater River basin, and coho salmon populations from the Columbia River and Washington Coast (Weitkamp et al. 1995; Myers et al. 1998).

Each federally-listed salmon species is broken into distinct groups, or Evolutionary Significant Units (ESUs). To be considered an ESU, a population or group of populations must (a) be substantially reproductively isolated from other populations, and (b) contribute substantially to the ecological or genetic diversity of the species (Myers et al. 1998). A total of 11 ESUs for coho and chinook salmon have been listed as either threatened or endangered under the ESA in Washington, Oregon, Idaho, and California (Table A-1). Partial overlap exists between the EFH and ESA-listed coho and chinook species/critical habitat in the project area, along the Pacific Coast (i.e., affected species may be listed but not managed, or managed but not listed). Potential impacts from the BLM treatment program to ESA-listed species are identified in the BA. Conservation measures identified in the EFH assessment pertain to both ESA listed and non-listed species in EFH areas.

Species and Life History Stages Affected

The natural ranges of the Pacific salmon species addressed within this EFH assessment include large portions of the Pacific Rim of North America and Asia. Anadromous salmonids exhibit a significant shift in habitats where adults migrate from the ocean to their natal streams to spawn (Groot and Margolis 1991). However, all anadromous salmonids follow the same general life history pattern, which includes incubation and hatching of embryos, and emergence and initial rearing of fry (a life stage of salmon between absorption of the yolk sac and juvenile salmonid) in freshwater; migration to oceanic habitats for extended periods of feeding and growth; and return to natal waters for completion of maturation, spawning, and death within a few weeks after spawning. Although all anadromous salmonids share the same general life cycle, there are substantial differences among species in the amount of time spent in freshwater and marine environments, as well as in the types of habitat they utilize for spawning and rearing (Table A-2).

Pink and chum salmon typically spawn in gravel beds along coastal streams, within close proximity to tidewaters. These salmonids have the shortest freshwater phases of all anadromous salmon, entering the ocean within a period of days after emerging from the gravel (Salo 1991; Heard 1991; Hard et al. 1996; Spence et al. 1996). Pink salmon are mature at 2 years of age, at which time they return to freshwater to spawn (Heard 1991), while chum are more variable, spending between 2 to 5 years in the ocean before returning to their natal area to spawn (Salo 1991).

Coho salmon generally spawn in small, low-gradient streams in both coastal and interior systems (Laufle et al. 1986; Sandercock 1991). Juveniles typically spend between 1 and 3 years in freshwater. However, in the southern portion of their range (including Washington, Oregon and California) most fish migrate to sea after just 1 year (Spence et al. 1996). Adults return after approximately 18 months at sea to spawn in natal streams (Sandercock 1991).

TABLE A-1
Endangered Species Act Status of Pacific Coast Coho and Chinook Salmon within Action Area

Common Name	Scientific Name	ESU Status[1]
Coho salmon	*Oncorhynchus kisutch*	Central California (E – 10/96)
		Southern Oregon/Northern California coasts (T- 5/97)
		Oregon Coast (T – 8/98)
Chinook salmon	*Oncorhynchus tshawytscha*	Sacramento River winter-run (E – 1/94)
		Snake River fall-run (T – 4/92)
		Snake River spring/summer run (T – 4/92)
		Lower Columbia River (T – 3/99)
		Upper Willamette River (T – 3/99)
		Upper Columbia River spring run (E – 3/99)
		Central Valley spring run (T – 9/99)
		California Coast (T – 9/99)

[1] E = Endangered; and T = Threatened; date given is month and year of listing.
Source: NOAA Fisheries (2002).

Chinook salmon generally spawn in various-sized rivers, from small streams to large systems such as the Columbia River (Healy 1991). Chinook salmon display two dominant life history types: ocean- and stream-types (Myers et al. 1998). Individuals exhibiting an ocean-type life history usually spend only a few months in freshwater before migrating to the ocean, whereas stream-type chinook may spend 1 to 2 years in freshwater before their migration to the sea (Healey 1991; Myers et al 1998). Both ocean- and stream-type fish can reside in the ocean between 2 and 5 years before returning to spawn (Healey 1991).

Sockeye salmon most often spawn in the inlet and outlet streams of lakes (Burgner 1991, Gustafson et al. 1997). Shortly after emergence, sockeye fry migrate into these lakes, where they reside for 1 to 3 years before migrating to the ocean. They then spend 2 to 3 years in the ocean before migrating back to their natal spawning areas (Burgner 1991). Most sockeye are known as lake-type sockeye. However, some populations of sockeye salmon spawn in rivers without the lake rearing period, and are known as either river- or sea-type sockeye. Juvenile sockeye salmon that are river-type rear in freshwater streams for 1 to 2 years before migrating to the ocean (Gustafson et al. 1997). Sea-type sockeye salmon migrate to the ocean as underyearlings after spending only a few months in their natal river, and therefore rear primarily in saltwater (Gustafson et al. 1997).

Effects of the Proposed Action

Because of the general similarities among species of Pacific salmon with regard to life history stages and habitat requirements, this section will discuss effects of vegetation treatments on these fish as a group, rather than on a species-by-species basis. The treatments, as proposed, would be administered exclusively in inland aquatic habitats, with no activities occurring either in estuarine or marine environments. Therefore, only freshwater life history stages would potentially be affected by the proposed treatments. These freshwater stages include adult migration to natal spawning areas, incubation and maturation of eggs, and rearing and migrating of juveniles to the ocean.

The proposed treatments would follow the general timing restrictions established by NOAA Fisheries. These restrictions are imposed by both the states and NOAA Fisheries, for specific bodies of water, watersheds, or geographic regions, as a means of protecting salmonid species from potential habitat disturbance during spawning. Typically, activities may occur around or within streams containing salmonids during the summer months (i.e., May through October); however, timing windows may vary depending on geographic location.

TABLE A-2
Generalized Biological and Habitat Requirements in Pacific Salmonids

Attribute	Salmon Species				
	Chinook	**Coho**	**Pink**	**Chum**	**Sockeye**
Spawning sites	Mainstem	Tributaries	Mainstem, tributaries, and intertidal	Mainstem, tributaries, and intertidal	Lakeshore and tributaries
Time in grave (eggs)	Fall: 90-150 days Spring: 90-150 days	80-150 days	90-150 days (odd years only)	90-150 days	90-150 days
Emergence	March-April	April-May	Late January; April-May	Late February; April-May	April-May
Rearing sites	Mainstem	Mainstem, side channels, and slack water	Saltwater	Saltwater	Lakes
Time in freshwater	Fall: 60-120 days Spring: 1-2 yrs April, July, May	1-2 yrs May – June (12-14 months)	Several days	Several days	1-3 years
Time in marine habitats	2-6 years	1-2 years	2 years	2-3 years	1-4 years
Return to freshwater	Spring: April Summer: July Fall: Nov	Late fall	Early fall	Early to late fall	Mid-summer
Sources: Laufle et al. (1986); Burgner (1991); Healy (1991); Heard (1991); Meehan and Bjornn (1991); Sandercock (1991); and Salo (1991).					

Vegetation treatments, which are a critical component of restoring and maintaining the health of the land, have been conducted by the BLM since the agency's inception in 1946. In order to meet the objectives of local public land use plans, the aim of this vegetation treatment program is to increase soil stability, improve the quality and sustained yield of water, reduce the spread of noxious weeds, control vegetative fuels that cause wildfires, and increase desirable plant species coverage to benefit fish and wildlife.

All five salmon identified in this assessment occur on public lands and therefore could all potentially be affected by the proposed vegetation treatments. Two important habitat features that could be impacted are water quality and quantity. Pacific salmonids require cool, clean water that is of sufficient depth and velocity to allow passage, migration, and spawning, where floods do not scour channels (Spence et al. 1996).

Habitat Requirements of Salmonids in Streams

Adult Pacific salmon typically migrate upstream at temperatures between 37 °F and 68 °F in with water depth between 7 and 9.5 inches (Bjornn and Reiser 1991). Salmon may spawn within this temperature range, although spawning typically occurs between 39 °F and 52 °F (Bell 1986 *cited in* Spence et al. 1996). Once spawning is complete, water temperature affects the timing of salmonid egg incubation (Iwamoto et al. 1978; Laufle et al. 1986; Sandercock 1991; Healey 1991; Spence et al. 1996, Myers et al. 1998). For example, the time to 50% hatch (i.e., the time it takes 50% of the larval salmonids to hatch) for Pacific salmon species ranges from 115 to 150 days at 39 °F and from 35 to 60 days at 54°F (Bjornn and Reiser 1991). However, the alevin stage (a larval salmonid that has hatched, but has not yet fully absorbed its yolk sac) is generally less temperature-sensitive than the embryonic stages (Spence et al. 1996). Fry and parr (juvenile salmonids) are variable with regard to their temperature requirements, although as parrs most species are at risk when water temperatures exceed 77 °F. Although juvenile salmonids may briefly tolerate such high temperatures, high water temperatures are potentially lethal.

Higher water temperatures also contribute to the reduction of dissolved oxygen (DO) concentrations. Embryos and alevins are very susceptible to low DO levels, generally requiring levels above 8 parts per million to survive (Phillips and Campbell 1961). Low DO concentrations lead to an increased incidence of morphological abnormalities in emerging alevins (Bjornn and Reiser 1991). However, upon hatching, alevins in the gravel are able to detect oxygen gradients and move to areas with more suitable DO levels. Salmon, when rearing in freshwater, also require a high level (6.5-7.0 parts per million) of DO. They may survive when DO concentrations are lower (<5 parts per million), but growth, food conversion efficiency, and swimming performance may be adversely affected.

Water temperatures can be altered by several factors, such as removal of vegetative cover over the stream, withdrawal and return of water for agricultural irrigation, or release of water from deep reservoirs. Riparian vegetation, which is vegetation growing on or near the banks of a stream, provides shade, covers salmon from predation, moderates the water temperature of a stream, stabilizes banks, and controls soil erosion and sedimentation. Furthermore, riparian vegetation provides nutrients to the stream, food for juvenile salmon, and may contribute large woody debris (LWD), which in turn increases channel complexity, creates backwater habitats, and increases the water depth of pools. Studies have shown a correlation between the amount of LWD and salmon production (Dolloff 1983, House and Boehne 1986). For example, coho production declined when LWD was removed from streams in southeast Alaska (Dolloff 1983). Not only can riparian vegetation and water temperature of a stream influence the quantity and quality of salmonid habitat, but velocity of the streamflow and substrate of the stream can also play a significant role.

Adult salmonids can successfully migrate any stream reach of reasonable length if the water depth is greater than 4.7 inches when substrate particles average larger than 3 inches in diameter, or if the depth is greater than 3.5 inches when particles are less than 3 inches (Bjornn and Reiser 1991). Adult salmonids, upon reaching spawning beds, will typically deposit eggs within a range of water depths and velocities that minimize the risk of desiccation over the coming incubation period. These depths and velocities vary depending on species and run of population (i.e., spring, summer, or fall runs). However, studies suggest a depth of 7 inches and velocity of 0.98 feet per second (ft/s) meet the minimum criteria (Thompson 1972, Neilson and Banford 1983, Bjornn and Reiser 1991, Healy 1991, Heard 1991).

Upon emerging from the substrate, fry between 0.7 and 1.4 inches long require water velocities of less than 0.32 ft/s, whereas juvenile salmon between 1.6 and 7 inches long usually occupy sites with velocities of up to 1.3 ft/s (Bjornn and Reiser 1991). When rearing in freshwater, juvenile salmon seek out low velocity areas adjacent to faster water for feeding, resting, and growing. Overall, velocities required and used by juvenile salmonids vary with the size of the fish, and may change seasonally. By occupying slow velocity areas, salmon are likely to use less energy. Invertebrate drift abundance increases with velocity across a stream. Therefore, darting into the stream to feed, and then resuming their position in slower waters may provide a potential energy benefit for fish. Salmon use less energy maintaining their position in low velocities, while at the same time benefiting from the increased food abundance provided by higher velocities.

Within the stream channel, salmon require sufficient clean and appropriately-sized cobbles and gravel (ranging from 0.5 to 4 inches) for spawning and incubation (Spence et al. 1996). Furthermore, riffles, rapids (a section of stream with considerable surface agitation, swift current, and drops up to 3 feet), pools, and floodplain connectivity with the stream, are important for production, rearing, cover, and aeration.

Increases in streamflow can lead to alterations in channel morphology. Doubling the speed of streamflow increases its erosive power by four times and its bedload and sediment carrying power by 64 times (USDA Forest Service 2002). Accelerated runoff can thus cause unstable stream channels to downcut or erode laterally, accelerating erosion and sediment production. Lateral erosion results in progressively wider and shallower stream channels. Pool/riffle (riffles are defined as shallow sections of the stream with rapid current and a surface broken by gravel, rubble, or boulders) and width/depth ratios, which are important habitat components for salmonids, may also be altered.

Turbidity and sedimentation may negatively affect the abundance of food, impact juvenile salmon behavior, adult spawning, and egg incubation habitats (Iwamoto et al. 1978; Laufle et al. 1986; Healey 1991; Sandercock 1991; Spence et al. 1996). An increase in turbidity can cause an increase in phytoplankton, inorganic, and organic materials that are suspended in the water column during high flow conditions, potentially diminishing light penetration into the stream (Spence et al. 1996). Diminished light levels can reduce algal productivity and change the instream plant composition (Samsel 1973). This reduction of plant material instream may allow sediment to drift within the water column, increasing siltation. Siltation contributes significantly to the reduction in diversity of aquatic insects and other aquatic invertebrates (Spence et al. 1996). Silt reduces the interstices (narrow spaces) in the substrate, thereby limiting the microhabitat for benthic invertebrates (i.e., a portion of the juvenile salmon diet) in a stream. For example, feeding and territorial behaviors of juvenile coho salmon are disrupted by short-term exposure (approximately 2-5 days) to turbid water (Berg and Northcote 1985).

Vegetation Treatment Effects on EFH

Salmon respond to a variety (or a combination) of environmental factors, either in their behavior or physiology. Low stream flows, high water temperatures, and excessive turbidities may impede access of adult salmon during migration, spawning, or negatively impact egg incubation. Requirements of salmon and their use of habitat vary seasonally. Therefore, in order to utilize the full extent of the resources offered within a stream, river, or watershed, salmon require unobstructed access throughout their habitat.

The proposed vegetation treatment activities could either directly or indirectly affect the physical characteristics required by salmon species within Alaska and Pacific Coast regions. Over the short term, there could be impacts to salmon habitat. However, over the long term, the proposed treatments should improve the overall ecosystem health of public lands, including aquatic habitats.

Prescribed Fire Treatments

Over the long term, a well-managed prescribed fire could have a beneficial effect on salmonids, as a result of a more healthy and functioning ecosystem, improved and rejuvenated habitat, as well as increased productivity (Minshall and Brock 1991, Burton 2000). These benefits would especially be true for riparian habitats that were historically subject to frequent, low intensity burns. Both the condition of the site prior to burning and the intensity of the burn would influence whether the end result of the fire was beneficial. Even a high intensity burn could eventually have a beneficial effect on riparian and aquatic habitats, especially if site restoration measures were followed post-burn.

A well-planned and managed prescribed burn would reduce the risks of a future, high-intensity wildfire in riparian habitats. Because the BLM would follow guidance provided by the National Fire Plan, high intensity fires would not be ignited in sensitive habitats, and many of the adverse effects listed in this section would therefore be minimized. The proper fire management plan would involve vegetative fuels reduction and other measures designed to reduce the intensity of a prescribed fire in areas of high wildfire risk. Removal of fuel sources through burning could reduce the future risks of high-intensity wildfires in ecosystems with altered disturbance regimes. A naturally occurring (or human-caused) fire in an area with fuel buildup, where fires have been suppressed for many years, would be expected to burn hotter, and over a larger area than a controlled fire.

In general, the intensity of the fire would determine the extent and severity of effects to fish species. Small fires, like those that historically occurred in many riparian habitats, would be expected to have minimal effects, and could help maintain habitat quality. Over the short term, negative effects from prescribed fire would be possible. Depending on its size and intensity, a prescribed burn in a riparian area or an adjacent upland area could rapidly increase the water temperature, potentially harming or causing mortality to aquatic species with strict temperature requirements. Such a burn could also cause temporary chemical changes to aquatic habitats, through the release of ash directly into these systems. Ash created by wildfires or prescribed burning has been documented to have life-threatening effects on some species of fish (Agyagos et al. 2001). While the introduction of ash into an aquatic habitat may be directly life threatening to salmonids, the indirect effects are uncertain.

Fires are capable of consuming a large amount of vegetation and exposing a large area of bare soil that would likely result in a pulse of nutrients into the aquatic system. A number of nutrients (such as nitrogen, phosphorus, and sulfur) entering a stream after fire appear to be below the tolerance threshold for aquatic organisms, and dissipate rapidly with stream dilution and flushing (Swanston 1991). This rapid increase of nutrients into an aquatic system could also temporarily benefit many salmonids by increasing their food production.

Prescribed burning in a riparian area or adjacent upland habitats could directly impact streams over the short term (i.e., days, weeks, or months) by causing increased delivery of sediment to channels, as well as increased channel flow, LWD, and nutrient levels in the stream (Swanson 1980). Fires alter the physical properties of the surface layers of soil, increasing both the total water yield and storm-flow of a watershed (Swanston 1991). This increase in sedimentation could lead to a reduction in spawning habitat, destroy eggs, and displace alevins already in the stream channel (Swanston 1991).

Snags and other LWD that fall into an aquatic habitat as a result of fire could benefit salmonids, as they provide the principal structural features that shape the stream’s morphology, linkages to the floodplain, habitat complexity, streambed materials and other characteristics (Salo and Cundy 1987, Meehan 1991). The addition of LWD after a fire also improves habitat diversity for juvenile salmonids by providing cover and additional rearing areas (Meehan 1991).

Activities associated with prescribed fire, such as creating wet lines and extinguishing hot spots after the majority of the fire has gone out, require the availability of a nearby water source. Water may be needed to fill portable pumps, pumps mounted to fire engines or water tenders, or 100- to 250-gallon buckets suspended by helicopters. Use of water from aquatic habitats that support salmonids could have adverse effects on those habitats, particularly in arid climates or during dry seasons, when limited water is available. If firelines were allowed to tie into aquatic habitats that support salmonids, additional effects would be possible through a reduction in water levels.

A foam line could also be used as a firebreak near an aquatic system to control fires, and aqueous firefighting foam could potentially leach into the water. Other chemicals that could be released or leach into aquatic habitats include ignition fuels, or fuels (e.g., gasoline) used to power equipment (e.g., helicopters, vehicles, and mechanical equipment), which would further degrade the water quality.

Another adverse effect to aquatic habitats could result from the construction of roads to gain access to treatment sites. New roads affect streams by accelerating erosion and sediment loading into the aquatic habitat, by altering the channel morphology, and by changing the runoff characteristics of the watershed (Furniss et al. 1991). While creating access to a site to treat fires, new roads create a potential for increased human disturbance in the future.

Mechanical Treatments

Few direct effects to salmonids as a result of mechanical treatment methods would be likely. The majority of effects would occur indirectly, thorough the alteration of salmonid habitat.

Apart from the removal of noxious weed species, mechanical treatment methods in riparian areas could have a long-term beneficial effect on aquatic habitats by reducing woody overgrowth. The removal of excess woody vegetation, which would not typically be present under historical fire regimes, could return riparian habitats to much healthier states. In addition, removal of this excessive woody vegetation would likely reduce the risk that a future stand-replacing or catastrophic fire would burn through riparian areas. It is for this reason that mechanical treatments are often used prior to prescribed burns to reduce fuels. With adequate buffers to ensure bank stability and LWD recruitment, and measures to reduce sedimentation into streams (see Conservation Measures section), mechanical treatments could help restore riparian areas to their historical states, without damaging aquatic habitats over the short term.

Some treatment activities in riparian areas could remove trees, shrubs, and other materials that would eventually become LWD, an important habitat element for salmonids. These effects to salmonid habitat would be greatest if woody vegetation within the distance of one tree height away from the channel were removed (Spence et al. 1996). Further from the water, the probability that a falling tree will enter the stream channel is much reduced, and the indirect effects of future LWD removal on aquatic habitats would be less significant.

Mechanical treatments that uproot plants would decrease slope stability in riparian areas. The root strength of plants in riparian areas, particularly trees and shrubs, contributes to slope stability and retards erosion. Internal changes in soil structure would take place after vegetation was removed, sediment filled soil pores, and compaction occurred (Chamberlin et al. 1991). Soil disturbance could also speed up water movement, resulting in increased peak flows in a stream. In addition, water flow over the ground surface would be more likely, which could accelerate erosion. Significant impacts would be most likely if woody vegetation on slopes directly adjacent to aquatic habitats were removed. Further from the water, the contribution of root strength to maintaining streambank integrity declines and effects would be proportionally less severe (National Fire Plan Technical Team 2002).

A number of mechanical treatments would disturb the soil during vegetation removal (e.g., tilling or skidding with tractors), increasing the potential for sediment transport into the stream. The closer these activities occurred to the aquatic habitat, the greater their potential to affect salmonids therein. Soil disturbance could also increase the likelihood that weeds would recolonize a site (Sheley et al. 1995). Therefore, reseeding or other forms of site restoration would be crucial to realize a long-term benefit as a result of mechanical treatment methods.

Fuel used to power equipment could potentially leak directly into the water, causing a decrease in water quality. In addition, the use of heavy equipment in riparian areas could lead to streambank collapse, increasing instream sedimentation, and covering possible salmonid spawning grounds. If vehicles were allowed directly into aquatic habitats, additional effects such as increased instream sedimentation, altered channel morphology, and increased potential for chemical contamination of the aquatic system, would likely.

Manual Treatments

Manual treatments methods would be expected to have few effects (either direct or indirect) on fish or their habitats, unless excessive amounts of riparian vegetation were removed. Trampling by workers and disturbance of soil from the removal of vegetation could result in some erosion and sedimentation into aquatic habitats, which would be localized rather than widespread. These treatment methods are likely to involve the removal of the smallest amount of riparian vegetation, with relatively minor effects to salmonid habitat caused by vegetation removal (as discussed above).

Biological Treatments

Livestock grazing in the western U.S., particularly in rangelands, has played a significant role in the degradation of riparian areas for over a hundred years (Heady and Child 1994). As a result, anadromous fish habitats have been degraded, particularly in arid rangelands (Waters 1995). The extent to which grazing by domestic animals affects fisheries is not completely understood, leading to controversy among scientists (Platts 1991).

Historically, riparian areas have been grazed more heavily than upland zones because they have flatter terrain, a water source, and more succulent vegetation (Armour 1977, Platts and Nelson 1985). The amount that grazing treatments would affect riparian habitats would vary depending on the type of animals (i.e., sheep, goats, or cattle), the size of the herd, and the intensity and duration of grazing. In more intensive grazing scenarios, mass erosion from trampling, hoof slide, and streambank collapse could cause soils to move directly into the stream (Platts 1991, Heady and Child 1994). Undercut banks, which often provide shelter to salmonids, could be damaged or collapse in grazed areas, thus decreasing the amount of available salmonid habitat (Platts 1991). In addition, heavy trampling could cause soil compaction, which would reduce the infiltration of overbank flows and precipitation into riparian soils (Johnson 1992). Soil compaction could also hasten surface runoff, resulting in a more rapid hydrologic response of streams to rainfall (Spence et al. 1996). The increase in instream hydrology during rainfall could result in increased channelized erosion of a stream (Kauffman et al. 1983).

Improper use of domestic animals to control weeds in riparian and adjacent upland areas could degrade the production of salmonids (Chapman and Knudsen 1980, Platts 1991). These effects would be heightened if animals were allowed to wallow and wade directly in the aquatic habitat. Such wading would likely cause direct mortality, primarily of eggs and pre-emergent fry, but also of adults and smaller fishes. Platts (1981) found fish densities were 10.9 times greater in lightly or ungrazed areas than in highly grazed sections. Chapman and Knudsen (1980) also found livestock-altered stream reaches contained less fish biomass.

Apart from the removal of vegetation, the disturbance to the soil caused by the movement of domestic animals over riparian habitats could induce increased sedimentation. Grazing could also widen stream channels, promote incised channels, lower water tables, reduce pool frequency, and alter water quality (Platts 1991). In addition, the input of feces into aquatic habitats could also degrade water quality.

Some grazing strategies have been developed that increase forage production and plant and litter cover of streams, and decrease soil erosion, all of which would benefit fish. Strategies that appear to be the most successful for fisheries are rest rotation with seasonal preference (for smaller domesticated animals such as sheep or goats) and corridor fencing (for larger domesticated animals such as cattle; Platts 1991). Under the seasonal rest rotation strategy for goats or sheep, riparian habitats are grazed at selected times of least impact, and the domestic animals are moved into different pastures to meet seasonal requirements. This seasonal movement of domestic animals would allow plants and streambanks to recover from past damage. Fencing a portion or the entire riparian corridor, although very costly, would eliminate domestic animals from the riparian areas, and allow riparian habitats to be completely rehabilitated. Literature also suggests that use of cattle to contain weeds in riparian areas would be more detrimental to these habitats than would the use of sheep (Platts 1991).

Herbicide Treatments

A wide variety of herbicides are used to control invading vegetation in order to enhance the suitability of an area for re-establishment of desired vegetative species. Most of the literature addressing the toxicity of herbicides to salmon comes from the laboratory rather than the field. Therefore, specific impacts to the various life history stages of salmon in nature, caused by the active ingredients in herbicides, are not well understood. However, it is assumed that any release of herbicides into aquatic habitats that support Pacific salmon could result in some direct impacts to those species.

Salmonids could potentially come into contact with herbicides if sprayed formulations were to enter aquatic habitats during the application process, either through direct spray of the water by herbicides approved for use in aquatic habitats (i.e., diquat, fluridone, and certain formulations of 2,4-D, glyphosate, imazapyr, and triclopyr), accidental spray of the water by terrestrial herbicides, or off-site drift or surface runoff of herbicides sprayed in nearby upland habitats into aquatic habitats.

Of the herbicides proposed for use, the following herbicides would potentially result in adverse health effects to salmonids if sprayed directly into aquatic habitats: 2,4-D, bromacil, diquat, diuron, fluridone, glyphosate, metsulfuron methyl, picloram, tebuthiuron, and triclopyr. Since 2,4-D, diquat, fluridone, glyphosate, and imazapyr are all either strictly aquatic herbicides or are approved for use in aquatic or riparian habitats, direct spray into an aquatic habitat would be a normal treatment application for these herbicides.

However, an analysis of the direct impacts of herbicides on salmonids should relate the site-specific exposure conditions (i.e., expected environmental concentration, bioavailability, and exposure duration) to the known or suspected impacts of the chemical on the health of exposed fish. It appears that the proposed herbicide use is unlikely to cause fish kills when used according to the USEPA label. Therefore, for these salmonid species, the vast majority of harmful direct effects are expected to be sublethal exposure.

Bioaccumulation is most likely to occur when salmon are exposed to persistent chemicals that have low water solubility and high lipid solubility (Norris et al. 1991). Typically, herbicides used around streams would not meet these criteria, although salmon could take up some of the chemical, at a sublethal level. Sublethal effects of herbicides on salmonids could include reduced growth, decreased reproductive success, altered behavior, and reduced resistance to stress (Beschta et al. 1995).

Under certain conditions, the spraying of herbicides in riparian areas would be inaccurate and difficult to control, and chemicals could easily enter the aquatic habitat. The risk of toxicological effects to salmonids would be greatest if herbicides were directly applied to surface water or reached surface water by wind drift (Spence et al. 1996). Many of the herbicides used around riparian areas have a half-life (the period required for half the molecules of the substance to decompose) ranging from 2 to 5 weeks (Norris et al. 1991). However, there are several persistent herbicides (i.e., hexazinone, atrazine, imazapyr, and triclopyr) that have half-lives of 2 to 6 months. Of the terrestrial herbicides proposed for use addressed in BLM ERAs, only diuron would potentially result in adverse health effects to salmonid species as a result of off-site drift into nearby aquatic habitats. Based on ERAs, salmonids within 100 feet of a diuron application (at the maximum application rate) would be at risk. The Forest Service risk assessments did not consider off-site drift scenarios. Risks to salmonids from drift of these herbicides, with the exception of triclopyr BEE, seem unlikely, given the results of surface runoff scenarios. To be conservative, however, it is assumed that adverse effects to salmonids could potentially occur as a result of drift of glyphosate, picloram, and triclopyr BEE. Considering the proximity of application areas to salmonid habitat and the possible effects of herbicides on riparian and aquatic vegetation, appropriate buffer zones around salmonid-bearing streams would be maintained (Table A-3).

Herbicides used in vegetation treatments could indirectly affect salmonid species if surface runoff from a contaminated upland area entered a water body. Of the terrestrial herbicides proposed for use, bromacil, diuron, tebuthiuron, and triclopyr BEE could result in adverse health effects to salmonids under certain scenarios of surface runoff. Of these herbicides, diuron would likely pose the greatest risks to salmonids via this exposure pathway, potentially resulting in adverse health effects salmonids in areas where precipitation is greater than 10 inches per year.

The potential indirect effects to salmon from herbicide treatment would be both positive and negative. Herbicides could alter natural patterns of plant succession along streams by reducing and slowing the development of deciduous trees. Typically, conifers do not begin to dominate riparian communities until after 20 years or so. Coniferous vegetation differs greatly from deciduous vegetation in the timing of litter fall and the quality of organic matter produced (Norris et al. 1991). Coniferous wood in streams does not break down as quickly as that of deciduous species, therefore maintaining instream habitats for longer periods. A benefit of the slow recovery of vegetation after herbicide treatments is the opportunity for larger, slow growing conifers to establish in riparian and adjacent upland areas. Alternatively, invasive species could outcompete native species and dominate plant communities during succession, having a negative effect on riparian habitat.

Removal of vegetation within a riparian area would increase the amount of solar radiation reaching the stream. This increased solar radiation could stimulate autotrophic production (i.e., growth of plants and algae), potentially increasing the food base for invertebrates and fish (Spence et al. 1996). However, herbicide application within this area could slow the recovery of vegetation, allowing continued disruption to the hydrologic and sediment delivery processes that affect the nearby streams.

TABLE A-3
Buffer Distances to Minimize Risk to TEP Fish and Aquatic Organisms from Off-site Drift Risk to BLM-evaluated Herbicides from Broadcast and Aerial Treatments

Application Scenario	BROM[1]	CHLR	DICA	DIFLU	DIQT	DIUR	FLUR	IMAZ	OVER	SULF	TEBU
Streams Containing ESA-listed Salmonids											
Typical Application Rate											
Aerial	NA	0	NA	NA	NA	NA	NA	0	NA	0	NA
Low boom	0	0	0	0	NA	0	NA	0	0	0	0
High boom	0	0	0	0	NA	0	NA	0	0	0	0
Maximum Application Rate											
Aerial	NA	0	NA	NA	NA	NA	NA	0	NA	0	NA
Low boom	0	0	0	0	NA	100	NA	0	0	0	0
High boom	0	0	0	0	NA	100	NA	0	0	0	0
Typical Salmonid-bearing Streams											
Typical Application Rate											
Aerial	NA	0	NA	NA	NA	NA	NA	0	NA	0	NA
Low boom	0	0	0	0	NA	0	NA	0	0	0	0
High boom	0	0	0	0	NA	100	NA	0	0	0	0
Maximum Application Rate											
Aerial	NA	0	NA	NA	NA	NA	NA	0	NA	0	NA
Low boom	0	0	0	0	NA	100	NA	0	0	0	0
High boom	0	0	0	0	NA	900	NA	0	0	0	0

[1] BROM = Bromacil; CHLR = Chlorsulfuron; DICA = Dicamba; DIFLU = Diflufenzopyr; DIQT = Diquat; DIUR = Diuron; FLUR = Fluridone; IMAZ = Imazapic; OVER = Overdrive®; SULFM = Sulfometuron methyl; and TEBU = Tebuthiuron.
NA = Not applicable.
Boom height = The Tier I ground application model allows selection of a low (20 inches) or a high (50 inches) boom height.

By exposing more surface area of soil directly to rainfall, and increasing the overland flow of water into the aquatic habitat, removal of vegetation could result in decreased water storage capacity of the soil. Over the long-term, overland flow could erode the topsoil and cut rills and gullies or deepen existing gullies, thereby concentrating runoff (USDA Forest Service 2002). As a result, sediment production would be increased. Reduced infiltration and increased runoff could decrease recharge of the saturated zone and increase peak flow discharge. Thus, the amount of water retained in the watershed to sustain base flows would also be reduced.

The different methods of applying herbicides to an area would cause varying degrees and types of disturbance. Using fixed-wing aircraft or helicopters would not result in any soil disturbance to riparian areas. However, this method would likely result in a release of large amounts of chemicals directly into the water, thereby directly affecting salmonid species. Use of trucks or ATVs in riparian habitats or adjacent to aquatic habitats would cause some soil disturbance, increasing the risks of erosion and sedimentation. In addition, use of these motorized vehicles could result in leaks of fuel or other toxic substances into aquatic systems. There is also likely to be some chemical drift associated with these methods, which could impact salmonids in adjacent aquatic habitats. Application by backpack sprayer would result in the least disturbance to riparian areas, and would have minimal effects on salmonid species. There would be a negligible amount of soil disturbance associated with this method, and applications would likely be accurate, allowing the applicator to avoid releasing chemicals into the water. Under this method, the least amount of riparian vegetation would likely be killed, resulting in overall limited erosion, sedimentation, and alteration of fish habitat.

Conservation Measures

The goal of this EFH assessment is to establish no net loss of freshwater habitat that is valuable to salmonids. For the purposes of developing conservation measures, riparian areas include traditional riparian corridors, wetlands, intermittent streams, and other areas that help maintain the integrity of aquatic ecosystems by (1) influencing the delivery of coarse sediment, organic matter, and woody debris to streams, (2) providing root strength for channel stability, (3) shading the stream, and (4) protecting water quality.

Activities associated with the proposed vegetation treatments would have the potential to adversely affect salmonids and their habitat. Implementation of the measures listed below would minimize these potential impacts to a negligible level.

General Measures

- Establish riparian buffer strips adjacent to salmonid habitats to reduce direct impacts to the various life stages of these species. Buffers widths should depend on the specific ecological function for which protection is desired (e.g., streambanks stabilization, control of sediment inputs from surface erosion, or maintenance of shade to stream channels). Local BLM field offices would consult BLM and Forest Service ERAs prepared for the BA and PEIS to obtain programmatic guidance on appropriate buffer distances. Field offices can also input information on local site conditions (e.g., soil type, vegetation type, precipitation, treatment method) into interactive spreadsheets developed for the ERAs to develop more site-specific, and in most cases less restrictive, buffers for individual projects.
- Implement SOPs to minimize sedimentation and disturbance of riparian vegetation.
- To avoid erosion and future recreational uses within close vicinity of aquatic areas, limit or exclude construction of new permanent or temporary roads within the boundary of treatment riparian areas.
- Where possible, to avoid increased instream sedimentation, choose low-intensity burns and manual treatment methods over mechanical treatment methods and use of domestic animals.

Prescribed Burning Treatments

- Where feasible, avoid ignition of fires within buffer strips.

Mechanical Treatments

- Minimize the use of mechanical treatment methods (including timber harvest and timber salvage) within buffer strips.
- To avoid damaging potential spawning areas, do not use mechanical equipment in perennial channels, or in intermittent channels with water, except at crossings that already exist.
- Minimize log hauling during wet weather, and on non-paved roads.
- Minimize skidding or ground-based yarding within buffer strips.
- Do not remove large woody debris from buffer strips during mechanical treatment activities.
- Do not plowing within buffer strips.
- Minimize ground disturbing activities (disking, drilling, chaining, and plowing) within buffer strips.
- Where feasible, avoid mowing within 100 feet or 1 site-potential tree height (whichever is greater) from the stream channel, except where required as part of road maintenance.
- Do not remove excess vegetation or slash, and do not subsoil, less than one site-potential tree height (or 100 feet) from the active channel (whichever is greater).

Herbicide Treatments

- Where feasible, minimize spray operations around aquatic habitats to days when winds are > 10 miles per hour for ground applications, and > 6 miles per hours for aerial applications, to avoid wind drift or direct application of herbicides into these habitats.
- Where feasible, minimize the use of terrestrial herbicides (especially bromacil, diuron, and tebuthiuron) in watersheds with downgradient ponds and streams if potential impacts to salmonids are of concern.
- Time herbicide applications near salmonid-bearing streams so that they do not overlap with sensitive life-history stages of these fish (would vary at the local level).

Biological Treatments

- In watersheds that support salmonids or that flow into watersheds where salmonids occur, to minimize the cumulative effect of grazing in areas that have been burned, do not conduct weed control by domestic animals in burned areas until they have recovered enough to control ash and sediment produced by the treatment.

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