Habitat Status Reports

Habitat Descriptions: Content and Methodology

Content: The following are descriptions of the topics covered in each section of the enclosed habitat reports.

Description: This section describes 1) fundamental physical characteristics of the habitat types, including species composition, structure, cover, and topographic setting; and 2) large-scale patterns such as the evenness or contiguity of the habitat on the landscape and any less common subtypes typically associated with the habitat. Habitat pattern is considered very important not only for understanding ecosystem processes but also for providing the appropriate context for management considerations.

Distribution and Abundance on National Forest System and Adjacent Lands: This section describes the estimated area for each habitat type, elevation zones, and occurrence within southern California and elsewhere on National Forest System lands and other lands. This section should answer the question, "How important is USDA Forest Service management for the overall quality, quantity, and connectivity of the habitat type?"

Ecological Processes: The purpose of this section is to explain the general biological and physical factors that largely explain why the habitat is located where it is and what factors are at work to keep it there. This section describes the major physical processes that control the distribution of each habitat type, such as soils, climate, wildfire, and topography. These factors are related to the autecology of dominant plant species.

Factors that Influence Ecological Processes: This section highlights ecological factors that influence the stability, quality, and distribution of each habitat type. Topics vary with each habitat, depending on their importance and the amount of scientific understanding and available literature on each topic. Some of the subheadings in this section include fire regime, commercial timber harvest, land uses, recreation, livestock grazing, invasion of nonnative undesirable species, mining, water storage and diversion structures, and climate change. The goal of this section is to provide scientific background that explains past, present, and potential future directional trends in the distribution, abundance, quality, and species composition of each habitat type.

Management Considerations: This section briefly identifies management considerations for the habitat types, including conservation and preservation priorities, measures to conserve habitat during planning and implementation of forest actions, and considerations for habitat restoration. This section also provides summaries of the special-status species present within each habitat.

Methodology

Habitat Types: Based on the purpose and need of the habitat descriptions to "describe the ecological context" for the individual species viability assessments (USDA Forest Service 2001), the habitats in the "habitat group codes" of the *Southern California Mountains and Foothill Assessment* (Stephenson and Calcarone 1999) were reorganized into a more logical framework for addressing ecosystem management issues.

The 33 habitat group codes are really an individual species-driven grouping that could function in a matrix or table of species-habitat relationships. The grouping, however, does not follow a logical habitat organization or hierarchy (Beyer pers. comm.), nor are the categories mutually exclusive. For example, some are vegetation communities (e.g., grassland), some are topographic settings (e.g., low elevation valley floor), and others are species-specific behavioral descriptions (e.g., habitat generalist). The 33 habitat group codes are also not compatible with other standard habitat classification schemes such as those in the California Natural Diversity Database (2001), the California Wildlife Habitat Relationship System (Mayer and Laudenslayer 1988), the *Preliminary Descriptions of the Terrestrial Natural Communities of California* (Holland 1986), or *A Manual of California Vegetation* (Sawyer and Keeler-Wolf 1995).

The reorganization facilitates the discussion of ecosystem-level processes and patterns that provide the context for the individual species assessments. The goal of reorganization was simply to make the categories consistent, mutually exclusive, and have a standard organization recognized by most ecologists. In actuality, the reorganization is relatively minor. Most of the original categories based on vegetation communities were retained or aggregated (e.g., riparian, montane meadows). The reorganized habitats prepared for this report include all of the 33 habitat group codes as either individual habitat types or elements within habitats, but provides a consistent and standard organization based on major vegetation physiognomy. Table 1 provides a "crosswalk" from the 33 habitat group codes to the reorganized format, and table 2 provides a "reverse crosswalk" from the reorganized format to the original 33 habitat codes.

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Alpine and Subalpine Habitats

Description: Alpine and subalpine habitats occur primarily at elevations above 8,500 feet (2,800 meters) to the highest peaks at around 11,500 feet (3,500 meters). On cooler aspects in the Transverse and Peninsular Ranges, subalpine plants predominate as low as 7,900 feet (2,400 meters) (Barbour 1988, Stephenson and Calcarone 1999).

The primary plant community is subalpine conifer forest dominated by lodgepole pine (*Pinus contorta* ssp. *murrayana*), limber pine (*P. flexilis*), and white fir (*Abies concolor* ssp. *concolor*). These are typically relatively sparse forests characterized by slow-growing small-diameter trees, open canopies, canopy heights of 33-59 feet (10-15 meters), and little understory shrub or herb cover. A rare and fragile alpine cushion or alpine scrub plant community occurs above treeline (Barbour 1988, Stephenson and Calcarone 1999.

Lodgepole pine typically dominates the lower parts of subalpine forests above 7,900 feet (2,400 meters). On mesic sites lodgepole pines grow to 59-69 feet (18-21 meters) and form relatively dense stands. The understory is species poor and has little vegetative cover. Lodgepole pine forest often occurs in areas of maximum snow depth, and canopy shade and snow duration inhibit understory growth. Common understory shrubs in canopy openings include whitethorn (*Ceanothus cordulatus*), greenleaf manzanita (*Arctostaphylos patula* ssp. *platyphylla*), and bush chinquapin (*Chrysolepis sempervirens*). (Barbour 1988.)

On higher peaks, lodgepole pine forms stands of krummholz, which consists of individual trees deformed into widely spaced, dense, low-growing, multi-stemmed prostrate mats. Krummholz is an environmental, rather than genetic, response to the harsh growing conditions near treeline. The height of krummholz clumps roughly corresponds to winter snow depth, as branches and needles exposed to extreme winter weather are vulnerable to winterkill (desiccation from strong winds and cold temperatures) and mechanical damage from blowing snow and ice. (Major and Taylor 1988, Thorne 1988

Limber pine mixes with lodgepole throughout the subalpine zone but is the dominant tree species at higher elevations and up to treeline, as well as in drier sites and rocky soils. Limber pine occurs with Jeffrey pine (*Pinus jeffreyi*) on Mount Pinos. Squaw currant (*Ribes cereum*), sagebrush (*Artemisia tridentata*), snowberry (*Symphoricarpos* sp.), and squirreltail grass (*Elymus elymoides*) are common understory associates. Above treeline, limber pine, like lodgepole pine, forms stands of krummholz. (Major and Taylor 1988, Thorne 1988

Montane chaparral scrub in the subalpine zone typically occurs on rock outcrops, ridges, and xeric slopes. This community consists of Sierra juniper (*Juniperus occidentalis* ssp. *australis*) and curl leaf mountain mahogany (*Cercocarpus ledifolius*), as well as shrub species typical of subalpine forests. Unlike low elevation chaparral communities, these habitats typically have open canopies. (Thorne 1988.)

Alpine and, to a lesser extent, subalpine habitats are naturally isolated and small in size because their occurrence is restricted to high mountain peaks. Because subalpine and alpine habitats are restricted to higher elevations they cover a very small portion of the southern California mountains. Some refer to alpine areas in Southern California as "subalpine barrens" because they are lower in elevation than the regional climatic treeline (Billings 1988). Analogous to island ecosystems, the large distances separating alpine peaks generally result in distinctive species composition on each peak. For example, the alpine flora of the San Bernardino and the San Jacinto Mountains are only about 40% similar (Barbour 1988).

These traits of isolated distribution and limited habitat, combined with the distance from other alpine habitats in the Sierra Nevada, result in the alpine flora of the southern California mountains being relatively species poor and fragmented, especially in the San Gabriel and San Jacinto Mountains (Barbour 1988). A few alpine plants in southern California are narrow endemics, while others have characteristically disjunct distribution patterns, also occurring on peaks in the Sierra Nevada, Great Basin, or Rocky Mountains (Major and Taylor 1988).

Distribution and Abundance on National Forest System and Adjacent Lands: Alpine and subalpine habitats on the four southern California National Forests occur only in the highest reaches of the San Gabriel, San Bernardino, and San Jacinto Mountains, and the summit of Mount Pinos. Alpine and subalpine habitats are most extensive on the high slopes of Mount San Gorgonio and Mount San Jacinto. (Stephenson and Calcarone 1999.)

Alpine and subalpine habitats total 15,604 acres (6,096 hectares), comprising 8,229 acres (3,330 hectares) of subalpine conifer forest; 2,504 acres (1,013 hectares) of mixed conifer-fir forest; 2,890 acres (1,170 hectares) of montane chaparral/scrub; 1,913 acres (774 hectares) of alpine scrub and barren areas; and 68 acres (28 hectares) of other habitats (Stephenson and Calcarone 1999).

Ecological Processes: Poorly developed, shallow, and rocky soils; short and variable growing seasons; annual and daily extreme temperatures; and high insolation characterize alpine and, to a lesser extent, subalpine environments; these factors control vegetation patterns and reproductive and growth processes (Major and Taylor 1988). Overall precipitation tends to be similar to levels in montane forest habitats, but average temperatures are significantly colder and a substantially greater percentage falls as snow (Barbour 1988).

Available heat, snow duration, soil moisture, and growing season length control subalpine tree establishment and the position of the upper treeline (Barbour 1988). Establishment of subalpine conifers tends to occur sporadically, may require seed caching by corvid birds, and tends to be greater in years of deep winter snow pack but less in years of prolonged snowpack duration. Once established, trees tend to be slow growing but long lived. Trees exhibit stress adaptations, such as high nutrient use efficiency and long needle longevity, to the poor soils and variable, short growing seasons (Barbour 1988).

The growth form, plant structures, and physiology of alpine cushion plants exhibit adaptation to the harsh climatic and edaphic conditions of alpine environments. Alpine plants tend to exhibit high nutrient use efficiency and quick, opportunistic growth and reproductive cycles to be able to flower and set seed in the short growing season. Cushion plants have low and dense growth forms and thick, leathery leaves to minimize water and heat loss and withstand daily temperature extremes, intense insolation, droughty soils, and persistent winds. Seedling establishment is limited by the availability of soil, extreme temperatures, and regular frost heaving (Billings 1988, Major and Taylor 1988).

Factors that Influence Ecological Processes: Alpine plants are vulnerable to trampling by hikers and other forms of ground disturbance (Billings 1988), but these impacts are limited to a small number of locations around developed recreation areas, roads, and trails (Stephenson and Calcarone 1999). Trampling and other ground disturbances resulting from hiking, rock climbing, camping, and road building have removed or degraded some areas of alpine and subalpine plants (Stephenson and Calcarone 1999). In general, however, alpine and subalpine ecosystems are considered to be largely intact, stable, and little disturbed with the exception of some heavy recreation use in the immediate vicinity of trails.

Alpine and, to a lesser extent, subalpine habitats are subjected to natural climatic disturbances on a regular basis. Prolonged snowpack, growing season frosts, extreme winter weather, and other factors can reduce growth and minimize or prevent reproduction (Barbour 1988, Billings 1988, Major and Taylor 1988).

Fires are naturally infrequent in subalpine forests because of low fuel loads; consequently, return intervals have not been significantly altered by suppression activities. Natural fires are likely confined to individual trees or small patches following lightning strike ignitions. (Minnich 1988.)

When subalpine forests do burn, it is usually because of anthropogenic ignitions in a stand-replacing crown fire during severe weather conditions. Several crown fires have occurred in recent decades in subalpine forests in the San Gabriel Mountains. While they were the result of human-caused ignitions, it is unclear if these fires were abnormally severe or frequent. The extremely steep terrain in the San Gabriel Mountains may make these forests more vulnerable as anthropogenic fire ignitions increase. (Stephenson and Calcarone 1999.)

Climate Change: Alpine plants and the upper treeline ecotone have received considerable scientific attention because they are considered to be relatively sensitive to changes in climatic conditions. On other mountain ranges in California, there is evidence of krummholz tree dieback and the lowering of the upper treeline during recent cooler climatic periods. As the climate has become warmer during the twentieth century, there is also evidence of the corresponding upslope movement of treelines, krummholz forms becoming more tree-like, and an increasing density and vigor of alpine plants.

Because plant growth in alpine and subalpine habitats appears to be generally highly responsive to small changes in climate, these habitats have been important proxy indicators of long-term and recent changes in temperature and precipitation. (LaMarche and Mooney 1967; LaMarche 1973; Scuderi 1987, 1994; Anderson 1990; Graumlich 1993; Jennings and Elliott-Fisk 1993; Holtmeier 1994.)

Management Considerations: Management considerations for alpine and subalpine habitats are mostly related to recreational activities and associated development, such as ski areas, roads and hiking trails, and preserving and restoring site-specific resources. These habitats generally do not require fuel or vegetation management activities, or support livestock use (Stephenson and Calcarone 1999). As interest in vegetation responses to climate change increases, the scientific value of studying alpine and subalpine habitats and ecological processes may increase, especially in the marginal and isolated alpine habitats of southern California.

A substantial portion of the alpine and subalpine habitat on the four southern California National Forests occurs within designated wilderness areas, including the Sheep Mountain and Cucamonga Wilderness Areas on the Angeles National Forest, the San Gorgonio and San Jacinto Wilderness Areas on the San Bernardino National Forest, and the Mount San Jacinto State Wilderness Area (Stephenson and Calcarone 1999).

The conservation of alpine and subalpine habitats has important scientific value. The demonstrated high degree of responsiveness to small directional climatic changes (including prevailing temperature and precipitation) and the long-lived and well-preserved tree specimens and other plant material continue to provide important insights on the rate and type of ecological responses to climate change. The relative isolation of the southern California mountains at the southwestern extreme of alpine habitats in North America increases the scientific interest of these habitats and the importance of their conservation. (Elliott-Fisk and Ryerson 1988, Betancourt et al. 1990, Holtmeier 1993,Thompson et al. 1993.)

Special-Status Species: Alpine and subalpine areas provide habitat for 15 special-status plants, including eight USDA Forest Service Region 5 Regional Forester's Sensitive Plant Species and one bird species. All are restricted to the Angeles and San Bernardino National Forests with the exception of one plant species found on Mount Pinos on the Los Padres National Forest. Little is known about the population status of most of these species (Stephenson and Calcarone 1999). Further research should be conducted on the distribution, population status, and reproduction and restoration requirements of alpine plants and habitats.

Habitat Restoration: Disturbed alpine and subalpine habitats often require active restoration activities for successful revegetation because of the natural impediments to seedling establishment, such as fragile and poorly developed soils and climatic extremes and variations. Because of the harsh growing conditions and fragmentation of alpine habitats, restoration projects should incorporate locally adapted seed sources whenever possible. Many species of conifer trees in southern California, for example, have comparatively high genetic diversity within and among populations because of the continuous occupation of habitats during Pleistocene glaciation and the current fragmentation of subalpine habitats (Ledig 1987). Because plant growth and production are slow, restoration sites typically require relatively long-term monitoring and protection during the establishment phase.

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Chaparral

Description: Chaparral refers to several types of plant communities dominated by dense, evergreen, sclerophyllous, multi-stemmed woody shrubs. There are many types of chaparral that vary widely in species composition, but they all share similar physiognomy and ecological relationships. Herbaceous vegetation in all chaparral types is uncommon, as most types have little or no understory cover, with the exception of the first few years following a wildfire event. While some slope affinities exist, slope exposure or topographic position is generally a poor predictor of species composition (Keeley and Keeley 1988). On National Forest System lands in southern California, eight types of chaparral are recognized on the basis of dominant species composition: chamise chaparral, scrub oak chaparral, montane chaparral, redshank chaparral, ceanothus chaparral, northern mixed chaparral, southern mixed chaparral, and serpentine chaparral (Stephenson and Calcarone 1999).

Chamise chaparral is the most common chaparral type in California and is dominated by nearly monotypic, even-aged stands of chamise (*Adenostoma fasiculatum*) up to 10 feet (3 meters) tall. Other commonly associated shrubs that occur in low numbers may include manzanitas (*Arctostaphylos* spp.), ceanothus (*Ceanothus* spp.), scrub oak (*Quercus berberidifolia*, *Q.* spp.), mission manzanita (*Xylococus bicolor*) and coastal sage scrub species such as black sage (*Salvia mellifera*) and Cleveland sage (*S. clevelandii*) (Beauchamp 1986, Hanes 1988). Chamise forms a closed canopy, and there is typically little or no understory herbaceous vegetation. Chamise is especially dominant on broad south- and west-facing slopes (Hanes 1988, Keeley and Keeley 1988).

Redshank chaparral is dominated by redshank (*Adenostoma sparsifolium*), which grows up to 10 feet (3 meters) in height (Beauchamp 1986). Typically, at least 50% of the total shrub cover consists of other commonly associated shrubs, including chamise, ceanothus, manzanita, sumac (*Rhus ovata*), and scrub oaks (*Quercus* spp.). (Hanes 1988.)

Scrub oak chaparral is dominated by species of scrub oak and is 10-18 feet (3-5 meters) in height. Commonly associated species that typically contribute more than 50% of the shrub canopy cover include ceanothus, chamise, manzanita, toyon (*Heteromeles arbutifolia*), red berry (*Rhamnus illicifolia*), California coffeeberry (*R. californica*), garrya (*Garrya* spp.), twinberry (*Lonicera* spp.), and poison-oak (*Toxicodendron diversilobum*). (Hanes 1988.)

Montane chaparral occurs at higher elevations in the lower montane and montane conifer zones. Montane chaparral occurs as a low shrub cover, typically less than 6 feet (2 meters) in height, both in openings in montane conifer stands and in broad stands without trees. Because it has a more open structure, grass and herb cover may be present in openings within stands of mature shrubs. Common dominant species include species of manzanita, ceanothus, garrya, scrub oak, bush chinquapin (*Chrysolepis sempervirens*), toyon, red berry, and California coffeeberry. (Hanes 1988, Keeley and

Keeley 1988.)

Ceanothus chaparral is dominated by species of ceanothus and is normally found on more mesic slopes such as northern exposures. Ceanothus forms a complete canopy cover 10-18 feet (3-5 meters) in height, and it is commonly associated with species of chamise, manzanita, scrub oak, sumac and toyon. (Hanes 1988.)

As their names imply, no single species predominates in canopy cover in northern mixed chaparral and southern mixed chaparral. *Northern mixed chaparral* is comprised of chamise, several species of ceanothus and manzanita, mountain mahogany (*Cercocarpus betuloides*), toyon, poison-oak, and scrub oak. *Southern mixed chaparral* forms a relatively tall, dense canopy 10-20 feet (3-6 meters), and sometimes as much as 35 feet (11 meters), in height. Dominance is commonly shared by chamise, yerba santa (*Eriodictyon* spp.), mission manzanita, toyon, ceanothus, manzanita, scrub oak, yucca (*Yucca whipplei*), lemonade berry (*Rhus integrifolia*), and laurel sumac (*Malosma laurina*). (Hanes 1988, Beauchamp 1986, Keeley and Keeley 1988.)

Chaparral occurring on serpentine soils is addressed in the account of serpentine habitats.

Distribution and Abundance on National Forest System Lands: Chaparral is the dominant ecological community in cismontane, lower montane and foothill areas on the four southern California National Forests at elevations below 5,000 feet (1,524 meters). In general, chaparral is an abundant plant community and wildlife habitat and is well represented on public lands. Only a few chaparral associations are considered relatively rare or poorly represented on public lands: southern mixed chaparral, ceanothus chaparral, and serpentine chaparral. In addition to their presence on National Forest System lands, chaparral habitats are common on other public and private lands throughout the California Coast Ranges and the western slopes of the Peninsular Ranges of Baja California Norte. (Keeley and Keeley 1988.)

There are approximately 2 million acres (809,375 hectares) of chaparral habitat on National Forest System lands. Northern mixed chaparral, which is well-distributed across all four southern California National Forests, comprises nearly 1.5 million acres (607,031 hectares), or three-fourths of all chaparral habitat. Chamise chaparral and scrub oak chaparral also occur throughout southern California, encompassing approximately 300,000 acres (121,406 hectares) and 62,000 acres (25,090 hectares), respectively. (Stephenson and Calcarone 1999.)

On National Forest System lands, red shank chaparral is limited to the Peninsular Ranges on the San Bernardino and Cleveland National Forests. Montane chaparral occurs on 76,405 acres (30,920 hectares) throughout the Peninsular and Transverse Ranges, with large stands on the southern Los Padres National Forest and in the San Bernardino Mountains on the San Bernardino National Forest. Southern mixed chaparral is limited to 27,915 acres (11,297 hectares) on the Cleveland National Forest. Ceanothus chaparral occupies 4,182 acres (1,692 hectares) of National Forest System lands from the eastern Transverse Ranges through the northern Santa Lucia Ranges; however, nowhere in its range is it abundant, and most stands are relatively small. Serpentine chaparral is limited to 1,270 acres (514 hectares) on the Los Padres National Forest (Stephenson and Calcarone 1999). Ceanothus chaparral is poorly represented on public lands. (Stephenson and Calcarone 1999.)

Ecological Processes: Chaparral is characterized by drought-tolerant, sclerophyllous shrubs and herbs that are adapted to cool and moist winters, long summers that are hot and dry, and periodic fire occurrence. Chaparral growth is positively correlated with precipitation. Despite its presence in a climate defined by long, hot summers, maximum shrub growth typically occurs during winter and early spring in response to available soil moisture. The amount of winter precipitation following a burn event can also be important in determining the regeneration success of individual species. (Hanes 1988, Keeley and Keeley 1988.)

Despite its location in one of the hottest and driest parts of the southern California coastal landscape, chaparral is very important for managing watershed health and riparian habitats. Chaparral occurs on shallow, nutrient-poor, coarse-textured soils, often on steep slopes. Intense and large-scale wildfires can remove vegetation from entire watersheds or substantial portions of watersheds, and soil erosion and sedimentation following fires in chaparral can degrade low-elevation riparian habitats. (Stephenson and Calcarone 1999.)

Periodic fire is an integral ecological process in all chaparral habitats, and the prevailing fire regime is a major influence on the patterns of chaparral species composition, habitat patch size, and habitat structure. Mature chaparral readily carries fire because of regular summer dryness, the density and structure of live and dead fuels, the high oil content of the wood and foliage, and periodic droughts that cause shrubs to die back. Chaparral species exhibit distinctive fire adaptations: they quickly reach reproductive maturity and produce seeds that persist for many years in the soil seed bank and that require heat scarification and high nutrient flux to germinate. Additionally, most common shrub species also have the ability to stump sprout from a lignotuber or root crown following a fire. (Hanes 1988, Keeley and Keeley 1988.)

Fire is the dominant regenerative force, with many species dependent on fire for reproduction. Fire initiates the regeneration process for most dominant chaparral shrubs by removing competition and shade, releasing nutrients and minerals to the soil, scarifying seeds, and removing allelopathic and phytotoxic compounds that accumulate in the soil beneath some shrub species. (Hanes 1988, Keeley and Keeley 1988.) Seeds of dominant shrubs and herbaceous annuals, for example, persist only in the seed bank until sunlight, heat, charcoal, and soil nutrients become available following a burn, enabling them to germinate and grow. Grass and herb species then decline in abundance as shrub cover develops from seedlings and stump sprouts (Hanes 1988).

While stump sprouts may appear within a few weeks following a burn, the first wet season following a burn normally produces a flush of new seedlings. Herbs and grasses, absent from mature chaparral stands but present in the seed bank, typically dominate the cover for the first few years. Within 5-10 years, stump sprouts and seedling shrubs begin to dominate the cover, and shrub cover usually peaks approximately 25 years after a fire. By approximately 50 years, shrubs are becoming senescent and the stand is ready to burn again, as total live shrub cover drops and the proportion of dead cover increases. (Hanes 1971, 1988; Keeley and Keeley 1988.)

Many species of chaparral reproduce both by seed and stump sprouting following a fire occurrence (Hanes 1988, Keeley and Keeley 1988). In general, low-intensity fires result in less mortality among stump-sprouting species and greater seed bank survival among obligate seed reproducers. The intensity and seasonality of fire, plant size and health, and site condition are also important influences on plant regeneration. Intense fires tend to result in a decrease in stump sprouting and to favor species with greater capacity for seed reproduction. In general, obligate seed reproducers are more common on xeric slopes and ridges where fires would be expected to burn hotter, whereas stump-sprouters are more common on mesic slopes where fires would burn with less intensity. (Keeley and Keeley 1988.)

Prefire floristic composition is normally a good predictor of postfire species composition. Many shrub species tend to form single-aged, monotypic stands following fire. Most chaparral shrubs appear to be resilient to fire-return intervals of anywhere from 25 to 100 years. (Keeley 1986.) However, a short fire-return interval (e.g., less than 15 years) can result in a shift in species composition to a greater representation of herb and grass species (Zedler et al. 1983, Malanson 1984). Obligate seed-reproducing shrubs typically require 5-15 years of growth to produce mature seeds and develop a seed bank in the soil for postfire regeneration.

Chamise regenerates vigorously by both seed and stump sprouting following a low-intensity fire, but chamise and other facultative stump sprouters such as *Ceanothus* spp. often decline after a very hot, high-intensity fire (Keeley 1986). Scrub oak, toyon, and laurel sumac are all obligate stump sprouters and can stump sprout vigorously even following intense fires (Keeley and Keeley 1988). While low-intensity burns favor the stump-sprouting success of chamise, some species of ceanothus do not stump sprout following a fire but have very vigorous seed regeneration following a very hot, intense burn, especially on drier slopes. Yerba santa and mission manzanita both reseed and stump sprout following fires (Keeley and Zedler 1978; Keeley and Keeley 1987, 1988).

Factors that Influence Ecological Processes: Changes in land use and the fire regime (fire intensity, magnitude, and frequency) are the most important factors influencing ecological processes in chaparral habitats. As with other habitats in wildland-urban interface areas, land development has resulted in habitat fragmentation, loss, and degradation for many wildlife species (California Partners in Flight 2000). Development has also created fire management problems because of increased ignition sources and difficulties conducting landscape-level fuel management practices.

The natural fire-return interval in chaparral is a widely debated issue, but research suggests the current average interval of 50-70 years has been in place throughout much of the Holocene (Minnich 1988). Fire-prone areas along the wildland-urban interface, however, burn more frequently, and this has resulted in some habitat type conversions (Stephenson and Calcarone 1999).

While the average fire return interval may be a relative constant, comparisons between historic presuppression wildfires in California and current chaparral fire behavior in Baja California Norte suggest that the average size and intensity of chaparral fires may have increased during the twentieth century (Minnich 1983, 1987, 1988). Chaparral fires without suppression tend to burn for long periods (weeks to months) but in a very patchy pattern with varying intensity. Fires smolder and burn with low intensity during cool, mild weather, then flare up with hot, dry winds. The result on the landscape is a mosaic of burned and unburned chaparral habitats; within burned patches, a mosaic of fire intensity results in differential reproduction success of shrub species. A landscape mosaic of chaparral age, species composition, and fuel type and structure in turn support a mosaic fire pattern in the future. (Minnich 1983, 1987, 1988.)

By contrast, active fire suppression may lead to fires burning only during extreme fire weather resulting from human-caused ignitions. These are typically large (e.g., more than 10,000 acres [4,045 hectares]), intense fires that create a homogeneous vegetation pattern with large tracts of brush of a single age class, favoring species that reproduce well with intense burns. There have been a number of these large fires over the last several decades, particularly on the Los Padres National Forest. This simplified age-class mosaic, while not a problem for the shrubs themselves, reduces habitat diversity, which generally leads to a reduction in wildlife diversity. Game animals like quail and deer, which tend to be habitat edge species, generally avoid large, unbroken tracts of chaparral. Large fires degrade riparian habitats by resulting in substantial increases in erosion and sedimentation in watersheds. Large chaparral fires also have the propensity to be self-perpetuating because they create a continuous block of single-age vegetation that becomes ready to burn again at the same time. (Minnich 1983, Stephenson and Calcarone 1999.)

The increase in size, the possible increase in intensity, and the increase of low-elevation starts of chaparral fires also affect adjacent vegetation communities. The fire regimes in lower montane conifer and mixed evergreen stands, for example, are largely controlled by fire occurrences in the chaparral habitats surrounding these low-elevation forest types. Chaparral fires are increasingly carrying into bigcone Douglas-fir (*Pseudotsuga macrocarpa*) stands, which are very slow to recover from stand-replacing fires. This has resulted in the reduction of bigcone Douglas-fir by an estimated 18% in San Bernardino Mountains since the 1930s, as well as declines in other areas (Minnich 1988, Stephenson and Calcarone 1999).

An issue that is not of particular concern is the potential for chaparral stand senescence or decadence due to the long-term absence of fire. Formerly believed to be a problem, long fire-free intervals of 70-100 years have been found in recent studies not to be detrimental to chaparral shrubs (Keeley 1986). Many species, in fact, continue to produce new stems and even seedlings during long fire-free intervals. The potential for negative effects associated with overly frequent fires or extensive fires is a much greater threat to chaparral and wildlife habitat diversity (Stephenson and Calcarone 1999). Large, contiguous patches of mature shrubs improve habitat value for many native bird species associated with chaparral habitats (California Partners in Flight 2000).

Management Consideration: Forty-seven special-status plant species occur or may occur in chaparral habitats on southern California National Forest System lands. Of these, six are federally listed and 28 are USDA Forest Service Region 5 Regional Forester's Sensitive Species. Most of these species are considered to have low vulnerability and relatively stable population trends on National Forest System lands because of the lack of management actions that degrade or modify chaparral habitats. The construction of fuel breaks is perhaps the greatest threat to special-status species in chaparral, but this is generally a site-specific action. The overall population status and trends of approximately 20 special-status plants, however, are unknown because of a lack of data. (Stephenson and Calcarone 1999.)

There are 13 special-status wildlife species associated with chaparral habitats on the southern California National Forest System lands. Only eight of these, however, are associated primarily with chaparral. In general, these species have low to moderate vulnerability on National Forest System lands, but their population trends are largely unknown. (Stephenson and Calcarone 1999.)

All four southern California National Forests are undertaking reasonable and prudent measures to avoid or minimize incidental take of federally listed chaparral species (USDI Fish and Wildlife Service 2001). The USDI Fish and Wildlife Service has provided terms and conditions to implement reasonable and prudent measures with which the USDA Forest Service must comply during the undertaking of management actions in order to be exempt from the take prohibitions for federally listed species in Section 9 of the Endangered Species Act (ESA). Section 7(a)(1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Recommended conservation programs include activities carried out by discretionary agencies to avoid or minimize adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information. The conservation actions recommended by the USDI Fish and Wildlife Service (2001) for chaparral habitats and associated listed species include:

conducting focused surveys and monitoring efforts to determine the ranges, sizes, and trends of populations of listed species in chaparral habitats;

protection of critical habitat;

prevention of off-highway vehicle use and other activities that may degrade occupied habitats; and

minimizing impacts on chaparral habitat occupied by listed species when implementing fuel management practices, trail and road use and maintenance, use of developed recreation sites, special use permit activities, grazing activities, administrative sites, and road construction.

Management of chaparral should also consider low-elevation riparian habitats. Following wildfires in chaparral, large soil erosion and sedimentation events can degrade instream and floodplain riparian habitats. Fuel management practices that create small, different-aged patches of chaparral within a given watershed tend to create conditions that are less likely to result in complete, uniform burns that could degrade the entire watershed. Fuel breaks and firebreaks should be in place to facilitate access and fire suppression activities at strategic locations for watersheds with sensitive resources at risk from sedimentation events.

On a landscape level, increasing fire frequency in some chaparral habitats in high-ignition areas (e.g., near the wildland-urban interface, along developed roads and recreation areas) may cause habitat conversion from chaparral to more grass- and herb-dominated communities. However, frequent fire occurrence in interface areas can be a beneficial management strategy by maintaining low overall fuel loads near ignition sources, creating more defensible space around developments, and protecting mature chaparral stands and wildlife habitats from catastrophic wildfire losses. (California Partners in Flight 2000.)

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Coastal Sage Scrub

Description: Coastal sage scrub is one of the two major scrub formations that occur in the California floristic province. It is characterized by low- to medium-height shrubs with semi-woody, flexible stems and soft leaves that are facultatively drought-deciduous. Characteristic species include California sagebrush (*Artemisia californica*), California buckwheat (*Eriogonum fasciculatum*), and several sage (*Salvia*) species (Mooney 1988). It co-occurs with chaparral but typically grows where there is less available soil moisture because of low rainfall, slope aspect, or edaphic factors (Harrison et al. 1971).

Coastal sage scrub is not a uniform plant community but is a vegetation formation that comprises many different subunits. Southern coastal sage scrub, which occurs from San Luis Obispo County south into Baja California, has been divided into three subformations: Venturan sage scrub, which occurs on the coastal side of the mountains from San Luis Obispo County to Los Angeles County; Diegan sage scrub, which occurs along the immediate coast of Orange County to San Diego County, within the zone where proximity to the ocean has a moderating effect on the climate; and Riversidian sage scrub, which occurs farther inland to the base of the Transverse Ranges and in the Pensinsular Ranges, from Los Angeles and Riverside Counties south to San Diego County (Axelrod 1950, Thorne 1976, Kirkpatrick and Hutchinson 1977, Westman 1983, Davis et al. 1994). Many plant series and associations have been identified within southern coastal sage scrub (Kirkpatrick and Hutchinson 1980, DeSimone and Burk 1992, Davis et al. 1994, White and Padley 1997, Weaver 1998); these have been summarized in Sawyer and Keeler-Wolf (1995).

Distribution and Abundance on National Forest System and Adjacent Lands: Coastal sage scrub is one of the dominant ecological communities in southern California on the coastal side of the mountains below 5,000 feet (1,524 meters). Coastal sage scrub occurs primarily at elevations below 2,500 feet (762 meters) and is most widespread in coastal valleys and plains west of the foothills. It occurs on all four southern California National Forests, with almost half of the total amount occurring on the Los Padres National Forest. The total amount of coastal sage scrub on the southern California National Forests is 522,735 acres (211,544 hectares): 95,292 acres (38,563 hectares) on the Cleveland National Forest; 51,505 acres (20,843 hectares) on the San Bernardino National Forest; 135,876 acres (54,987 hectares) on the Angeles National Forest; and 240,062 acres (97,150 hectares) on the Los Padres National Forest (Stephenson and Calcarone 1999).

Ecological Processes: The plant species composition of coastal sage scrub has been shown to be associated with habitat variables such as altitude, slope aspect, and substrate (Cole 1980, Kirkpatrick and Hutchinson 1980, O'Leary 1988, DeSimone and Burk 1992). However, the vegetation makeup does not appear to be determined by a direct response to these habitat variables, but instead appears to be dependent on individual species responses to factors such as soil moisture availability, disturbance, and fire (Wells 1962, Freudenberger et al. 1987, O'Leary 1988, Gill and Hanlon 1998).

Factors that Influence Ecological Processes: Disturbance, especially fire-related disturbance, appears to play a major role in the coastal sage scrub community. Coastal sage scrub has been characterized as a seral community that needs disturbance to become established and, in the absence of disturbance, would be replaced by chaparral (Epling and Lewis 1942, Axelrod 1978). Most authorities currently regard coastal sage scrub as a self-replacing vegetation type that exists in a dynamic relationship with other formations, although disturbance is believed to have a role both in affecting the composition of the coastal sage scrub community and in affecting the interaction between coastal sage scrub, chaparral, grassland, and other plant communities (Westman 1979, Callaway and Davis 1983, Gray 1983, Freudenberger et al. 1987, O'Leary and Westman 1988, DeSimone and Zedler 1999).

Fire: The role of fire in the coastal sage scrub ecosystem is still poorly understood. Coastal sage scrub has always been subject to periodic fires. Historically, fires occurred at low frequency, generally during the summer months; were of low intensity; and generally were of small size (Minnich 1983). Human-caused fires and fire management practices in Southern California have altered these parameters.

Fire has both short-term and long-term effects on structure and composition of coastal sage scrub. In the short term, aboveground woody cover is greatly reduced and mortality is high, although the ability to withstand fire varies among species. Following removal of the woody canopy, many seedlings are able to become established, and herbaceous cover is highest following fire (Westman 1981a). Alteration of the vegetation structure and composition also alters the composition of the fauna (Price and Waser 1984, Schwilk and Keeley 1998). Burning reduces the amount and diversity of foraging habitat for both resident and nonresident species and reduces cover for nesting (Stanton 1986).

Coastal sage scrub responds quickly after fire; the vegetation structure and composition reestablishes within a few years. Wildlife use also resumes its prefire pattern within a short time. However, the plant associations present following fire may or may not be the same as before the fire. Postfire recovery patterns vary among sites, and identifying the factors that influence the vegetation response is difficult because of the many variables involved (White 1995). Fire characteristics, such as intensity and fire interval, may affect both the structure and composition of the coastal sage scrub community. Following burns, shrubs on sites with lower intensity fire were larger than those on sites with higher intensity fire (Malanson and O'Leary 1982). Frequent fires can alter the composition of the vegetation by favoring sprouters over seeders (Zedler et al. 1983). With shorter fire intervals, growth of herbaceous species may inhibit establishment of shrubs by seed (Malanson and O'Leary 1982). Short fire intervals may reduce or eliminate some species, and greater diversity is expected with longer fire intervals (Malanson 1985, Haidinger and Keeley 1993). Disturbance may accentuate these postfire successional patterns (O'Leary and Westman 1988).

Land Use: The extent of coastal sage scrub has been greatly reduced by land conversion to agricultural, industrial, and residential uses; flood control projects; rock quarries; and other projects (Hanes 1976, O'Leary 1995, Davis et al. 1994). Estimates of the reduction in the historical extent of southern coastal sage scrub range as high as 90% (Westman 1981a). Much of the remaining coastal sage scrub habitat has been fragmented or degraded (O'Leary 1995, Minnich and Dezzani 1998). Only about 7% of coastal sage scrub is on public lands managed for conservation, and most is on private lands (Davis et al. 1994).

Other threats include fire management practices, air pollution, grazing, and exotic species. Construction of fire breaks in coastal sage scrub removes habitat, causes habitat fragmentation, and creates a disturbance corridor allowing invasion of nonnative species; moreover, aggressive fire suppression has led to fuel buildup in chaparral, resulting in catastrophic fires that spread into adjacent coastal sage scrub stands (Vogl 1976). Air pollutants such as ozone and sulphur dioxide appear to have adverse effects on plant physiology and growth (Westman 1985, Preston 1988). Disturbance by grazing may allow invasion of nonnative species, although such species may also be able to invade by competitive exclusion (Minnich and Dezzani 1998). Because many coastal sage scrub species are not widely distributed, there is a high potential for extinction, both locally and globally (Westman 1981a).

Management Considerations:

Fire Management: Coastal sage scrub and chaparral have different responses to fire and fire management, and this difference needs to be addressed in fire management practices (Minnich 1983, Malanson and Westman 1985, White 1995). Fire may not have been as necessary for the persistence of coastal sage scrub as it is for chaparral (White 1995). However, because this ecosystem is increasingly adjacent to or surrounded by the urban interface, it is becoming more difficult to prevent frequent fires and subsequent habitat degradation.

Prefire plans should be created that specifically prescribe measures for coastal sage scrub, such as let-burn areas, controlled burns, and fire intervals (White 1995). Prescribed burns, if used as a management technique, should be planned at long-spaced intervals and implemented in fall, to emulate the natural fire pattern (Westman 1982, Malanson and Westman 1985, O'Leary 1995). Spring and summer burns are especially damaging to coastal sage scrub species, and lower fuel loads in coastal sage scrub indicate that longer fire intervals are needed (Malanson 1985, O'Leary 1989).

Fire-resistant landscaping, rather than fuel-free zones, should be used at wildland boundaries (White 1995). Ryegrass should not be planted for postfire erosion control because it may interfere with shrub and herb recovery (O'Leary 1988). The use of nitrogen fertilizer in fire retardant should be evaluated; although some coastal sage scrub species respond positively to nitrogen availability (Gray and Schlesinger 1983, Zink and Allen 1998, Padgett and Allen 1999), the available nitrogen might promote the growth of nonnative weeds over coastal sage scrub species (Zink and Allen 1998).

Habitat Restoration: Habitat restoration appears to have limited usefulness for conserving coastal sage scrub. Restoration of coastal sage scrub has been attempted through planting of seeds or nursery stock and by employing various soil treatment and irrigation regimes (Hillyard and Black 1987, Sproul 1988, O'Connell and Erickson 1995). Although some coastal scrub restoration attempts have produced a plant community with a shrub canopy that emulates natural coastal scrub, such restoration is still experimental, and the habitat value of restored coastal sage scrub has yet to be determined (Sproul 1988, O'Leary 1989, Bowler 2000). California gnatcatchers (*Polioptila californica*) have been observed nesting in restored coastal sage scrub (O'Connell and Erickson 1998, Miner et al. 1998). However, restoration efforts that have utilized a simple model of coastal sage scrub have resulted in reduced habitat diversity (Bowler 2000). Irrigation has not been shown to be an effective method for enhancing establishment of coastal sage scrub and may have adverse effects (Padgett et al. 2000).

Special-Status Species: Many special-status plant and animal species are associated with coastal sage scrub. Nearly 200 special-status plant species occur in southern coastal sage scrub, and 39 of these are known to occur or could potentially occur on the Cleveland, San Bernardino, Angeles, and Los Padres National Forests (California Natural Diversity Database 2001, California Native Plant Society 1994-2001). Atwood's (1993) tabulation of special-status animal species associated with coastal sage scrub in southern California included eight invertebrates, one amphibian, eight reptiles, five birds, and nine mammals. There are 13 special-status wildlife species that occur or potentially occur on the Cleveland, San Bernardino, Angeles, and Los Padres National Forests (Stephenson and Calcarone 1999).

Establishing Habitat Reserves: Conservation of coastal sage scrub on public lands is important because most coastal sage scrub is on private lands (Davis et al. 1994), and conservation of coastal sage scrub on private lands faces strong opposition by self-interest groups (DeSimone and Silver 1995). Because coastal sage scrub comprises a mosaic of different species associations, use of a single concept of coastal sage scrub is inappropriate (Westman 1981b, DeSimone and Burk 1992, White and Padley 1997, Bowler 2000). Single species are poor indicators of species richness; focusing on single species, such as special-status species, does not focus on high-quality habitat (Bowler 2000, Chase et al. 2000, Rubinoff 2001). For example, the California gnatcatcher (federally listed as threatened) has not been found to be a good umbrella species, nor is it an indicator of species richness, and its presence/absence should not be used as the primary criterion for selecting habitat for preservation (Chase et al. 1998, Fleury et al. 1998, Rubinoff 2001). Therefore, preserves should be established that encompass the diversity of associations within a given area (Westman 1981b). Coastal sage scrub preserves also may need active management to prevent displacement by invasive nonnative species (Minnich and Dezzani 1998).

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Desert Montane

Description: Pinyon-juniper woodlands, semi-desert montane chaparral, and Great Basin scrub occur on semi-arid desert-side slopes of the Transverse, Peninsular, and Tehachapi Ranges of southern California. Single-leaf pinyon pine (*Pinus monophylla*) generally dominates the higher elevation slopes and extends into the lower montane forests and woodlands, while California juniper (*Juniperus californicus*) and western juniper (*J. occidentalis* var. *australis*) codominate some stands with pinyon, especially on gentle slopes or alluvium.

Pinyon-juniper woodlands are open-canopy stands with sparse understory vegetation. Mature stands typically have a 20-35 foot (6-10 meter) canopy height and 30% canopy cover with 5-10% shrub and herb cover (Everett and Koniak 1981, Koniak 1986, West 1988, Burwell 1999). Understory shrubs are primarily those from semi-desert montane chaparral and basin sagebrush communities.

Four-needle pinyon (*Pinus quadrifolia*) occurs with single-needle pinyon and California juniper in the San Jacinto Mountains and further south on the eastern slopes of the Sierra Juarez and Sierra San Pedro Martir in Baja California Norte. It replaces single-needle pinyon in some stands.

Semi-desert montane chaparral and Great Basin scrub have lower and more open structures and different species compositions than cismontane foothill chaparral. Well-developed semi-desert montane chaparral typically has approximately 50% shrub cover, and mature stands of Great Basin scrub are usually open with less than 50% total shrub cover. Average shrub canopy height is about 6 feet (two meters). Common species in montane chaparral include flannelbush (*Fremontodendron californicum*), bitterbrush (*Purshia tridentata*), scrub oak (*Quercus* spp.), mountain mahogany (*Cercocarpus betuloides*), and cupleaf ceanothus (*Ceanothus greggii* var. *perplexans*).

Great Basin scrub dominated by sagebrush (*Artemisia tridentata*) and rabbitbrush (*Chrysothmnus nauseosus*) is common in alluvial fans, dry meadows, and washes. Great Basin scrub is particularly prevalent on alluvial soils in the low elevations surrounding Mount Pinos and in the Garner Valley region south of Mount San Jacinto. Both shrub-dominated communities intergrade with pinyon-juniper woodlands.

Distribution and Abundance on National Forest System and Adjacent Lands: Public lands on and adjacent to the four southern California National Forests encompass approximately 288,096 acres (116,589 hectares) of pinyon-juniper woodlands, 192,802 acres (78,025 hectares) of semi-desert montane chaparral, and 21,700 acres (8,782 hectares) of Great Basin scrub (Stephenson and Calcarone 1999). All of these habitat types are very common with wide distribution on public and private lands in eastern California and the Intermountain West.

Ecological Processes: Pinyon-juniper woodland is a relatively stable, self-replacing vegetation type. Unlike other pine species, pinyon is very shade tolerant; it requires the ameliorated microclimate beneath mature nurse plants, such as the canopy of other trees or large shrubs, for seed germination and successful establishment (Drivas and Everett 1988, Barton 1993, Burwell 1999). Exhibiting classic stress-adapted growing traits, pinyon and juniper are very slow growing, have high water and nutrient use efficiencies, and opportunistically utilize soil moisture from rare summer rainfall (Drivas and Everett 1988; DeLucia et al. 1989; DeLucia and Heckathorn 1989; DeLucia and Schlesinger 1990, 1991; Everett and Thran 1992; Barton and Teeri 1993).

Pinyon-juniper woodlands are more dominant on relatively rocky, coarse textured soils with low nutrient and water availability (Burwell 1999). These sites act as fire refugia because of the low cover and productivity of shrubs and herbaceous plants, as well as their position on steep slopes and dissected topography (West 1988). In addition, the root structures and efficient use of water and nutrients give pinyon and juniper a competitive advantage on sites with poorly developed soils and low nutrients and water availability (Drivas and Everett 1988, DeLucia et al. 1989, DeLucia and Heckathorn 1989, DeLucia and Schlesinger 1991, Everett and Thran 1992).

Semi-desert montane chaparral and Great Basin scrub are more dominant on sites with deeper alluvial soils and broad, gentle slopes. Unlike pinyon and juniper trees, montane desert shrubs tend to use soil nutrients and groundwater aggressively as a competitive exclusion strategy. Shrubs are less efficient with resource use but more tolerant of prolonged soil moisture drought. Shrubs uptake water until it is depleted from storage, then shut down primary production. After depleting groundwater stored from winter rains, basin sagebrush drops its larger leaves and is largely inactive the remainder of the summer with small, thick, leathery leaves that minimize water loss (Drivas and Everett 1988, DeLucia et al. 1989, DeLucia and Heckathorn 1989, DeLucia and Schlesinger 1991, Everett and Thran 1992).

Desert montane habitats are important for migratory montane wildlife species. During fall, winter, and spring, pine nuts are an important food source for corvid birds and rodents, which cache the seeds (Vander Wall and Balda 1977, Balda and Masters 1980, Stotz and Balda 1995, Vander Wall 1997). Pinyon-juniper woodlands provide cover, and bitterbrush is important browse for deer as they migrate downslope from higher elevations during winter (Leach 1956, Nord 1965, Short et al. 1977).

Factors that Influence Ecological Processes: Over the last 150 years, pinyon-juniper woodlands have expanded their range by invading adjacent shrub-dominated communities, and existing stands have become denser (Hastings and Turner 1965, Blackburn and Tueller 1970, Burkhardt and Tisdale 1976, West et al. 1979, Rogers 1982, Burwell 1998, West 1988). The reasons for this expansion are most likely a combination of introduced livestock grazing, reducing the cover of competing grass species and fuel to carry wildfires; and an increase in summer precipitation during this period (Minnich 1988, Wigand and Nowak 1992, Burwell 1999). Some early reports suggest that desert montane chaparral and Great Basin scrub were grass-dominated communities at the time of European settlement and became degraded and shrub-dominated as a result of overgrazing. A more probable scenario is that these communities were always shrub-dominated, but livestock grazing has reduced overall plant cover and especially herbaceous cover (Vale 1975, Minnich 1988).

Fire: In contrast to cismontane chaparral and conifer forests, desert montane vegetation types are not adapted to periodic wildfire occurrences. Pinyon-juniper woodlands do not readily carry fire; when fires do occur, they are typically intense, stand-replacing events. Mature pinyon and juniper trees are readily consumed by fire and are not resistant to even low-intensity burns because they have thin, resinous bark, dense branching and foliage, and are not self-pruning (Leopold 1924, Barton 1993). There is little fuel accumulation or contiguity, and tree canopies are widely spaced. Consequently, fires require severe fire weather conditions, such as hot temperatures, low humidity, and strong winds (Bruner and Klebenow 1979, Young and Evans 1981, Minnich 1988).

Pinyon-juniper woodlands recover very slowly from crown fires. Several studies estimate that more than 100 years is required for pinyon and juniper trees to become dominant on a site following a stand-replacing wildfire (Erdman 1970, Barney and Frishcknecht 1974, Young and Evans 1981, Everett and Ward 1984, Koniak 1985, Everett 1987, Tausch and West 1988, Wangler and Minnich 1996). Pinyon does not stump sprout and its seeds do not survive fire. For pinyon to regenerate following a burn, seeds have to be cached by corvid birds or rodents (Vander Wall and Balda 1977, Stotz and Balda 1995, Vander Wall 1997), and seedlings require mature shrubs for nurse plant habitats to become established (Koniak 1985, Burwell 1999). Consequently, 20-40 years of shrub growth is commonly required before tree seedlings can become established (Koniak 1985, Wangler and Minnich 1996).

Fires have historically been infrequent in sparsely vegetated desert-side habitats and, for the most part, this continues to be the case because of the naturally low fuel loads. One study of pinyon-juniper woodlands in the San Bernardino Mountains estimated that the average fire-return interval is 480 years and that active fire suppression has had little effect on this vegetation type (Wangler and Minnich 1996).

In recent years, however, there have been several large fires in pinyon stands in the San Bernardino Mountains and Peninsular Ranges, and some pinyon-juniper woodlands have been reduced because of an increase in human–caused wildfires (Beauchamp 1986, Stephenson and Calcarone 1999). The current fire regime in pinyon-juniper woodlands is largely controlled by the proximity to urbanized areas and level of human use of the montane desert landscape. For example, the high occurrence of multiple fires on the desert side of the San Jacinto Mountains is concentrated in the relatively accessible areas near Beaumont and Palm Springs. By contrast, the most remote desert montane areas, such as the southern Los Padres ranges around Mount Pinos and the extreme northeastern corner of the San Bernardino Mountains, have had very little recorded fire occurrence (Stephenson and Calcarone 1999).

The exceptionally long recovery time and the increase in human-caused fires have converted some pinyon-juniper woodlands to desert chaparral or desert scrub (Wangler and Minnich 1996). Cheatgrass (*Bromus tectorum*) and red brome (*Bromus madritensis* ssp. *rubens*), nonnative undesirable grass species, have invaded some of these former stands. The presence of these pyrophytic grasses tends to increase fire frequency and to cause a type conversion from pinyon-juniper woodland to a grass- and scrub-dominated community with reduced habitat value for wildlife and forage value for livestock.

(West and van Pelt 1987, West 1988, Stephenson and Calcarone 1999.)

Fires are more frequent in montane desert chaparral and Great Basin scrub communities than in pinyon-juniper woodlands because they have greater connectivity of fuels and occur on broad, gentle slopes that carry fire more evenly. There is no evidence, however, that fire regimes in semi-desert chaparral vegetation types are outside the range of natural variability. As in pinyon-juniper woodlands, cheatgrass and red brome can colonize and degrade montane desert chaparral and Great Basin scrub communities and increase fire frequency. (Stephenson and Calcarone 1999).

Diseases: Disease is second only to wildfire as a cause of single-leaf pinyon pine mortality. Black stain root disease has caused the mortality of an estimated 8,000 acres of single-leaf pinyon pine in several stands, particularly in portions of the San Bernardino Mountains. Single-leaf pinyon can be killed directly by the disease or by an attack of pinyon pine engraver beetle, which may be a disease vector (Stephenson and Calcarone 1999).

Land Uses: Site-specific land uses that have degraded desert montane vegetation communities include mining, off-highway vehicle (OHV) recreation, and target shooting. Large limestone deposits in the northeastern portion of the San Bernardino Mountains have resulted in the development of several open-pit mines, and substantial areas are under mining claims that may become active in the future. Exploratory drilling for oil and gas deposits is a major activity in the southern Los Padres National Forest. Target shooters and OHV enthusiasts concentrate in desert-side habitats because the sparsely vegetated terrain is conducive to these activities. Uncontrolled target shooting has raised safety and pollution (particularly from lead accumulation) concerns, and several of the National Forests are currently reviewing their policies on this activity (Stephenson and Calcarone 1999).

These activities have caused mostly localized losses or degradation and fragmentation of habitats and sensitive resources. However, these land uses are not affecting the overall integrity, abundance, or distribution of the desert montane landscape on National Forest Service System lands (Stephenson and Calcarone 1999).

Management Considerations: The stated fire management goal in pinyon-juniper woodlands is to maintain long fire-free intervals (100-300 years) to allow this vegetation type to reach and maintain structural maturity for an extended period of time (Stephenson and Calcarone 1999). Pinyon-juniper woodland and all desert montane communities, however, are not fire-adapted communities, but are stable and self-replacing; complete fire exclusion will generally result in the maintenance of these habitats. Moreover, fire exclusion will help to reduce the probability of invasion by nonnative undesirable species such as cheatgrass and red brome in the desert montane landscape. As described above, fire ignitions are concentrated around popular and accessible areas on or adjacent to National Forest System lands, especially near the wildland-urban interface. Therefore, land use and fuel management planning should consider the risks of new fire starts around popular use areas.

In the absence of disturbance, shrub-dominated communities that have been invaded by pinyon and juniper trees are likely to convert to pinyon-juniper woodlands. Once established, the trees eventually overtop the shrubs, and shrub, grass, and herb cover becomes dramatically reduced (Arnold and Schroeder 1955, Arnold 1959, Arnold et al. 1964, Blackburn and Tueller 1970, Burkhardt and Tisdale 1976, West et al. 1975, West et al. 1979, Everett and Koniak 1981, West and van Pelt 1987). This type conversion dramatically lowers the forage productivity for livestock, and the stocking levels in affected allotments may need to be adjusted downwards.

Special-Status Species: There are 26 special-status plant species associated with desert montane habitats on the four southern California National Forests. Most (18) of these occur or have potential habitat on the San Bernardino National Forest. The trend and status of most populations are unknown, but these species generally have low or moderate vulnerability because of the low level of land and resource use pressures in these habitats (Stephenson and Calcarone 1999).

There are 16 special-status wildlife species associated with desert montane habitats: six invertebrates, one reptile, seven birds, and two mammals. Most (13) of these occur or potentially occur on the San Bernardino National Forest. Peninsular Ranges bighorn sheep (*Ovis canadensis cremobates*) (federally listed as endangered) is highly vulnerable to disturbance and is found in the San Jacinto and Santa Rosa Mountains. The main part of its range, however, is below the desert montane communities on private and other public lands (Stephenson and Calcarone 1999).

Surveys should be conducted to map the extent of occupied habitats for those species on which there is little information and to collect baseline population information in order to determine trends in population sizes and distribution over time.

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Desert Scrub

Description: Desert scrub habitats include a wide variety of vegetation types below the pinyon-juniper and scrub oak chaparral zones, but on National Forest System lands they consist primarily of creosote bush (*Larrea tridentata*) scrub on the north and east sides of the San Gabriel, San Bernardino, and San Jacinto Mountains. Creosote bush scrub, with an understory of burroweed (*Ambrosia dumosa*) and short-lived annuals, is common on stable sites, such as bajadas and some alluvial fans, with coarse-textured soils. (Beauchamp 1986, Vasek and Barbour 1988.)

Vegetation structure in desert scrub varies with soil depth, drainage, and moisture, but rarely exceeds 25% canopy closure and is generally less than 6 feet (2 meters) in height (Vasek and Barbour 1988). Desert scrub habitat types share many species with desert montane habitats where their elevational ranges overlap.

Other desert scrub habitat types include Mojave Desert associations dominated by Joshua tree (*Yucca brevifolia*), saltbush (*Atriplex* spp.), blackbush (*Coleogyne ramoissima*), and shadscale (*Atriplex canescens*). Colorado Desert vegetation associations occur in the vicinity of San Gorgonio Pass and the Coachella Valley. San Joaquin saltbush scrub, a desert scrub habitat particular to the southwestern Central Valley, occurs along portions of the eastern edge of the southern Santa Lucia Ranges and near the Cuyama Valley.

Distribution and Abundance on National Forest System Lands: An estimated 20,682 acres (8,443 hectares) of desert scrub habitat occurs on National Forest System lands in southern California. This represents a very small fraction of the total area of these habitat types in southern and eastern California. The majority of desert scrub on National Forest System lands occurs on the San Bernardino National Forest along the base of the San Jacinto Mountains, with lesser amounts scattered along the lower elevations of the San Bernardino Mountains. The Cleveland National Forest supports nearly 5,000 acres (2,023 hectares) of desert scrub on the eastern slopes of the Peninsular Ranges (Stephenson and Calcarone 1999).

Ecological Processes: Plant growth and productivity generally follow moisture availability in desert scrub habitats. Many species have their most substantial growth period in the spring when soil moisture from winter precipitation is still available. Short-lived annual species persist for several years in the soil seed bank, then germinate and grow in short periods of enhanced soil moisture immediately following occasional summer rainfall. Relatively little is known about the successional stages in many desert scrub vegetation types. Although various structural stages occur, time to recovery after disturbance is unknown. Because of aridity, cool winters, and poor soil development in desert scrub areas, plant growth and productivity are slow. Estimates of post disturbance recovery times range in excess of several hundred years. (Vasek and Barbour 1988.)

Factors that Influence Ecological Processes: Although some livestock grazing has occurred in desert scrub habitats on National Forest System lands, most current sources of habitat degradation in desert areas are site specific; these include mining, oil and gas exploration, off-highway vehicle use, target shooting, and other recreational activities. These activities have caused mostly localized losses or degradation and fragmentation of habitats but are not affecting the overall integrity, abundance, or distribution of desert scrub on National Forest Service System lands. (Stephenson and Calcarone 1999.)

Fire has apparently not been a significant disturbance factor historically. However, the recent spread of nonnative undesirable plant species such as cheat grass (*Bromus tectorum*) into desert scrub habitats typically increases fire occurrence and can initiate type conversion to desert grassland (Young and Evans 1978, Wright et al. 1979, West 1988).

Management Considerations: Low-elevation desert scrub provides habitat or potential habitat for approximately 21 special-status wildlife species, eight of which are state- or federally listed, and two plant species that are federally listed as endangered. Only a few of these species have been observed on National Forest System lands, and their main range and distribution occurs on other private or public lands. While site-specific conservation of habitat for special-status species is important throughout their ranges, the conservation of desert scrub habitats on National Forest System lands is clearly insufficient to maintain viable habitat and population sizes for these species (Stephenson and Calcarone 1999).

The invasion of nonnative undesirable species, such as cheat grass, is an important management consideration in some areas. Nonnative species can degrade habitats and alter ecological processes, such as fire regimes.

Desert scrub habitats on National Forest System lands are isolated, small, and disjunct administratively but not necessarily ecologically. The majority of these habitats occur as the upper reaches of desert washes at the base of mountains. While the total amount of desert scrub habitats on National Forest System lands is extremely small, some of these habitats are relatively undisturbed because of their remoteness and inaccessibility (Stephenson and Calcarone 1999). Consequently, conservation of several desert scrub areas may be locally important for wildlife species sensitive to human disturbances.

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Desert Montane **Gabbro Outcrops**

Gabbro Outcrops

Description: Gabbro outcrops comprise soil types that support unique or distinct plant communities. Gabbro soils, like serpentine, are also described as ultramafic because of their high concentration of mafic minerals such as magnesium and iron and their corresponding low concentration of calcium. The low calcium-magnesium ratio can be toxic or stressful to the establishment and growth of some plant species by limiting their ability to uptake essential soil nutrients (Marschner 1995). Soil pH, soil texture, and other factors can also limit nutrient uptake. Accordingly, there are similarities between vegetation on gabbro soils and that on serpentine iron- and magnesium-rich soil. Soil series formed from gabbro include Las Posas, Boomer, and Auld. Soil derived from gabbroic igneous rock is highly erodable, often poorly draining and clayey, and weathers into a dark reddish color. (Stephenson and Calcarone 1999.)

Gabbro habitats occur mostly within chaparral communities and contain many common chaparral shrub species. Gabbro outcrops, however, have an inherently patchy and fragmented distribution and form distinct ecological islands within more common substrates, such as granodiorite, and support unique plant communities or species composition, such as Cuyamaca cypress (*Cupressus stephensonii*) and Tecate cypress (*C. forbesii*) groves. Additionally, plant species that are largely endemic to gabbro outcrops or that have range extensions on gabbro outcrops occur in these habitats because their tolerance of the stressful soil conditions allows them to avoid competition from intolerant species. Such species include San Diego thornmint (*Acanthomintha ilicifolia*) (federally listed as threatened) and Mexican flannelbush (*Fremontodendron mexicanum*). (Vogl et al. 1988, Stephenson and Calcarone 1999.)

Cuyamaca cypress groves grow in gabbro-derived clay soils on steep slopes along drainages. This species can be dominant in the canopy or codominant with Coulter pine. Groves are typically surrounded by chaparral vegetation composed of chamise (*Adenostoma fasciculatum*), manzanita (*Arctostaphylos* spp.), and scrub oak (*Quercus berberidifolia*). The taxonomy of Cuyamaca cypress is disputed. Some consider Cuyamaca cypress a variant of Arizona cypress (*C. arizonica* ssp . *arizonica*), which occurs in the mountains of Arizona and Baja California Norte and Sonora, Mexico. (Stephenson and Calcarone 1999.)

Tecate cypress groves grow in alkaline, clay soils derived from ultramafic gabbroic rocks or metavolcanics and are usually found on mesic eastern or northern aspects. Tecate cypress groves were once more widespread but are now restricted to these unusual soils where the species lacks competition. Like Cuyamaca cypress, Tecate cypress can be the defining component of southern interior cypress forest, a dense, fire-maintained low forest that forms even-aged stands surrounded by chaparral. Ceanothus, scrub oak, and chamise species are commonly found with Tecate cypress, and the trees may also be viewed as a phase of chaparral vegetation.

Distribution and Abundance on National Forest System and Adjacent Lands

Gabbro outcrops occur on the Cleveland National Forest in the foothills and mountains of San Diego County and the Santa Ana Mountains in Orange County. In the San Diego region, the outcrops are known from Cuyamaca, Guatay, McGinty, Potrero, Viejas, Poser, Los Pinos, Corte Madera, and Iron Mountains (Beauchamp 1986). An estimated 81,680 acres (33,055 hectares) of gabbro-derived soils occur in southern California; 41% of this acreage occurs on public lands, mostly on the Cleveland National Forest (Stephenson and Calcarone 1999).

Cuyamaca cypress is known only from the Cuyamaca Peak/King Creek area in the mountains of San Diego County. The most narrowly distributed cypress in California, Cuyamaca cypress forms several groves that represent a single population encompassing 230 acres (93 hectares) on the Cleveland National Forest and in Cuyamaca Rancho State Park. An apparently introduced population occurs in the Agua Tibia Wilderness Area near Palomar Mountain (Stephenson and Calcarone 1999).

While Tecate cypress has a relatively broad geographic range, it is uncommon (Dunn 1987). The largest occurrences of Tecate cypress are found in Baja California Norte. Some large groves also occur on National Forest System lands, including a 50-acre (20-hectare) grove on Guatay Mountain in San Diego County and a 960-acre (389-hectare) grove in the Sierra Peak/Coal Canyon area of the northern Santa Ana Mountains. The Guatay Mountain grove is the smallest in California (Dunn 1987). The Sierra Peak/Coal Canyon groves are the northern limit of Tecate cypress distribution and its only Orange County locality. The largest stand of Tecate cypress in California encompasses more than 5,000 acres (2,023 hectares) but is located off of National Forest System lands at Otay Mountain along the border with Mexico (Stephenson and Calcarone 1999).

Tecate cypress occurs at elevations from as low as 65 feet (20 meters) in Baja California Norte to 4,200 feet (1,280 meters) in the San Diego Ranges and Santa Ana Mountains. Numerous groves of Tecate cypress occur on the Mexican side of Otay and Tecate Peaks and in the coastal mountains of Baja California Norte, extending about 150 miles (241 kilometers) south of the border. In the United States, 85% of Tecate cypress occurs on public lands (Stephenson and Calcarone 1999).

Ecological Processes: Like other plant communities on nutrient-poor soils, vegetation on gabbro soils is typically slower growing, more open, and lower in stature than vegetation on other soil types, and postdisturbance recovery can be slow. Because these communities typically occur within chaparral, fire is a significant ecological process influencing species composition and habitat structure on gabbro soils.

Like many native cypresses in California, Cuyamaca cypress and Tecate cypress are obligate seed reproducers following stand-replacing wildfires (Dunn 1987). Cuyamaca cypress and Tecate cypress have relatively thin, exfoliating bark that provides little protection from fire, and they are usually killed in wildfire events. They produce serotinous or closed cones at maturity. Serotinous cones require the intense heat generated by fire to open and disperse their seeds as well as to prepare the soil for enhancement of germination. Moreover, these species are shade intolerant and do not compete well with chaparral shrubs during the establishment phase (Vogl et al. 1988, Dunn 1987).

Factors that Influence Ecological Processes:

Fire: Cuyamaca and Tecate cypress stands have been reduced in size and extent on National Forest System lands in southern California by a combination of too-frequent fires in some stands and fire exclusion in others. While periodic fires are necessary for regeneration, short fire-return intervals decrease stand densities by preventing trees from reaching reproductive maturity and producing sufficient seed to reproduce the stand (Stephenson and Calcarone 1999). Because Cuyamaca cypress typically reaches maturity and begins to produce viable seed at approximately 40 years of age, a fire-return interval longer than 40 years is needed to maintain the seed pool and seed bank. Similarly, Tecate cypress trees begin cone production by about age 10 but require about 50 years for maximum cone production.

Shortened fire return intervals have reduced the size and extent of the Sierra Peak stand of Tecate cypress (Dunn 1987). In a fire history study, fire return intervals of at least 52 years in Tecate cypress groves resulted in the stands being fully replaced, while fires less than 33 years apart resulted in reduced stand density. However, substantially longer fire return intervals or fire exclusion may also adversely affect populations by reducing the amount of regeneration within a stand (Stephenson and Calcarone 1999).

Land Uses: Strip mining of gabbro-derived clay deposits has occurred on private lands and removed some Tecate cypress stands in the northern and eastern Santa Ana Mountains; there is potential for increased mining to occur (Stephenson and Calcarone 1999).

A recent potential threat to gabbro habitat on National Forest System lands is the construction of communication facilities on mountaintops. Other threats to gabbro habitats and vegetation in some areas include cattle grazing, unauthorized off-highway vehicle use, frequent fire occurrence, land development, and the invasion of nonnative undesirable plant species. Impacts on the habitat off National Forest System lands are largely unknown (Stephenson and Calcarone 1999).

Management Considerations: Gabbro outcrop plant communities are naturally small in size and have an inherently fragmented and disjunct distribution pattern. Because of unique soil and substrate conditions, plant species typical of gabbro habitats exhibit high levels of local adaptations and between-population genetic variability; accordingly, management or restoration of gabbro habitats should focus on conserving site-specific resources. Therefore, in situ conservation of gabbro plant communities and populations of gabbro-associated special-status species is important and should be a management priority.

Because Cuyamaca cypress has become increasingly rare on National Forest System lands, the Cleveland National Forest in 1991 established the King Creek Research Natural Area to protect the species and its habitat. The Cleveland National Forest has also developed a species management guide that summarizes approaches to fire management and provides guidelines for the use of fire in enhancing cypress stands on National Forest System lands (Stephenson and Calcarone 1999).

The Cleveland National Forest has also produced a species management guide for Tecate cypress. The guide summarizes approaches to fire management and provides guidelines for the use of fire in enhancing cypress stands on the Forest. The Guatay Mountain occurrence, which supports an estimated 64,000 trees, is proposed for designation as a Research Natural Area. A large portion (540 acres [219 hectares]) of the Coal Canyon population was recently purchased by the California Department of Fish and Game and designated as an ecological reserve.

There are 13 special-status plant species associated with gabbro habitats, including four that are state- or federally listed, and seven that are USDA Forest Service Region 5 Sensitive Species. All of these species occur or have suitable habitat on the Cleveland National Forest. The population status of six of these plants is considered to be in decline, four are stable, and two are unknown. They mostly are considered to have a low level of vulnerability on National Forest System lands.

Larvae of Thorn's hairstreak butterfly (*Mitoura thornei*), a federal species of concern, are found exclusively in association with Tecate cypress; however, the butterflies have only been observed in the grove on Otay Mountain off of National Forest System lands. Further research should be conducted on the floristic composition and population status of vegetation and associated special-status invertebrates on gabbro soils on the Cleveland National Forest (Stephenson and Calcarone 1999).

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Lakes and Reservoirs

Description: Aquatic habitats are found in several natural lakes and constructed reservoirs in the mountains of Southern California. Only a few small natural basins exist that hold water all or most of the time on the four southern California National Forests. Most occur along the San Andreas Fault; these include Jackson Lake, Elizabeth Lake, Lake Hughes, and Lost Lake. In addition, Crystal Lake in the San Gabriel Mountains, Dollar Lake in the San Bernardino Mountains, and Hidden Lake in the San Jacinto Mountains are small natural lakes.

Baldwin Lake is a large ephemeral lake in the eastern San Bernardino Mountains. This shallow basin fills with water during wet periods. It can retain water year-round for several years. When full, its shallow waters attract large numbers of waterfowl and also provide habitat for the endangered Shay Creek population of unarmored threespine stickleback fish (*Gasterosteus aculeatus williamsoni*). (Stephenson and Calcarone 1999, USDA Fish and Wildlife Service 2001).

The vast majority of lakes in the mountains of southern California are reservoirs behind dams. These reservoirs provide important habitat for several species of waterfowl, and most support popular recreational fisheries stocked with various species of bass, trout, catfish, and sunfish. Reservoirs also provide important winter foraging habitat for special-status raptors such as osprey (*Pandion haliaetus*) and bald eagle (*Haliaeetus leucophalus*) (USDA Fish and Wildlife Service 2001), but they provide only marginal habitat for amphibians and reptiles. Osprey have been reported from Lake San Antonio in Monterey County. Big Bear Lake supports an estimated 20-30 wintering bald eagles, while other reservoirs generally support 2-10 eagles between November and March each year (Stephenson and Calcarone 1999). Breeding pairs, however, have only been confirmed in a few locations around the Los Padres National Forest, such as Nacimiento Lake, San Antonio Lake, and Cachuma Lake (USDI Fish and Wildlife Service 2001).

Distribution and Abundance on National Forest System Lands and Adjacent Areas: Naturally occurring lakes in southern California are uncommon, small, and isolated habitats. Of the naturally occurring lakes, Jackson Lake, Elizabeth Lake, Lake Hughes, and Crystal Lake are located on the Angeles National Forest. Dollar Lake, Hidden Lake, Lost Lake, and Baldwin Lake occur on the San Bernardino National Forest. (Stephenson and Calcarone 1999).

Developed private property or lands managed by other agencies border most of the reservoirs in the mountains of southern California. However, some or all of the surrounding watershed areas are located on National Forest System lands. Reservoirs adjacent to National Forest System lands include the following:

Cleveland National Forest: Sutherland Lake, Lake Cuyamaca and Lake Henshaw;

San Bernardino National Forest: Lake Hemet, Big Bear Lake, Lake Arrowhead, Lake Gregory, Lake Fulmor, Jenks Lake, and Silverwood Lake;

Angeles National Forest: Morris Reservoir, San Gabriel Reservoir, Cogswell Reservoir, Pacoima Reservoir, Littlerock Reservoir, Castaic Lake, and Pyramid Lake; and

Los Padres National Forest: Lake Piru, Matilija Lake, Lake Jameson, Gibraltar Reservoir, Lake Cachuma, Twitchell Lake, Lopez Lake, Santa Margarita Lake, Lake San Antonio, and Nacimiento Lake.

Ecological Processes: Natural lakes in the mountains of southern California are small, isolated aquatic systems. Some, like Elizabeth Lake and Baldwin Lake, have highly variable water levels with dry periods, while some of the small, isolated lakes along the San Andreas Fault have more predictable water regimes. Although small in extent, these small lakes can be very important habitats for some reptiles and amphibians and important foraging habitat for some mammal and bird species.

Reservoirs are essentially distinct ecosystems, with an aquatic fauna that bears little resemblance to that naturally occurring in local streams or natural lakes. The water level in most reservoirs is managed for multiple uses, including water storage for municipal and agricultural uses, hydroelectric power generation, flood control, and recreation. Depending on the agency and the original purpose of construction, certain uses are given precedent over others. It is a common practice, for example, to maintain a high minimum water level during the popular summer recreation season, and then to draw down the reservoir to increase storage capacity for retaining runoff from winter precipitation. (Stephenson and Calcarone 1999.) Instream releases below reservoirs are generally maintained at some minimum level or at the level of inflow to the reservoir throughout the year.

Management Considerations: Because they are isolated and rare habitats, management of natural lakes should focus on the conservation of watershed health and the preservation of natural hydrologic regimes. However, the small size, easy accessibility, and popularity of small lakes for recreation may limit some conservation opportunities. In addition, demands for consumptive uses through groundwater pumping or upstream diversion may also limit conservation opportunities, especially when those uses occur within the watershed but are located on private lands.

Because water level management in reservoirs is controlled by other local, state, or federal resource management agencies, and because reservoirs are surrounded by private lands or lands managed by other agencies, management of National Forest System lands generally has little direct effect on reservoirs. However, management activities on National Forest System lands may indirectly affect them (Stephenson and Calcarone 1999).

Reservoirs are major recreational resources that draw many visitors to adjacent National Forest System lands. Fishing is a very popular recreational activity at reservoirs, which attract far more anglers than do streams. Some reservoirs managed for municipal water uses, however, such as **Morris, Cogswell, Big Tujunga, Pacoima, and Bouquet Reservoirs on the Angeles National Forest,** do not allow recreational fishing. (Stephenson and Calcarone 1999.)

The construction of reservoirs has increased habitat for some special-status species and removed habitat for others. For example, some reservoirs have inundated important montane meadow habitats and altered animal movement corridors. Water diversion and storage structures have also fundamentally altered downstream riverine aquatic and riparian habitats and impeded habitat connectivity to upstream habitats. Consequently, restoration of riverine aquatic and riparian habitats below dams should be integrated with reservoir management and with the activities of agencies and water users that are associated with each water project. (Stephenson and Calcarone 1999.)

Reservoirs have attracted special-status species, such as bald eagle and osprey, that were formerly very rare in the southern California mountains; effects on these species should be considered in the development and implementation of timber and recreation management activities (USDI Fish and Wildlife Service 2001). These reservoirs also, however, facilitate the introduction of a wide variety of nonnative undesirable aquatic species into the surrounding streams (Stephenson and Calcarone 1999).

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Gabbro Outcrops **Limestone / Carbonate Outcrops**

Limestone/Carbonate Outcrops

Limestone/Carbonate Outcrops

Description: Limestone and carbonate outcrops and soils derived from them are named for calcium carbonate, their primary chemical constituent. This substrate is derived from sedimentary rock type that weathers into alkaline limestone- and dolomite-derived soils. Carbonate soils support a variety of plant communities including blackbush scrub, pinyon-juniper woodlands, Jeffrey pine-western juniper woodlands, and Joshua tree woodlands. These vegetation types are described in other habitat accounts; here we focus on limestone and carbonate outcrops because of their distinct soil characteristics and high level of plant species endemism.

Carbonate habitats typically have shallow, rocky, coarse-textured soils, with limited nutrient availability. Productivity and total vegetative cover is low, and the flora contains many drought-tolerant plant species that can tolerate nutrient-poor soils. Consequently, plant communities on carbonate soils are typically more open, less productive, and slower growing than similar those growing on other soil types in the vicinity. A number of plants (listed below) are endemic to or strongly associated with carbonate soils (Stephenson and Calcarone 1999, USDI Fish and Wildlife Service 2001).

Distribution and Abundance on National Forest System and Adjacent Lands: Located mainly on the San Bernardino National Forest, carbonate habitats occupy an estimated 20,893 acres (8,455 hectares), of which 87% (18,177 acres) (7,356 hectares) are situated on public lands. Carbonate- and limestone-derived outcrops and soils are found in several east-west trending bands over approximately 20 miles (32 kilometers) of the northern desert-facing slopes in the San Bernardino Mountains. The outcrops occur from the White to Blackhawk Mountains and southeast to Tip Top and Onyx Peaks. Elevations range from 4,000–8,000 feet (1,219–2,434 meters). Disjunct carbonate outcrops occur just south of Sugarlump Ridge and to the east as far as the Sawtooth Hills. Carbonate habitats are also found outside of the eastern Transverse Ranges on the mountain ranges in the Mojave Desert and southwestern Great Basin. These areas support disjunct populations of several plant species associated with carbonate soils on the San Bernardino National Forest (Stephenson and Calcarone 1999, USDI Fish and Wildlife Service 2001).

Ecological Processes: The dominant plants on carbonate soils are drought-tolerant perennials with adaptations that allow them to persist on these dry, nutrient-poor soils. Information on the population dynamics and ecophysiology of many carbonate plants is lacking. In general, however, these species typically have slow growth and low stature on carbonate soils compared to plants on other soil types.

Carbonate habitats have are highly sensitive to ground disturbance and vegetation removal. Once disturbed, carbonate vegetation is slow to recover due to low plant productivity, thin, impoverished soils, and dry climate in this part of the San Bernardino Mountains. Because of low plant biomass, carbonate vegetation is far less likely to carry wildfire, to support intensive livestock grazing, or to support fuel and timber management activities. Like other forms of disturbance, wildfire and grazing can significantly delay the recovery of these plant communities (Stephenson and Calcarone 1999, USDI Fish and Wildlife Service 2001).

Factors that Influence Ecological Processes: Mining activities, such as the soil removal, road development, and dumping of overburden rock have led to an overall decline in the quantity and quality of carbonate habitat. Carbonate in these mountains is one of just three high-quality deposits in the western United States. As a result, the high-grade carbonate deposits in the San Bernardino Mountains have been mined for commercial use. Limestone is used in a number of commercial applications and most carbonate habitats on National Forest System lands are under mining claims that may become active in the future (Stephenson and Calcarone 1999).

Most direct losses of carbonate habitat associated with mine development occurred historically, but continued mining operations and associated road maintenance and overburden storage continue to affect carbonate plants and habitat. Perhaps the biggest potential effect of these operations is the loss of new populations of special status plants during mine reclamation activities (e.g., mine-tailings) (USDI Fish and Wildlife Service 2001).

Ongoing mining operations indirectly affect carbonate habitats through fugitive dust, changes to surface hydrology, soil erosion, and the invasion of nonnative undesirable plant species. Nonnative undesirable plant species can competitively exclude native plants. On the San Bernardino National Forest, Spanish broom (*Spartium junceum*), arundo (*Arundo donax*), yellow star-thistle (*Centaurea solstitialis*), tamarisk (*Tamarisk* spp.), and cheat grass (*Bromus tectorum*) have become established.

Cheat grass, the most widespread, has the ability to alter natural fire-related disturbance regimes by carrying fire into areas that would otherwise not ordinarily burn because of low fuel loads that are spatially discontinuous (USDI Fish and Wildlife Service 2001).

Approximately 16 miles of National Forest System roads cross or are adjacent to carbonate habitat on the San Bernardino National Forest. Unauthorized off-highway vehicles, mountain bikes, dispersed uses around developed facilities, and special-use permit activities have adversely affected some carbonate plant populations and habitats (Stephenson and Calcarone 1999).

Because of their naturally low productivity and open structure, carbonate habitats support little livestock grazing or timber and fuels management activities (Stephenson and Calcarone 1999). However, because of the prolonged recovery time required following disturbance on carbonate soils, the effects of such activities where they have occurred can persist long after the action is completed.

Management Considerations: Because carbonate habitats are restricted to a narrow geographic region and have a discontinuous distribution pattern, *in situ* conservation of carbonate habitats and populations of special-status species is recommended. In addition, some carbonate species exhibit local adaptations to edaphic conditions and exhibit between-population variability in these adaptations. Accordingly, both management and restoration of carbonate habitats must consider the edaphic needs of individual species.

Threats to carbonate/limestone habitats and their associated special-status plant populations include limestone mining, unauthorized off-highway vehicle use and, to a lesser extent, livestock grazing and transmission line construction. High levels of ground disturbance associated with mining, the potential for additional claims to become active, and insufficient protection mechanisms necessitated the federal listing in 1994 of five plant species endemic to carbonate soils. This listing created an unusual situation in which two powerful laws (the 1872 Mining Law and the 1973 Endangered Species Act) appear to conflict with one another (Stephenson and Calcarone 1999).

To resolve the cross purposes of these two laws, a collaborative effort involving the San Bernardino National Forest, the Bureau of Land Management, the USDI Fish and Wildlife Service, mine operators, researchers, and other affected parties, was initiated to develop a reserve system for listed carbonate plants and their habitat. This *Carbonate Endemic Plant Conservation Strategy* is being designed to incorporate many occurrences and populations of varying size of the five listed plants over a wide range of habitat mosaics throughout their geographic ranges. Buffers to minimize conflicts and to ensure defensible areas, corridors between populations and habitats, and inclusion of sites that contain multiple listed species are other important elements being considered in the reserve design. Studies on genetic variation, soils, and vegetation combined with field studies of individual plant occurrences are being completed; these efforts will provide information necessary for the reserve design. GIS mapping of plant locations, carbonate rock locations, approved and proposed future mining activities, land use conflicts, and other resource values have been developed in the *Conservation Study for Five Carbonate Plant Species: A Study of Land Use Conflict in the San Bernardino National Forest* (Stephenson and Calcarone 1999).

There are 10 special-status plants associated with carbonate soils, all restricted to the San Bernardino National Forest. These include four species federally listed as endangered, one federally listed as threatened, three on the USDA Forest Service Region 5 Regional Forester’s Sensitive Species list, and five that are endemic to the San Bernardino Mountains. Carbonate soil habitats have an inherently fragmented distribution, and special-status plants associated with this habitat type typically are distributed in small, isolated, or disjunct populations. Such distribution patterns increase the risks of extinction of local populations from natural or human-caused disturbances (Stephenson and Calcarone 1999, USDI Fish and Wildlife Service 2001).

Most carbonate habitats that support special-status plants on the San Bernardino National Forest have been affected by past mining activities or are considered to be at risk from potential mine development. The populations of six special-status plant species are declining, and all occurrences of special-status plants in carbonate habitats on National Forest System lands are considered to be highly vulnerable. Although the majority of occurrences of federally listed plants on carbonate soils are in undisturbed areas, 76% of the occurrences are located within mining claims or on private lands (Stephenson and Calcarone 1999, USDI Fish and Wildlife Service 2001).

The USDI Fish and Wildlife Service (2001) provided recommendations for conserving carbonate habitats and preserving federally listed plants during implementation of activities on the San Bernardino National Forest. Section 7(a)(1) of the Endangered Species Act (ESA) directs federal agencies to utilize their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Recommended conservation programs include activities carried out by discretionary agencies to avoid or minimize adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information. The conservation actions recommended by the USDI Fish and Wildlife Service (2001) for carbonate habitats and associated federally listed plants include:

Develop and implement regional, multi-species habitat conservation plans, particularly the West Mojave Habitat Conservation Plan.

Participate in any future habitat and species conservation programs in Big Bear Valley and other communities on the San Bernardino National Forest.

Develop and participate in the Carbonate Habitat Management Strategy (CHMS) and incorporate carbonate plant conservation efforts on Bureau of Land Management and private lands into this strategy. Upon release of the final recovery plan for the carbonate plants, incorporate recovery tasks, as appropriate, into the CHMS and other management strategies.

Endeavor to restore and protect all remaining carbonate habitats and associated physical features on the San Bernardino National Forest. The restoration program should include species and restoration ecologists to develop and to refine methods of restoring carbonate plant habitat on disturbed surfaces.

Emphasize the control and removal of all invasive, nonnative undesirable plants and animals to the maximum degree possible, especially nonnative undesirable grasses.

To the maximum extent possible, restrict unauthorized human presence and activities in areas occupied by carbonate plants, through patrols and other means.

Habitat Restoration: Many restoration activities in carbonate habitats have focused on the restoration and revegetation of mine tailings. The successful establishment of native vegetation is difficult because of poor soils and, in some cases, the lack of site-adapted seed material and plant stock. Reclamation of mine tailings typically involves moving large amounts of soil and recontouring of slopes. Some habitat improvement projects associated with surface mining reclamation activities have caused the loss of some special-status plants; generally, though, such projects have resulted in the enhancement or expansion of carbonate habitats. Some past restoration projects have used undesirable nonnative plant species. As more reclamation projects are completed, an increasing knowledge base will facilitate the planning and design of future restoration projects on carbonate soils (USDI Fish and Wildlife Service 2001).

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Lower Montane Forest

Description: Lower montane forest occurs as a discontinuous vegetation type in a transition zone from foothill chaparral to montane conifer habitats mainly between 3,000 and 5,500 feet (914 and 1,676 meters), but it extends to higher and lower elevations on some exposures. This habitat type occurs on the Santa Lucia Ranges and the coastal slopes of the Transverse and Peninsula Ranges. Stands of lower montane forests and woodlands grow in small, scattered patches of roughly 50-800 acres (20-325 hectares), often restricted to specific topographic settings, surrounded by large expanses of chaparral (Stephenson and Calcarone 1999).

Lower montane forests are often described as mixed evergreen forests because they are dominated by a combination of needle-leaf conifer and broadleaf angiosperm evergreen trees. Bigcone Douglas-fir (*Pseudotsuga macrocarpa*), Coulter pine (*Pinus coulteri*), canyon live oak (*Quercus chrysolepsis*), coast live oak (*Q. agrifolia*), and black oak (*Q. kelloggii*) are the primary lower montane tree species in the Transverse and Peninsula Ranges. Other tree species, especially in the Santa Lucia Ranges, include foothill pine (*Pinus sabiniana*), madrone (*Arbutus menziesii*), California bay (*Umbellularia californica*), tanoak (*Lithocarpus densiflorus*), Santa Lucia fir (*Abies bracteata*), Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*), ponderosa pine (*Pinus ponderosa*), and knobcone pine (*P. attenuata*). Understory shrubs include common chaparral species (Barbour 1988, Stephenson and Calcarone 1999).

Coulter pine is a major component in the lower montane forests at 3,950-5,900 feet (1,200-1,800 meters), where it typically forms a canopy with 40-65% cover with little shrub and herb layer (Thorne 1988, Barbour 1988). In the Transverse and Peninsular Ranges, Coulter pine is often associated with canyon live oak; it mixes with chaparral at lower elevations and with ponderosa pine and black oak at higher elevations. Stands occur on all aspects, including south-facing slopes and disturbed sites (Barbour 1988, Vogl et al. 1988, Stephenson and Calcarone 1999).

Stands of bigcone Douglas-fir and canyon live oak typically occur on mesic sites such as shaded canyons and draws and steep north- and east-facing aspects. On moist sites, bigcone Douglas-fir and canyon live oak occur with big-leaf maple (*Acer macrophyllum*), black cottonwood (*Populus balsamifera* ssp. *trichocarpa*), and California bay. At the upper end of its elevational range this forest type occurs on all aspects and becomes less restricted to canyons (Barbour 1988, Stephenson and Calcarone 1999). Bigcone Douglas-fir and Coulter pine rarely co-occur, but both are frequently associated with canyon live oak.

Lower montane forest in the northern Santa Lucia Range reaches as low as to 1,000 feet (305 meters) because of greater precipitation and the moderating affects of the coastal climate. Forests and woodlands are characterized by 50-80% canopy cover and a canopy height of 50-100 feet (15-30 meters). Common dominant tree species include a greater mix of broadleaf angiosperm species such as California bay, Pacific madrone, coast live oak, and canyon live oak, with big-leaf maple common in draws and northern slopes. The needle-leaf conifer species mix is unique in southern California, comprising Santa Lucia fir, Douglas-fir, ponderosa pine, foothill pine, and Coulter pine. In some stands conifers add an additional, higher canopy layer up to 100-200 feet (30-60 meters) in height and more than 50% cover. This is the only place on the four southern California National Forests where Douglas-fir and madrone are common trees; Santa Lucia fir is endemic to the northern Santa Lucia Range (Barbour 1988, Thorne 1988, Stephenson and Calcarone 1999).

Small, disjunct stands of knobcone pine occur in a transitional zone between chaparral and lower montane woodlands and higher elevation montane conifer forest (Vogl et al. 1988). Knobcone pine is usually restricted to dry, rocky sites with shallow soils and infertile substrates that limit competition from other conifers in southern California mountains (Stephenson and Calcarone 1999).

Distribution and Abundance on National Forest System and Adjacent Lands: Lower montane mixed evergreen and conifer forests have discontinuous distribution and variable species composition in southern California. For example, on the four southern California National Forests, Douglas-fir, ponderosa pine, foothill pine, and mixed broadleaf evergreen forests are largely restricted to the Santa Lucia Ranges; Santa Lucia fir is endemic to the northern Santa Lucia Range (Stephenson and Calcarone 1999). By contrast, canyon live oak is the most widely distributed oak in California (Sawyer et al. 1988).

Bigcone Douglas-fir is restricted to the Transverse and Peninsular Ranges. Coulter pine is distributed sporadically throughout the lower montane habitats of southern California, with some occurrences extending into the Sierra Juarez and Sierra San Pedro Martir of Baja California Norte (Vogl et al. 1988, Thorne 1988). Knobcone pine occurs in just a few locations, including the northern Santa Lucia Range in Monterey County, Cuesta Pass in San Luis Obispo County, the western San Bernardino Mountains between City Creek and Government Canyon, and the slopes of Sugarloaf, Pleasants, and Santiago Peaks in the Santa Ana Mountains on (Thorne 1988). Knobcone pine is more common throughout northern California; it also grows in the foothills of the Baja California Norte mountains (Vogl et al. 1988).

Public lands on and adjacent to the four southern California Forests encompass a total of 272,929 acres (110,451 hectares) of lower montane conifer and mixed evergreen forests and woodlands. This area includes 81,018 acres (32,787 hectares) of bigcone Douglas-fir and canyon live oak, 64,091 acres (25,937 hectares) of Coulter pine and canyon live oak, 108,759 acres (44,013 hectares) of canyon live oak, 5,171 acres (2,093 hectares) of foothill pine and knobcone pine, 17,917 acres (7,251 hectares) of California bay forests, and 1,844 acres (746 hectares) of other mixed broadleaf evergreen forests (Stephenson and Calcarone 1999).

Ecological Processes: The ecological relationships of lower montane forests vary by stand composition, especially in relation to soil moisture and fire. Most lower montane forests occur in areas with dry, hot summers and occasional winter snows but little to no accumulation. Broadleaf trees with sclerophyll leaves and drought-tolerant conifers dominate the lower montane forests. Bigcone Douglas-fir and many broadleaf species are more dominant on sites with greater soil moisture availability (Barbour 1988, Thorne 1988).

Bigcone Douglas-fir and canyon live oak forests occur on steep, fire-resistant terrain such as moist, shaded canyons and northern aspects; these are generally fire refugia in the chaparral landscape (Minnich 1988). Bigcone Douglas-fir and canyon live oak stands also typically have few understory shrubs or flammable ground cover because of dense overstory canopy shade; this sparse understory reduces the ability of the stand to carry a wildfire (Barbour 1988, Thorne 1988). By contrast, closed-cone pines such as knobcone and Coulter are dependent on fire for reproduction, and often occur on broad, exposed west- and south-facing slopes (Vogl et al. 1988).

Factors that Influence Ecological Processes: Fire is the most important disturbance event in lower montane habitats. Because many lower montane and mixed evergreen stands occur as small patches within broad areas dominated by chaparral, the fire regime and effects of fire in these forest types is largely controlled by chaparral fire regimes. For example, fires seldom start in bigcone Douglas-fir stands but rather carry into them from the surrounding chaparral (Minnich 1988).

The primary fire issue in lower montane habitats is the apparent loss of bigcone Douglas-fir in stand-replacing wildfires. Bigcone Douglas-fir forests have been slow to recover from crown fires and many stands show little or no postfire regeneration. An estimated 18% reduction in bigcone Douglas-fir/ canyon live oak forests has occurred in the San Bernardino Mountains because of these species' high mortality in wildfires and poor regeneration following a stand-replacing fire (Stephenson and Calcarone 1999).

Recent changes in fire patterns and behavior, including more frequent and intense crown fires started in surrounding chaparral habitats, may be a cause of the observed decline in bigcone Douglas-fir forests during the twentieth century (Minnich 1988). Although quantitative data are lacking, increasing fire intensity may be a result of increased ignitions in urban interface environments during extreme fire weather conditions such as hot summer days and Santa Ana wind conditions. These weather conditions are unlikely to have been associated with natural wildfire occurrences. Importantly, these fires normally start at the base of the mountains and are prone to high-intensity runs up steep lower montane slopes and canyons where bigcone Douglas-fir occurs (Stephenson and Calcarone 1999).

Consistent with the idea that bigcone Douglas-fir occurs on fire refugia such as steep, north-facing slopes, the stands most vulnerable to wildfires occur on gentle slopes. One study observed 37% survival of bigcone Douglas-fir following wildfires on slopes of less than 20 ° , but more than 90% survival on slopes of more than 40 ° . Although bigcone Douglas-fir has the ability to resprout from the crown or base, resprouting is less likely following an intense crown fire or on mature trees. Seedlings of bigcone Douglas-fir can germinate in mineral soil, but generally require some tree canopy shade for successful establishment. Consequently, if a stand is lost in a crown fire, regeneration of the stand may require that canyon live oak become established first, after which viable seeds must be distributed to the site from disjunct stands (Stephenson and Calcarone 1999).

Coulter pine, in contrast to bigcone Douglas-fir, exhibits classic fire-adapted traits. It reproduces primarily after periodic crown fires; it has a relatively short life span (50-100 years); the seedlings thrive in full sunlight; and it bears semi-serotinous cones that open when burned. As a result, Coulter pine is more resilient than bigcone Douglas-fir to the current chaparral fire regime. The biggest threats to Coulter pine are multiple fires in short succession (e.g., less than 25 years) and complete fire exclusion. Multiple short-return fires can kill overstory trees before an adequate seed crop has developed, while complete fire exclusion can preclude regeneration (Sawyer et al. 1988,Thorne 1988). Like Coulter pine, knobcone pine bears serotinous cones and is dependent on fire for seed dispersal and regeneration (Vogl et al. 1988).

When overstory Coulter pines become senescent and die without an accompanying fire to prepare a mineral soil seed bed, seedling establishment can be impaired. For example, during the height of the drought in the late 1980s, a major bark beetle epidemic killed approximately 70% of overstory Coulter pines on Palomar Mountain. Reestablishment has been poor in some areas due to competition and shading from chaparral. In such cases of short fire return intervals or fire exclusion, Coulter pine stands can fail to regenerate and convert to chaparral (Stephenson and Calcarone 1999).

Although fires are generally smaller and less frequent in the relatively mesic northern Santa Lucia Range, the Marble Cone fire burned 180,000 acres (72,844 hectares) in the late 1970s. It is unclear if large but infrequent fires represent a typical historic pattern, but there is little evidence to suggest that the area is experiencing changes to lower montane conifer and mixed evergreen vegetation as a result of shifting fire regimes (Stephenson and Calcarone 1999). Many of the common tree species in the lower montane forests of the northern Santa Lucia Range, including Douglas-fir, Santa Lucia fir, tanoak, madrone, and California bay, are at least moderately shade tolerant and do not regenerate well following fire. This suggests that this community is not dependent on fire for reproduction, or that fire is a very infrequent disturbance event (Sawyer et al. 1988).

Management Considerations: The primary management considerations for lower montane forests and woodlands relate to fire occurrence and wildlife habitat values. The desired condition for fire management in bigcone Douglas-fir and canyon live oak forests is to minimize occurrence of high-severity crown fires. This condition is necessary to maintain or increase the distribution and extent of bigcone Douglas-fir and canyon live oak forest stands (Stephenson and Calcarone 1999).

The desired condition for fire management in Coulter pine forests is fire return intervals of approximately 60-75 years. Coulter pine requires fire for regeneration, but stands can be adversely affected if fire occurs prior to stands reaching maturity or if fire is completely excluded. At maturity, stands should be fairly open (40-60% canopy closure) with trees reaching 20 inches (51 centimeters) in diameter measured at a height of 4.5 feet (1.5 meters) (Stephenson and Calcarone 1999). The isolated, disjunct stands of knobcone pine should have similar fire management considerations as those of Coulter pine, with fire intervals adjusted for the time required for stands to reach reproductive maturity.

The fire relationships of lower montane conifer and broadleaf mixed evergreen forests in the Santa Lucia Range are not well understood. In general, these forest types do not appear to require fire for regeneration. Much of the northern Santa Lucia Range has rugged terrain with limited road development, and most of the lower montane habitats are within the Ventana Wilderness Area. Fire management, therefore, is largely restricted to suppression activities of human-caused ignitions in the few areas with road access. However, further research should be conducted prior to the recommendation of specific fire management actions.

The lower montane forests on the four southern California National Forests are distinctive in terms of species composition and the degree of natural fragmentation; many stands are largely restricted to specific topographic settings. Accordingly, they provide forest and woodland structure within an otherwise chaparral-dominated landscape, and form an important transition between chaparral and montane habitats. They provide a mosaic of cover, perches, and foraging opportunities for many wildlife species. Consequently, the *in situ* conservation of many of these stands is important for managing wildlife habitats and movement corridors.

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Montane Conifer Forest

Description: Montane conifer forests are habitats dominated by varying combinations of ponderosa pine (*Pinus ponderosa*), Jeffrey pine (*P. jeffreyi*), white fir (*Abies concolor* var. *lowiana*), black oak (*Quercus kelloggii*), canyon live oak (*Q. chrysolepis*), sugar pine (*P. lambertiana*), incense cedar (*Calocedrus decurrens*), and western juniper (*Juniperus occidentalis* var. *occidentalis*). Mature montane conifer forests typically have a canopy height up to 100-200 feet (30-60 meters), 50-80% canopy cover, and trees up to 40 inches (1 meter) in diameter at breast height (dbh) (4.5 feet [1.5 meters] above ground). Grass and herbaceous cover averages 5-10% (Barbour 1988). Drier slopes and transmontane areas often have smaller stature trees (40-65 feet [12-20 meters]) and more open canopies (20-40% cover), and are often described as "parklike" stands (Thorne 1982, 1988; Barbour 1988; Vasek and Thorne 1988). Montane conifer habitats are also called midmontane conifer and mixed conifer (Thorne 1982).

Ponderosa pine forest is the most common conifer forest type at 5,000-6,900 feet (1,524-2,100 meters) on cismontane slopes; at its lower limits it mixes with lower montane forest types. Above 6,900 feet (2,100 meters) Jeffrey pine is more dominant. More tolerant of drought and cold temperatures than ponderosa pine, Jeffrey pine replaces ponderosa pine at all montane elevations on the transmontane slopes. Ponderosa pine and Jeffrey pine co-occur and hybridize where their range overlaps on cismontane slopes, and they can be difficult to distinguish.

Both ponderosa and Jeffrey pine can also form open woodlands with less than 50% canopy cover, especially on gentle slopes and drier south- and west-facing exposures. Understory shrubs are common in openings and canopy gaps, and include manzanita (*Arctostaphylos* spp.), mountain mahogany (*Cercocarpus ledifolius*), sagebrush (*Artemisia tridentata*), bitter cherry (*Prunus emarginata*), western chokecherry (*Prunus virginianus* var. *demissa*), mountain whitethorn (*Ceanothus cordulatus*), and snowberry (*Symphoricarpos* spp.). Canyon live oak and black oak can also be common understory components (Thorne 1982, Barbour 1988). On lower transmontane slopes, Jeffrey pine woodland merges with pinyon-juniper woodlands.

White fir and sugar pine stands occur on mesic and steep slopes, especially on north- and east-facing aspects. In the San Gabriel and San Bernardino Mountains, white fir and sugar pine between 5,500-9,800 feet (1,675-2,590 meters). Incense cedar is a frequent component of these stands, and common understory shrubs include species of currant and gooseberry (*Ribes* spp.), snowberry, blue elderberry (*Sambucus caerulea*), and thimbleberry (*Rubus parviflorus*). (Thorne 1982.)

The northern Santa Lucia Ranges and the San Diego Ranges have depauperate, marginal montane conifer habitat. In the northern Santa Lucia Ranges montane conifer stands consist of canyon live oak, sugar pine, Santa Lucia fir (*Abies bracteata*), and Douglas-fir (*Pseudotsuga douglasii*) (Stephenson and

Calcarone 1999). Montane conifer forests have patchy distribution on the San Diego Ranges. Relatively dense stands of ponderosa pine, white fir, and sugar pine occur on mesic slopes in the Cuyamaca Mountains and Palomar Mountains. The drier Laguna Mountains have a more transmontane affinity and support more open stands of Jeffrey pine, black oak, and canyon live oak (Beauchamp 1986).

Distribution and Abundance on National Forest System and Adjacent Lands: Montane conifer forests are the dominant land cover type at 5,000-8,500 feet (1,524-2,590 meters) in southern California and above 1,000 feet (305 meters) in the northern Santa Lucia Range. This habitat type is absent from the southern Santa Lucia Ranges because of their low elevations, but occurs on the higher portions of the Western Transverse Ranges on the southern Los Padres National Forest (Stephenson and Calcarone 1999). The approximate elevational ranges of montane conifer forests are 5,700-8,900 feet (1,750-2,700 meters) on Mount Pinos, 4,600-7,900 feet (1,400-2,400 meters) in the San Gabriel Mountains, 5,250-8,500 feet (1,600-2,600 meters) in the San Bernardino Mountains, and 4,800-8,200 feet (1,450-2,500 meters) in the San Jacinto Mountains (Barbour 1988).

There are a total of 378,414 acres (153,139 hectares) of montane conifer habitats on public lands on or adjacent to the four southern California National Forests. This total comprises 127,592 acres (51,635 hectares) of pine-dominated mixed conifer, 119,233 acres (48,252 hectares) of white fir-dominated mixed conifer, 118,947 acres (48,136 hectares) of ponderosa and Jeffrey pine forests, and 12,642 acres (5,116 hectares) of black oak stands. More than half of this habitat is on the San Bernardino National Forest, and about one-fifth is located in the San Gabriel Mountains and southern Los Padres Ranges on the Angeles and Los Padres National Forests, respectively (Stephenson and Calcarone 1999).

Ecological Processes: Species dominance in the montane conifer forests is generally a function of hydroclimatic relationships and the disturbance regime. In general, white fir and sugar pine are more common on mesic sites and steep northern and eastern slopes. Jeffrey pine is more dominant on drier transmontane areas and higher elevations, while ponderosa pine is more dominant at lower elevations on cismontane slopes.

Historically, many montane conifer stands were dominated by large-diameter, widely spaced pines interspersed with dense patches of multi-layered old-growth stands and open patches. Patchy, low- to moderate-intensity understory fires were frequent occurrences and maintained the structure and species composition of these forests, with fire return intervals averaging 15-30 years. Frequent fires maintained open understories, reduced the density of shade- tolerant species such as white fir, and maintained the open structure of stands. These conditions favored recruitment and retention of shade intolerant, long-lived, fire-resistant pine species. (Minnich 1988.)

Factors that Influence Ecological Processes: Montane conifer forest structure, species composition, and ecosystem processes have changed dramatically since the early 1900s because of fire suppression and commercial timber harvests, and to a lesser extent because of air pollution and other stressors such as drought, diseases, and bark beetle infestation. There has been substantial increases in the number of small-diameter and understory trees, especially shade-tolerant species such as white fir and incense cedar, which have colonized the understory of ponderosa pine and Jeffrey pine stands (Minnich 1988). There have also been reductions in the number of large canopy dominant trees, and shifts in species composition towards more white fir and incense cedar and fewer Jeffrey pine, ponderosa pine, and black oak. By comparison, however, Jeffrey pine forests on the drier, transmontane slopes have remained relatively stable in terms of species composition and stand structure (Stephenson and Calcarone 1999).

Fire: Fire suppression has been very effective in montane conifer forests in the last 90 years. Understory fires have been virtually eliminated from large areas, particularly in interior forest areas, and crown fires have been rare (Minnich 1988). Currently, about 66% of montane conifer forests have no recorded fires in the modern era, and 88% of this habitat type has not burned in the last 40 years (Stephenson and Calcarone 1999).

Over the last 50 years, there have been very few "forest" fires in southern California's montane conifer region. Fires that have occurred were either driven primarily by steep terrain and extreme winds (e.g., the 1980 Thunder Fire in the San Gabriel Mountains) or took place in forest stands that were surrounded and interspersed with mature chaparral that facilitated the spread of high-intensity fires into them (e.g., the 1950 Conejos and 1970 Laguna Fires in the Cuyamaca and Laguna Mountains, the 1970 Bear Fire in the San Bernardino Mountains). There are few instances where the severity of these fires was attributed primarily to increased fuel loads in overcrowded forests. (Stephenson and Calcarone 1999.)

Fire exclusion has resulted in a dramatic increase in the number of understory trees, especially shade-tolerant white fir and incense cedar. The increased stand density has indirectly resulted in substantial changes to ecosystem patterns and processes. There is increased tree mortality and disease, and reduced recruitment of large trees because of increased understory competition. Importantly, under current stand conditions, fires behave differently than under natural fuel conditions, resulting in different effects on species composition and succession. There is increased risk of stand-replacing crown fires because of large fuel accumulations, fuel ladders, and standing dead or dying trees. Because of high fuel loads, fires tend to be intense and result in more complete burns. In some places hydrophobic soils result, and revegetation is slow. (Minnich 1988.)

Despite the relative lack of large, stand-replacing wildfires in southern California conifer forests, fires on other National Forests with similar recent changes in the fuel profiles of montane conifer forests resulting from fire exclusion and logging give cause for concern. On the Boise National Forest in Idaho, for example, many years of near-complete fire exclusion came to an abrupt end when a series of intense crown fires from 1986 to 1995 consumed 45% of the area's ponderosa pine forest. The severity of these fires was attributed to excessive fuel loading resulting from the prolonged absence of fire. Similar large and unusually intense stand-replacing wildfires have occurred in recent years in the Sierra Nevada and the mountains of northern Arizona (Stephenson and Calcarone 1999).

The greatest risks of catastrophic fire are primarily in mesic forests where understory trees develop rapidly. Stand densities in drier transmontane forests are not experiencing fuel accumulations and understory tree densities as substantial as those on cismontane slopes. The USDA Forest Service developed a predictive, GIS-based spatial model to estimate the amount of area likely to have overcrowded forest conditions and associated crown fire risk. The model suggests that almost 30% (108,500 acres [43,909 hectares]) of montane conifer stands on the four southern California National Forests are predicted to be overstocked by understory trees. (Stephenson and Calcarone 1999.)

The increase in stand density, ladder fuels, and fuel loading makes the forests more susceptible to large, stand-replacing crown fires that cannot be controlled or contained by active suppression. Where these conditions occur in wildland-urban interface areas, there are high wildfire risks to life and property.

Timber and Fuelwood Harvest: Most of the land base suitable for commercial timber harvest in southern California is on the San Bernardino National Forest. Localized timber harvesting began with European settlement, and an active timber program was sustained on the San Bernardino National Forest from the late 1940s through the mid-1980s (Minnich 1988). Harvest levels peaked in the 1960s, with 27.4 million board feet (MMBF) harvested on the San Bernardino National Forest in 1963 alone. A total of 362.3 MMBF were harvested in Los Angeles and San Bernardino Counties during this time. This total includes some timber harvested from the San Gabriel Mountains on the Angeles National Forest, but the vast majority came from the San Bernardino National Forest (Stephenson and Calcarone 1999).

A small number of commercial harvests also took place on the Los Padres and Cleveland National Forests during the peak years in the 1960s and 1970s. The small volumes produced in these areas were not sufficient to support economically viable sawmill operations, and timber had to be trucked long distances to the nearest mill. The same problem ultimately affected timber harvest operations on the San Bernardino National Forest. The forest timber program ended when dropping harvest levels in the late 1970s led to closure of the last mill in the area. (Stephenson and Calcarone 1999.)

Recent small timber programs on the four southern California National Forests have focused on forest health issues (i.e., treatment of insect and disease centers, understory thinning, and fuels reduction); administering individual permits to accommodate local demand for fuelwood; and identifying and removing hazard trees. Small-scale salvage operations to remove trees killed by wildfire or bark beetles are also occasionally undertaken. (Stephenson and Calcarone 1999.)

In analyzing the effects of timber harvesting on montane conifer forests, historic commercial harvests and current forest health programs have dramatically different effects on stand structure, species composition, and ecosystem processes. Past timber harvesting, particularly in the San Bernardino Mountains and San Jacinto Mountains, resulted in a substantial reduction in the number of large, healthy, and long-lived trees, with ponderosa and Jeffrey pines incurring the greatest reductions; it also contributed to the increased dominance of white fir. (Stephenson and Calcarone 1999.)

By contrast, current forest health and thinning projects remove smaller, diseased, and understory trees, while retaining larger, long-lived, healthy trees. Current timber management accordingly results in increased average tree size; stands that are more resilient to diseases, drought, and wildfire; and forests that are more likely to develop late-successional/old-growth characteristics.

In addition to changes in forest structure and species composition, past timber harvesting and current fuelwood harvesting, salvage, and forest health practices reduce the availability of coarse woody debris, downed logs, snags, and other important habitat elements for cavity-nesting birds, many small mammals, fungi, and other decomposition organisms (Stephenson and Calcarone 1999). Current snag retention policies are considered adequate for providing snag habitat across montane conifer forests. With the overall younger and smaller size of trees in commercially harvested stands, however, the abundance and quality of downed logs and coarse woody debris may be lacking in some areas (Stephenson and Calcarone 1999).

Air Pollution: Several long-term studies have examined the effects of air pollution on montane conifer vegetation in the San Bernardino and, to a lesser extent, the San Gabriel Mountains. Prevailing climatic conditions transport most of the air pollution from the South Coast Air Basin into the eastern Transverse Ranges and northern Peninsular Ranges. Eastern Los Angeles County and western San Bernardino and Riverside Counties receive the highest concentrations of air pollutants. In the mountains, the coastal side of the eastern San Gabriel Mountains and western San Bernardino Mountains are the most polluted. (Stephenson and Calcarone 1999.)

The two depositional components of air pollution that have the greatest effect on ecosystems are ozone, a powerful oxidant, and nitrogen oxides (NOx). Ponderosa and Jeffrey pine trees, and to a lesser extent bigcone Douglas-fir, are susceptible to damage from ozone. Symptoms include leaf discoloration, slow growth, and decreased stature (Barbour 1988). Other tree species are considerably more tolerant of ozone. Chronic ozone injury to ponderosa pines was first identified in the San Bernardino Mountains in the 1950s. Mortality and damage of ponderosa pine and Jeffrey pine peaked with high ozone concentrations in the 1970s, and has declined with improving air quality since 1976 (Stephenson and Calcarone 1999).

Ozone damage also renders trees more vulnerable to other stressors, such as drought and bark beetle infestations. Pine mortality has been highest during extended droughts. Trees with chronic ozone injury enter periods of drought without the energy reserves required to withstand bark beetle infestations. (Stephenson and Calcarone 1999.)

In the San Bernardino Mountains, there is a clear west-to-east gradient in both ozone levels and tree damage. Forests on the western side of the range are exposed to much higher levels of ozone and are experiencing the most damage. Monitoring of study sites in the westside forests over a 14-year period (1974-1988) found that ponderosa and Jeffrey pine are losing basal area in relation to competing species that are more tolerant of ozone; namely white fir, incense-cedar, and black oak. The accumulation of more ozone-tolerant understory species creates fuel ladders that increase risks of catastrophic losses from wildfire (Stephenson and Calcarone 1999).

Quantitative information on ozone damage is scarce for other mountain ranges in southern California. However, given the location of major pollution sources and prevailing wind patterns, areas of high potential for damage are probably confined to the eastern San Gabriel Mountains and western San Bernardino Mountains (Stephenson and Calcarone 1999).

The effects of NOx deposition are not as well documented as those of ozone deposition. High rates of NOx deposition have increased soil fertility and surface litter decomposition rates in some montane conifer forests in the San Bernardino Mountains. However, excessive NOx inputs can lead to various negative effects, including nutrient deficiencies; soil acidification; altered species composition; decreases in mycorrhizal root symbiosis; and elevated concentrations of nitrate in soil, groundwater, and streams. In montane conifer forests, the fertilizing effect of NOx may be accelerating understory development of white fir and incense-cedar, thereby elevating fuel loads and potentially increasing the risk of stand-replacing fires. (Stephenson and Calcarone 1999.)

Management Considerations: The management considerations discussed below will facilitate the improvement of montane ecosystem health, habitat diversity, and habitat connectivity. The management of montane conifer forests should focus on creating conditions favorable to the following: development of late-successional/old-growth forest characteristics; stands that are resistant to crown fires and disease outbreaks; increasing or maintaining habitat connectivity between stands; the reintroduction of low-intensity fire as a management tool; and the establishment of defensible fuel profile zones (DFPZs) in wildland-urban interface areas, ridgelines, and road margins. Management should also focus on creating a continuous and well-distributed supply of large trees. Large trees are characteristic of forests prior to fire suppression and logging, as well as being vital to many wildlife species. They have declined in number and are further threatened by increasing stand densities, which increases competition and risk of crown fire.

The relationships between existing montane conifer forest health, density, habitat quality, and fuel conditions indicate that fire and fuel management is an integral and necessary part of any conifer forest management program. The establishment of a DFPZ network allows a broader range of forest management practices to occur and reduces the risks of catastrophic losses from uncontrolled wildfires. A DFPZ network facilitates fire and fuel management as well as the restoration and enhancement of montane conifer forest habitat quality and connectivity.

Fire and Fuel Management Considerations: Options for managing fire or mimicking its ecological function fall into three general categories (Stephenson and Calcarone 1999):

Prescribed burning where a fire is intentionally set under controlled conditions to manage fuels and ecosystem health;

Fuel treatments where vegetation is manually or mechanically cut to reduce fire behavior or to mimic the structural effect that a low-intensity understory fire would have; and

Wildfire suppression strategies that include confine-and-contain approaches, where an unplanned or natural ignition is allowed to burn within a predefined containment area.

An important preliminary step to managing for more natural fire regimes, supporting confine-and-contain approaches, and conducting prescribed burns is the creation of a network of DFPZs. The DFPZ is a fuels treatment option being implemented throughout the western United States for reducing crown-fire hazards in conifer forests.

A DFPZ is a low-density fuel zone averaging 0.25 mile (0.4 kilometer) in width, located mostly along roads, ridgelines, and wildland-urban interface areas. DFPZs are designed to support suppression activities and can be likened to shaded fuel breaks, only on a larger scale. The focus of DFPZs is to break up fuel continuity over landscapes. A network of DFPZs is expected to reduce wildfire severity in treated areas, create broad zones where suppression efforts can be conducted safely and effectively, reduce fuel continuity, and become anchor lines for future fuel treatments (USDA Forest Service 2001).

In general, montane conifer stands that can be managed with low-intensity prescribed burns have fuel characteristics similar to those of DFPZs. Treated stands typically have approximately 40-50% canopy cover of trees at least 20 inches (45 centimeters) dbh and understories with little shrub cover and ladder fuels (USDA Forest Service 2001).

Under existing conditions, heavy fuels create a high potential for wildfires that kill a large fraction of overstory trees. Consequently, forest density and vertical and horizontal fuel continuity problems must be treated at a landscape scale (e.g., watersheds) to avoid large stand-replacing fires. Treatments include mechanical thinning and hand thinning of small-diameter trees and understory shrubs. After thinning, fuel conditions may allow the use of prescribed fire to further reduce fuels in stands or to reduce understory fuel loads in the future (USDA Forest Service 2001).

Subsequent thinning projects or prescribed understory burning should occur in each forest stand every 20-40 years, depending on rates of fuel accumulation. Some mortality of overstory trees during these events (2-4% of the stand) is desirable to create small openings for pine and oak regeneration and for snag recruitment (Stephenson and Calcarone 1999).

Away from developed areas, DFPZs should be placed primarily on ridges and upper south- and west-facing slopes. Where possible they should also coincide with existing roads to simplify construction and maintenance and to facilitate their use by suppression forces. This treatment will result in a fairly open stand, which is dominated by large trees of fire-tolerant species. Post-treatment canopy closure is recommended to be no more than 40% (Stephenson and Calcarone 1999).

One of the key features of the DFPZ approach is the general openness and the horizontal and vertical discontinuity of crown fuels to ensure a very low probability of sustained crown fire. Reducing canopy closure to levels below 40%, however, can be counterproductive in the long run, because such a condition may allow sufficient sunlight at the forest floor to encourage understory shrub growth. The establishment of a dense shrub layer in the understory creates a new fuel management problem and prevents tree recruitment (USDA Forest Service 2001).

In conclusion, the creation of DFPZs is generally consistent with forest health management. The creation of DFPZs favors long-lived, large-diameter, fire-resistant pine species; reduces understory competition and stand density; facilitates the reintroduction of low-intensity surface fires, and reduces the risks of catastrophic wildfire losses (USDA Forest Service 2001).

Special-Status Species: There are 30 special-status plant species, including 16 USDA Forest Service Region 5 Regional Forester's Sensitive Species, associated with montane conifer habitats on the four southern California National Forests. Most (25) of these species occur or have potential habitat on the San Bernardino National Forest. The population trends and status of most (21) of these taxa are unknown, but these plant species generally have low or moderate vulnerability because of the low level of land and resource use pressures in these habitats (Stephenson and Calcarone 1999).

There are 25 special-status wildlife species associated with montane conifer habitats: three invertebrates, six reptiles, nine birds, and seven mammals. Most (22) of these species, and all three invertebrates, occur or potentially occur on the San Bernardino National Forest. In general, montane conifer species have low to moderate vulnerability; the greatest vulnerability is a result of increasing private land development and historic logging practices. Very little information on the distribution or size of existing populations is known, and the population trends of all 25 special-status wildlife species is not known (Stephenson and Calcarone 1999).

Surveys should be conducted to map the extent of occupied habitats of these special-status species and to collect baseline population information in order to determine trends in population sizes and distribution and to identify appropriate responses to habitat modification.

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Montane Meadows

Description: Montane meadows are grass- and herb-dominated vegetation communities within lower and upper montane conifer and mixed hardwood-conifer forests. Montane meadows are typically highly productive habitats with continuous vegetation cover and a species-rich flora dominated by sedges, rushes, grasses, and herbs (Thorne 1988). Floristic composition in montane meadows varies with moisture, elevation, and geographic location. Saturated soil conditions during the growing season and competition from grasses and herbs prevent upland vegetation from becoming established in montane meadow habitats.

Montane meadows have a patchy, uneven distribution pattern in mountainous environments. They are restricted to areas with gentle slope gradients and relatively impervious bedrock that are conducive to the accumulation of fine-grained alluvial sediments, high retention of soil moisture, and shallow depth to groundwater. In suitable topographic settings, the size and distribution of montane meadows are generally controlled by local hydrology and soil texture. At a local scale, the proper combination of moisture and fine texture soils must exist for montane meadow habitat to occur. Meadows that occur narrowly along a stream course are referred to as *stringer meadows* (Holland 1986). In many cases meadows form along fault zones or other geologic contact points that conduct or impound groundwater (Stephenson and Calcarone 1999).

The San Bernardino and San Gabriel Mountains are characterized by very high local relief and rugged topography typical of recent orogeny and support mostly small meadows in a fragmented distribution pattern. The Peninsular Ranges and the southern Los Padres Ranges, which are characterized by less dissected topography with more broad, intervening valleys, have more expansive meadow habitats. The northern Santa Lucia Ranges and Santa Ana Mountains are mostly devoid of meadows due to a lack of suitable topography.

Meadows can be classified as wet, dry, or alkaline but are usually relatively mesic, even in late summer, compared to adjacent upland montane plant communities (Holland 1986). Stable meadows have shallow slopes and small drainage basins, where fine-grained alluvium accumulates.

Like most montane vegetation, montane meadows occur at lower elevations in the northern parts of California and higher elevations in the south because of greater precipitation in the north (Holland 1986). Some high-elevation meadows occur in protected north-facing drainages and are supported primarily by snowmelt (Thorne 1988).

Distribution and Abundance on National Forest System and Adjacent Lands: Montane meadows occur sporadically at elevations above 3,200 feet (975 meters) on all four southern California National Forests. An estimated 55,446 acres (22,438 hectares) of montane meadows occur in southern

California, of which 38% (21,070 acres [8,527 hectares]) occurs on public lands. The largest montane meadows and meadow complexes on and adjacent to National Forest System lands in southern California are on the Los Padres National Forest near Mount Pinos and on the Cleveland National Forest in the Laguna Mountains and near Mount Palomar (Stephenson and Calcarone 1999).

The majority of the large meadows in San Diego County are located on private lands (e.g., Cuyamaca, Mendenhall, French, and Dyche meadows). Other expansive meadow systems are found in the San Jacinto Mountains, with the largest located in Garner Valley. The many meadows in the San Bernardino Mountains on the San Bernardino National Forest are mostly small in size and have a fragmented distribution. On the Angeles National Forest, there are very few meadows because of the steep topography of the San Gabriel Mountains, but Big Pines Meadow is significant as the primary location for the rare San Gabriel Mountains greenish blue butterfly (*Plejebus saepiolus aureolus*). In the Castaic region, Knapp Ranch Meadow is estimated to cover almost 200 acres (81 hectares).

Montane meadows on the Los Padres National Forest include Toad Springs, Thorn and Chula Vista meadows; Yellow Jacket Creek (a series of stringer meadows); and several unnamed meadows near Lockwood Valley, Grade Valley, Cuddy Valley, Mount Abel, and the San Emigdio Mesa area (Stephenson and Calcarone 1999). Montane meadows are generally absent in the northern Santa Lucia Ranges.

Ecological Processes:

Two physical conditions characterize meadows: (1) poor drainage with a shallow water table usually within 2 feet (0.6 meter) of the soil surface in mid-summer, and (2) fine-textured soil enriched with organic matter (Stephenson and Calcarone 1999). Geomorphology and hydrology in combination control the distribution, stability, and quality of montane meadow habitats.

Factors that Influence Ecological Processes: Factors that alter hydrologic and geomorphic processes can affect meadow stability and species composition. In mountain environments throughout the western United States, changes to montane meadow environments over the last 150 years have resulted from water storage and diversion, development of roads and trails, introduction of livestock grazing, and changes in the fire regime. Many of these historic changes are interrelated but all have resulted in similar effects: stream channel incision, the desiccation of meadow habitats, a decrease in the cover and vigor of native vegetation, and the invasion of montane tree species and nonnative undesirable plant species. In general, the level of disturbance to meadow habitats decreases with elevation.

Meadow habitats are sensitive to activities and disturbances that alter hydrology, remove vegetation, or cause soil erosion, especially during the winter and spring when the ground is most saturated. In meadow systems, particularly those on steeper slopes, erosion removes topsoil and fine-textured alluvium, resulting in gully formation. The resulting channelized surface runoff causes increased erosion and stream incision, channeling water away from the meadow and effectively lowering the water table. Over time, increased drainage of meadow soils can lead to a shift in floristic composition to more drought-tolerant species and tree and shrub species.

Grazing and trampling by livestock and other ground disturbances by recreational users such as hikers, mountain bikers, and off-highway vehicles create conditions favorable to the establishment and spread of nonnative undesirable plants (Stephenson and Calcarone 1999).

Livestock Grazing: Livestock grazing has been a traditional activity in montane meadows in the southern California mountains since at least the early 1800s (Minnich 1988). Livestock have fundamentally altered or degraded vegetation and hydrogeomorphic processes in many montane meadows on National Forest System lands by increasing erosion, gully formation, stream incision, destabilization of streambanks, and by shifting floral composition from native perennials to nonnative annual species. However, there are few scientific assessments that quantify the ecological effects of grazing (Stephenson and Calcarone 1999).

When assessing the effect of livestock grazing on meadow processes, it is important to distinguish the impacts of historic grazing from those of current grazing levels. Abusive grazing practices at the peak of livestock use in the late 1800s and early 1900s caused permanent environmental changes that are orders of magnitude greater than those resulting from current practices. Livestock use was particularly high in the montane meadows of the San Bernardino and San Jacinto Mountains; it is reported that 30,000 sheep were herded into Big Bear Valley in the late 1860s. There are numerous accounts of extensive devegetation and erosion damage occurring during this period. Although sheep grazing was discontinued in the mountains in the early 1900s, degradation to meadow habitats persists (Minnich 1988, Stephenson and Calcarone 1999).

Since 1900, summer has been the principal period of livestock use of montane meadows on the four southern California National Forests. During the 20th century, there was a gradual decline in the intensity and extent of cattle grazing in the region (Minnich 1988). Once the dominant use of meadows in the San Bernardino and San Jacinto Mountains, grazing has been greatly reduced as ranching in the surrounding valleys has declined and other land use activities, particularly recreation, have received greater emphasis. Today, grazing is concentrated on the Los Padres National Forest and, to a lesser extent, on the Cleveland National Forest and adjacent areas. Over time gully systems may become stabilized and form riparian habitat; such a process has transpired at Knapp Ranch on the Angeles National Forest and Thorne Meadow on the Los Padres National Forest (Stephenson and Calcarone 1999).

Fire: Fire suppression activities may have affected meadow habitats, especially forest-meadow ecotones. Located in valley bottoms and characterized by high moisture content, montane meadows probably have a low probability of burning from lightning fires. There is evidence, however, that Native Americans may have deliberately set fire to some montane meadows to clear vegetation, increase forage quality for game animals, or to increase the productivity of desirable pyrophytic plant species (Lewis 1973, Denevan 1992). Shepherds in the 19th century continued this practice in many areas by setting fires in meadows to improve forage for sheep as they descended from the mountains in the fall

(Minnich 1988). Fire suppression and the cessation of deliberate burning practices in the 20th century may have led to conditions that favored the encroachment of trees and shrubs into meadows, especially along meadow edges. (Vankat 1977; DeBenedetti and Parsons 1979, 1984; Benedict 1989; Helms 1987; Helms and Ratliff 1987; Vale 1987; Taylor 1990.)

Recreation: In southern California, montane meadows are popular locations for recreationists. Organizational camps, recreation areas, and public campgrounds are numerous in conifer forests and montane meadows (e.g., Laguna Mountain, Cuyacama, Idyllwild, Barton Flats, Big Bear/Holcomb Valley, Lake Arrowhead, Big Pines, Crystal Lake, Mount Pinos/Cuddy Valley, and Pine Mountain). Recreation activities have affected meadow environments through road and trail building and maintenance, trampling, and unauthorized motor vehicle and mountain bike uses. These activities can result in soil erosion, devegetation, and channelized surface runoff, all of which contribute to gully formation, stream incision, drying out of meadow habitats, and the establishment of upland and nonnative undesirable plant species. Meadow soils are especially vulnerable to compaction when they are moist or saturated and are easily rutted. These effects are typically localized, but persist and encourage other unauthorized off-trail uses (Stephenson and Calcarone 1999). Concentrated human activity is also more likely to cause incidental animal mortality and can cause disturbance-sensitive species to avoid what would otherwise be suitable habitat. Mule deer largely avoid areas regularly occupied by humans, to the extent that they do not utilize meadow habitats that would otherwise be high quality foraging and fawning areas (Stephenson and Calcarone 1999).

Land Development: Because meadows occur on level or gently sloping ground within mountainous environments, many have been directly affected by development and indirectly affected because of increased surface runoff from adjacent developed areas.

Runoff from road and trail systems appears to be one of the most severe impacts to meadow habitats on National Forest System lands. Road and trail systems, and adjacent developed areas that channel surface runoff and discharge to meadows, can result in erosion and gully formation. A number of nonnative plants commonly used in landscaping are invasive and could spread into meadow habitat when surface runoff from adjacent developed areas is channeled into meadow habitats. Invasion of nonnative undesirable plant species has affected floristic composition of many meadows, especially in lower elevation meadows. The nonnative common dandelion (*Taraxacum officinale*), for example, is hybridizing with the California dandelion (*T. californicum*), a species listed as endangered under the federal Endangered Species Act that is endemic to montane meadows on the San Bernardino National Forest (Stephenson and Calcarone 1999).

Climate Change: In the last 150 years, tree and woody shrub invasion has occurred in some meadow habitats on National Forest System lands in southern California. Numerous studies describe tree invasion of montane meadows in California and the western United States and have found a correlation between tree establishment and increased drying of meadows. While some studies suggest that synoptic-scale climatic warming since the Little Ice Age (circa 1850) has caused meadow desiccation and tree invasion, the timing of climatic warming coincides with the introduction of livestock grazing and alteration of fire regimes in montane meadow habitats (Vankat 1977, Helms 1987, Helms and Ratliff

1987, Vale 1987, Benedict 1989, Taylor 1990).

Management Considerations: Because of the degraded state of many meadow habitats, management considerations include reducing further direct and indirect effects on vegetation and hydrogeomorphic processes that support montane meadow habitats, as well as restoring meadow systems. Recent changes in the management of livestock grazing allotments on National Forest System lands have included (1) fencing to exclude cattle from some sensitive meadow habitats; (2) reductions in stocking rates; and (3) changes in seasonal use to reduce the amount of summer/fall grazing in favor of a winter/early spring use regime. The hydrologic and soil moisture regimes of many montane meadows, however, have been fundamentally altered by stream incision and gully formation that are likely to persist or intensify in the absence of restoration actions (Stephenson and Calcarone 1999).

Restoring natural hydrologic function and geomorphic conditions typically drives reclamation of meadow habitat. The assessment of disturbances such as grazing, alterations to the hydrologic regime, and recreation-related uses is an important first step in restoration. Common measures for restoration include: stoppage and reversal of gully formation using within-channel sedimentation structures, stabilizing streambanks, reestablishing native vegetation, and controlling surface runoff from developed areas or roads above meadows. When hydrologic and geomorphic conditions are stabilized or restored, revegetation has a higher probability of success and there is greater probability of the long-term sustainability of the meadow system.

Restricting recreation activities to system roads and trails, the installation of causeways and other elevated travel routes, and restoring topography and vegetation generally avoid or minimize recreation-related effects on meadows (Stephenson and Calcarone 1999).

Twenty-nine special-status plant species are associated with montane meadow habitats in the southern California National Forests, 20 of which occur on the San Bernardino National Forest, and seven on the Cleveland National Forest. Approximately 11 species are in decline, but the population status of most special-status meadow plants is poorly known. Because montane meadow habitats are small in size and have a fragmented distribution pattern, *in situ* conservation of meadow habitats and populations of special-status species is vital.

Five special-status wildlife species are associated with meadow habitats: Laguna Mountains skipper butterfly (*Pyrgus ruralis lagunae*), San Gabriel Mountains greenish blue butterfly, calliope hummingbird (*Stellula calliope*), Lincoln's sparrow (*Melospiza lincolnii*), and MacGillivary's warbler (*Oporornis tolmiei*). In addition, many other special-status wildlife species, such as raptors, bats, and mule deer, use meadow habitats for foraging.

The Cleveland National Forest has developed a habitat management guide for four sensitive plants that grow in riparian montane meadows: Cuyamaca larkspur (*Delphinium hesperium* ssp. *cuyamacae*), lemon lily (*Lilium parryi*), Parish's meadowfoam (*Limnanthes gracilis* var. *parishi*) and San Bernardino blue grass (*Poa atropurpurea*). Current management direction for protecting or enhancing montane meadow habitat for federally listed species on National Forest System lands include the following measures (USDI Fish and Wildlife Service 2001):

- prohibit livestock grazing until after seeds set in meadows supporting federally listed species;
- remove facilities from meadow areas;
- allow only emergency use of roads in meadow habitats, and allow administrative use only when meadows are dry;
- remove trails from meadows or replace them with boardwalks;
- reduce or alter the season of commercial water extraction where it is adversely affecting federally listed plants;
- remove wild burros if they reoccupy sensitive meadow habitats;
- close meadow areas to dispersed camping; and
- conduct additional biological surveys of meadow habitats.

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Montery Coastal Habitats

Description: The Monterey coast landscape is unique on the southern California National Forests. Because of its northern latitude and high relief in close proximity to the coast, the northern Santa Lucia Ranges receive substantially more precipitation than other parts of the southern California National Forests. As a result, the floristic composition and structural characteristics of the vegetation are more closely related to northern California.

The low-elevation (less than 4,500 feet [1,372 meters]), mesic conifer forests in the northern Santa Lucia Ranges are distinct from other montane conifer forests found at higher elevations in southern California. The vegetation is characterized by coastal prairies, the southernmost occurrence of coastal redwood (*Sequoia semiervirens*) forest, the southernmost concentrations of Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*) and Pacific madrone (*Arbutus menziesii*), and Santa Lucia fir (*Abies bracteata*).

Coastal redwood forests in Monterey County are at the southern, dry edge of their range. Unlike stands in northern California, these stands are restricted to moist but well-drained canyons and north-facing slopes adjacent to the coast; they are surrounded by a mosaic of grasslands, chaparral, coastal sage scrub, Santa Lucia fir, and other conifer and hardwood forests (Zinke 1988, Noss 2000). The Monterey County coastal redwood population is genetically distinct from those north of San Francisco, and stands support unique structure and composition of understory plant species and mycorrhizae (Noss 2000).

Santa Lucia fir and coastal redwood forests provide the southernmost breeding habitat for marbled murrelet (*Brachyramphus marmoratus*), and support populations of California spotted owl (*Strix occidentalis occidentalis*) and California giant salamander (*Dicamptodon ensatus*). Santa Lucia fir is endemic to the northern Santa Lucia Mountains, where it occurs in relatively inaccessible areas such as steep north- or east-facing slopes, along ridges, in canyon bottoms, and on raised stream benches and terraces (Sawyer and Keeler-Wolf 1995). In some stands, canyon live oak (*Quercus chrysolepsis*) shares dominance with Santa Lucia fir. At lower elevations Santa Lucia fir occurs in drainages with coast live oak (*Q. agrifolia*), Pacific madrone, and coast redwood. At higher elevations it grows with tanoak (*Lithocarpus densiflorus*), interior live oak (*Quercus wislizeni*), and incense cedar (*Calocedrus decurrens*).

Grass-dominated coastal prairies occur in a mosaic with sagebrush/buckwheat-dominated coastal scrub along the coastline of the northern Santa Lucia Ranges. These habitats are unique on the southern California National Forests and support several sensitive plants and butterflies, including Hutchinson's larkspur (*Delphinium hutchinsoniae*), adobe sanicle (*Sanicula maritima*), Smith's blue butterfly (*Euphilotes enoptes smithi*), and Doudoroff's elfin butterfly (*Incisalia mossii doudoroffi*).

This region also supports mixed evergreen forest (dominated by coast live oak, California bay (*Umbellularia californica*), and Pacific madrone); oak woodlands; and savanna. Lower montane coniferous forest of ponderosa pine (*Pinus ponderosa*), Santa Lucia fir, and Douglas-fir occur at the highest elevations on the inland side of the Santa Lucia Mountains. A mosaic of chaparral, mixed evergreen forest, and oak woodland occurs below the conifer forest belt. All of these habitat types are discussed in other habitat descriptions.

Distribution and Abundance on National Forest System and Adjacent Lands: Monterey coastal habitats occur only along the coastal slopes of the northern Santa Lucia Ranges. On the Los Padres National Forest, the Monterey coast landscape covers 99,872 acres (40,417 hectares), including 35,264 acres (14,271 hectares) of chaparral and coastal sage scrub, 32,369 acres (13,099 hectares) of redwood and other conifer forests, 25,466 acres (10,306 hectares) of mixed evergreen forest, and 6,751 acres (2,732 hectares) of grasslands and oak savanna (Stephenson and Calcarone 1999). While many of the species found within these habitat types are more common in northern California or further south, along the Monterey coast they constitute an unusual mosaic and juxtaposition of habitats and species.

Santa Lucia fir is endemic to a 13-mile by 55-mile area in the northern Santa Lucia Mountains. Most Santa Lucia fir grows above 3,200 feet (975 meters) up to the summit of Cone Peak, but it is also found in drainages near the coast adjacent to redwood. The southernmost stands occur at lower elevations in drainages in San Luis Obispo County. There are 16,658 acres (6,741 hectares) of coast redwood and Santa Lucia fir forest, with areas dominated by Santa Lucia fir variously estimated as 7,576 acres (3,066 hectares) in a survey conducted in the 1970s, and 1,400 acres (567 hectares) in the current *Los Padres National Forest Land and Resource Management Plan*. Virtually all (95%) stands of Santa Lucia fir occur on public lands. Most stands are located in the Ventana Wilderness area on the Los Padres National Forest, with a smaller portion occurring on U.S. Army Fort Hunter Liggett to the east. (Stephenson and Calcarone 1999.)

Ecological Processes: The proximity of the Pacific Ocean renders the Monterey coastal habitats one of the most temperate landscapes in the southern California National Forests. Winters are cool and moist and summers are mild because of the ameliorating climatic effects of cold ocean water offshore and persistent summer fog near the coast. Soil moisture, especially below forest canopies but also in coastal sage scrub and chaparral, is augmented by fog drip during the summer (Zinke 1988, Noss 2000).

Factors that Influence Ecological Processes: The remoteness and rugged topography of the northern Santa Lucia Ranges combine to foster high ecological integrity, characterized by natural hydrologic regimes, relatively few exotic species, low levels of human impact, and extant populations of native species that have disappeared from many areas. Much of the region is roadless and a substantial portion is designated as wilderness. Most of the streams on the coastal side of the mountains have natural hydrologic regimes with unimpeded flow (Stephenson and Calcarone 1999).

The primary factors affecting ecosystem patterns and processes are recreation activity in the coastal watersheds, livestock grazing, and the invasion of nonnative undesirable species. Recreation use is high along the lower stream reaches near Highway 1; several developed campgrounds are situated along the Big Sur River, and a trail system extends into upper portions of the drainage (Stephenson and Calcarone 1999).

Because many species are, in this area, near the southern and drier margins of their range, natural habitats along the Monterey Coast can be vulnerable to disturbances. Isolated populations, for example, can have difficulty recovering if large distances separate a disturbed site and a potential source population. Moreover, because mature woody vegetation is the primary mechanism for trapping moisture from fog, vegetation removal through disturbances such as fire or logging can reduce available soil moisture and affect successional processes (Zinke 1988, Noss 2000).

Limited logging has taken place in coastal redwood, Santa Lucia fir, and lower montane coniferous forests from the 1880s to the 1970s. Many stands were selectively cut or high graded, and smaller-diameter trees dominate the remaining stands (Stephenson and Calcarone 1999, Noss 2000).

Livestock grazing has been widespread and continues in many areas of the northern Santa Lucia Ranges. There are currently 51,898 acres of active livestock grazing allotments in the northern Santa Lucia Ranges, most of which are concentrated at lower elevations near the coast or on the inland side. Grazing is most intense on coastal prairie habitats, which have suffered a reduction in native perennial species and an increase in nonnative and annual species (Heady et al. 1988).

The relationships between fire and vegetation patterns are not well understood in Monterey coastal communities. While many of the plant species occur in habitats where fire is a natural ecosystem component, natural fire occurrence is probably rare in these coastal habitats. Although recorded fires have been smaller and less frequent in this mesic area, a single event (the approximately 180,000-acre Marble Cone fire) burned much of the region in the late 1970s. It is unclear if this is a typical historic pattern, but there is little evidence to suggest that the area is experiencing vegetation changes as a result of shifting fire regimes (Stephenson and Calcarone 1999).

Santa Lucia fir is generally regarded as intolerant of fire, yet some mature stands have survived wildland fires. There are few differences between past and present fire intensities within stands, despite changing fire regimes statewide (Stephenson and Calcarone 1999). Although coastal redwood often resprouts following wildfire, several stands have been degraded because of frequent fires and the short stature of the stands (Zinke 1988, Noss 2000).

Santa Lucia fir is considered stable but at risk due to its highly restricted natural range. As a narrow endemic, individual stands and the population as a whole are vulnerable to natural and human-caused threats such as diseases, cone parasites, catastrophic wildfire occurrences, and the invasion of nonnative undesirable species into the understory. The rhizomatous shrub French broom (*Genista monspessulana*) is particularly invasive and difficult to eradicate once established. It directly competes with seedlings of Santa Lucia fir and other native understory species (Stephenson and Calcarone 1999).

Like Santa Lucia fir, plant species populations at the southern extension of their ranges, such as coast redwood and tanoak, are more at risk from changes in the disturbance regime and from catastrophic occurrences. Associated wildlife species, such as marbled murrelet, are similarly vulnerable. In general, many of these populations are relatively isolated from the main centers of their more northern distributions and, consequently, exhibit local genetic adaptations (Noss 2000).

Climate Change: Because they are at the southern edge of their natural ranges in the northern Santa Lucia Ranges, many habitats along the Monterey coast may be highly responsive to changes in precipitation, temperature, oceanic circulation, and other potential changes resulting from global climate change. In contrast to other parts of the southern California National Forests (particularly other low-elevation habitats), this area is characterized by remote, naturally functioning ecosystem patterns and processes. While logging and grazing have affected some habitats, their effects have been less severe than in other areas (Stephenson and Calcarone 1999). Consequently, this setting may be scientifically valuable for describing and analyzing the potential effects of climate change on individual species and ecosystem processes in low-elevation and coastal environments.

Management Considerations: Land use and forest activities in this region are relatively limited and strongly oriented towards recreation. The Big Sur coastline receives most of the recreation activity. Rugged terrain in the area historically limited road development and led to the establishment of the Ventana Wilderness, which encompasses the majority of the northern Santa Lucia Range.

The management of nonnative undesirable species in the coastal Monterey region is considered a high management priority. Nonnative undesirable species are among the greatest threats to the integrity of natural communities in coastal Monterey habitats. The removal of pampas grass and species of broom along highways, roadsides, and in forest clearings is especially important. (USDI Fish and Wildlife Service 2001.)

Almost all Santa Lucia fir and coastal redwood forests in the northern Santa Lucia Ranges are protected from timber harvesting, but they are at risk from stand-replacing fires. Fire suppression activities such as fire lines and fuel breaks are not currently allowed in stands on the Los Padres National Forest. Fire is not prevented from burning through stands nor is it directed toward stands. The ruggedness of the landscape, however, reduces the effectiveness of active fire suppression in many areas. (Stephenson and Calcarone 1999.)

The conservation of the unique flora and natural riparian systems in Monterey coastal habitats should be a management priority. However, more information needs to be gathered on the distribution, population status, and trends and ecological relationships of many species in this area.

Special-Status Species: There are 10 special-status wildlife species, including six federally listed species, that are restricted to coastal areas of the northern Santa Lucia Ranges on National Forest System lands. Only a very limited portion of the total range of most of these species occurs on National Forest System lands (Stephenson and Calcarone 1999). Three butterflies and California giant salamander, however, have a substantial percentage of their known ranges on the Los Padres National Forest; but the trends of these species are unknown. The southernmost population of marbled murrelet (federally listed as threatened) occurs in coastal redwood groves in Monterey County.

Of five special-status plant species that occur in this habitat, four are USDA Forest Service Region 5 Regional Forester's Sensitive Species. All of these species have a low level of vulnerability on National Forest System lands. Some are associated with the Arroyo de la Cruz endemic area, which is mostly on private property (Stephenson and Calcarone 1999).

Surveys should be conducted to map the extent of occupied habitats of these special-status species and to collect baseline population information in order to determine trends in population sizes and distribution.

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Oak Woodland, Savanna and Grassland

Description: Foothill oak woodlands and savannas occur as open canopy (10-50% canopy cover) to nearly closed-canopy woodlands (50-80% canopy cover) in canyons or along streams, and as savannas (less than 10% canopy cover) in broad valleys and rolling hills. Oak woodlands and savannas are typically two-layer communities, with the understory in canopy gaps and beneath crowns consisting of a nearly complete cover of grasses and forbs of grassland habitats, chaparral shrubs, or coastal sage scrub species.

The principal tree species in southern California are generally 16-49 feet (5-15 meters) in height and include coast live oak (*Quercus agrifolia*), blue oak (*Q. douglasii*), Engelmann oak (*Q. engelmannii*), and valley oak (*Q. lobata*). Woodlands of southern California black walnut (*Juglans californica* var. *californica*) and Alvord oak (*Q.* x *alvordiana*) are uncommon. Foothill pine (*Pinus sabiniana*), Coulter pine (*P. coulteri*), canyon live oak (*Q. chrysolepsis*), and black oak (*Q. kelloggii*) occur in some areas, but these species are primarily in lower montane, montane, and mixed evergreen forest habitats, and are described in those sections (Barbour 1988, Stephenson and Calcarone 1999). Oak savannas commonly contain valley oak on the Los Padres National Forest and Engelmann oak on the Cleveland National Forest, with coast live oak occasional throughout southern California in coastal areas (Stephenson and Calcarone 1999).

Coast live oak is an evergreen tree and the most abundant oak tree in southern California coastal areas with moderate climates, especially on moist slopes, valley bottoms, and the margins of riparian corridors. Coast live oak is the most dominant species in oak woodlands of the western Santa Lucia Ranges and western Transverse Ranges. In San Diego County it occurs with Engelmann oak, and on interior slopes on the Transverse and Santa Lucia Ranges it occurs with blue oak and valley oak. (Barbour 1988.)

Blue oak is a deciduous tree and commonly occurs in the interior coast ranges in monotypic stands or with coast live oak or valley oak. Some blue oak woodlands occur on serpentine soils. (Stephenson and Calcarone 1999.)

Valley oak is the largest deciduous tree in the western United States. Endemic to California, it occurs in areas with relatively mild winters west of the Sierra Nevada (Barbour 1988). True to its name, valley oak typically occupies valley floor and lower foothill communities with a grass-dominated understory on deep soils with a shallow depth to perennially available soil moisture. Valley oak savannas are often the dominant plant community in broad valleys that surround the mountains of Santa Barbara, San Luis Obispo, and Monterey Counties. Along drainages, valley oak forms denser riparian forests and is often found with blue oak, black oak, coast live oak, sycamore (*Platanus racemosa*), and black walnut (Sawyer and Keeler-Wolf 1995). Valley oaks extend up mountain slopes in places, for example, they occur up to 5,000 feet (1,524 meters) on Chews Ridge in the northern Santa Lucia Range and up to 5,600 feet (1,707 meters) in the Tehachapi Mountains (Stephenson and Calcarone 1999).

Engelmann oak, a deciduous species, has a small natural range and is the only species of subtropical white oaks in California. It occurs most commonly in savannas with grassland understory on valley floors, foothill slopes, and raised stream terraces within riparian corridors in the northwestern Peninsular Ranges in San Diego and Orange Counties (Barbour 1988, Sawyer and Keeler-Wolf 1995). Engelmann oak woodlands are distributed from the San Gabriel Mountains south to Baja California Norte, Mexico; however, most occur in the foothills of San Diego County and southwestern Riverside County. The greatest concentration of Engelmann oak woodlands occurs in the foothills of San Diego County between Palomar Mountain and Cuyamaca Peak. Another major occurrence is located on the Santa Rosa Plateau in the southeastern Santa Ana Mountains (Stephenson and Calcarone 1999). Engelmann oak is rarely the dominant tree where it occurs; coast live oak is the dominant tree in most stands of Engelmann oak. Where they occur together, coast live oak tends to be more dominant on moist sites, and Engelmann oak on more xeric sites (Barbour 1988). On the upland side of riparian corridors, Engelmann oak occurs in dense stands with coast live oak, sycamore, and other hardwood species (Holland 1986). Some of the most successful stands of Engelmann oak grow on clay soils formed from a gabbro or basalt substrate. In San Diego County, Engelmann oak sometimes occurs on rocky, north-facing slopes with an understory of coastal sage scrub or chaparral, where it often hybridizes with scrub oak (*Q. berberidifolia*) or Muller's oak (*Q. cornelius-mulleri*) (Stephenson and Calcarone 1999).

Southern California black walnut woodlands are an uncommon foothill habitat distributed from Santa Barbara County to northern San Diego County. On north-facing slopes, Southern California black walnut can be the dominant tree in the canopy, or it can occur in mixed stands with other hardwoods, such as coast live oak. Southern California black walnut usually occupies mesic areas (i.e., riparian corridors, floodplains, and north-facing slopes) and prefers soils with a high clay content. At Los Pinetos Spring in the western San Gabriel Mountains, southern California black walnut grows with bigcone Douglas-fir and canyon live oak.

Distribution and Abundance on National Forest System and Adjacent Lands: Oak woodlands and savannas are among the most widely distributed communities in cismontane California and perhaps the most characteristic vegetation community in the state, but they have poor representation on National Forest System lands and other public lands in southern California. Oak woodland and savanna habitats occur in foothill and lower montane portions of the four southern California National Forests. These habitats occur mostly in low-elevation coastal areas on the Los Padres National Forest below 3,937 feet (1,200 meters) and below 4,593 feet (1,400 meters) on the Cleveland National Forest (Barbour 1988). There is relatively little oak woodland or savanna habitat in the San Gabriel, San Bernardino, and San Jacinto Mountains on the Angeles and San Bernardino National Forests (Stephenson and Calcarone 1999).

There are an estimated 168,208 acres (68,072 hectares) of coast live oak woodlands on public lands in southern California. Most of the occurrences of these types on National Forest System lands are on the Los Padres National Forest, with some extensive areas of coast live oak and Engelmann oak also found on the Cleveland National Forest. Coast live oak woodlands are widely distributed in the foothills and are better represented than savannas on public lands (Stephenson and Calcarone 1999).

Blue oak woodlands cover approximately 49,584 acres (20,066 hectares) of public lands in southern California. Extensive patches of high-quality blue oak woodland generally occur on the inland side of the southern Santa Lucia Ranges on the southern Los Padres National Forest west of Garcia Mountain, particularly in the upper Avenales Valley, the Joughlin Ranch area, and the vicinity of Branch Mountain. Blue oaks attain one of their highest elevations (5,906 feet [1,800 meters]) near Mount Pinos on the Los Padres National Forest (Stephenson and Calcarone 1999).

Valley oak on National Forest System lands in southern California is limited to 680 acres (275 hectares) on the Los Padres and Angeles National Forests. Valley oaks are most prevalent at low elevations in the Santa Lucia Ranges but also extend into portions of the southern Los Padres and Castaic regions and northern San Gabriel Mountains. The southernmost occurrences of valley oak woodland are in the San Fernando and Santa Clarita Valleys and the Santa Monica Mountains in Los Angeles County. Valley oak woodlands are poorly represented on public lands, although a more detailed mapping effort is needed to better quantify the exact amount (Stephenson and Calcarone 1999).

Approximately 2,530 acres (1,024 hectares) of Engelmann oak woodlands occur on public lands in southern California. Estimates of the extent of Engelmann oak woodlands, however, vary depending on mapping criteria. The USDA Forest Service estimates that Engelmann oak woodland is the dominant vegetation type on 17,054-21,083 acres (6,902-8,532 hectares) in San Diego County and 4,029 acres (1,631 hectares) in the Santa Rosa Plateau region. Another effort using aerial photographs and ground-truthing mapped approximately 78,000 acres (hectares) that contain at least some Engelmann oak. In that study, however, Engelmann oak was the sole dominant tree on only 1,300 acres (526 hectares), and was subdominant to coast live oak on more than 39,000 acres (15,783 hectares) (Stephenson and Calcarone 1999).

An estimated 2,828 acres (1,145 hectares) of Southern California black walnut woodlands occur on public lands in southern California. Distribution maps, however, date to the 1930s; consequently, the current extent and status of this woodland type is not well known. A small 30-acre (12-hectare) stand of Southern California black walnut woodland is located on the Angeles National Forest, and 3,896 acres (1,577 hectares) occur on the Los Padres National Forest. A small extent is also reported from the San Bernardino National Forest (Stephenson and Calcarone 1999).

Large stands of Southern California black walnut woodlands occur in Ventura, Los Angeles, and northern Orange Counties. The easternmost stands occur in southwestern San Bernardino County in Day, Etiwanda, and San Sevaine Canyons at the foot of the San Gabriel Mountains. Southern California black walnut woodlands are scattered in low foothills surrounding the Santa Clara River drainage (including the Santa Susana and Sulphur Mountains), in the Santa Ynez Mountains, along the north side of the Santa Monica Mountains, along the base of the San Gabriel Mountains, and in the Simi, San Jose, Puente, and Chino Hills. Other stands occur in the lower foothills of the southern Los Padres and Castaic regions. Some isolated stands occur in chaparral and coastal sage scrub (Stephenson and Calcarone 1999).

Alvord oak woodlands occur on about 65 acres (26 hectares) on the Los Padres National Forest in the northern Santa Lucia Range (Stephenson and Calcarone 1999).

Ecological Processes: Oak and walnut trees in general grow slowly compared to conifers. Valley oak growing in rich valley bottom soil with a shallow water table is an exception to this generalization. Growth rates are vary greatly with species and site conditions, and tree size is typically a poor indicator of age. To survive long, dry summers in foothill environments, most oak seedlings rapidly produce a deep taproot before the aboveground shoot emerges. Some species are also opportunistically deciduous during prolonged drought periods to avoid moisture stress (Griffin 1988, Giusti and Tinnin 1993).

Reproduction can be naturally slow and episodic for many oak species. Acorn production begins when trees reach 20-30 years of age, and acorns can require up to 2 years to mature. The size of the acorn crop is varies widely by year, stand, and species. Typically, a large volume is produced once every 3-7 years in what is called a "mast" year. Rodents, gopher, deer, and livestock consume acorns, and deer and livestock browse on seedling foliage (Griffin 1988, Giusti and Tinnin 1993). Most observed oak seedling reproduction occurs on mesic slopes and other protected sites (Barbour 1988).

The root systems of many oak trees are a combination of deep taproots and an extensive lateral root system extending beyond the canopy. Lateral roots are vulnerable to soil compaction and other ground disturbances or changes in hydrology, including summer irrigation, which promotes fungal growth (Giusti and Tinnin 1993).

The soils and growing environment beneath an oak canopy is significantly different than in canopy openings. Leaf litter and leaf leachates provide greater nutrients and organic matter to soil, and understory grasses and forbs in canopy shade mature later, grow up to twice as fast, and accumulate twice the biomass as plants in openings (Griffin 1988).

Oak woodlands provide high-quality habitats for many species of wildlife. In particular, mature woodlands provide specialized habitats such as large canopy trees, nest cavities and downed woody debris in close proximity to grasslands and chaparral, riparian habitats, and other habitat types.

The role of fire in oak and California black walnut woodlands is poorly understood. Trees and acorns are killed by wildfire, but many species resprout after burning, especially coast live oak and walnut. Large valley oak and blue oak trees, however, generally do not stump sprout. Mature trees with light fuels and litter accumulation in their understory may be able to withstand fire occurrence (Barbour 1988).

Factors that Influence Ecological Processes: Woodlands dominated by oaks and southern California black walnut are threatened by a lack of seedling establishment and regeneration and by land development (Griffin 1988, Barbour 1988). Stands are becoming more senescent, and older trees, as they are lost, are not being replaced by younger ones. Many mature oak woodlands are reported to be especially devoid of trees established in the last 100-150 years (Griffin 1988). In some areas with no oak seedlings, foothill pines are becoming established in mature oak woodlands (Griffin 1988). Nonnative annual grass species and intensive livestock grazing are reported as the two most prominent impediments to oak seedling establishment.

Lack of oak regeneration, however, does not appear to be the result of any single factor but is rather the combined effect of competition from nonnative grass species, livestock grazing, and an unnatural abundance of acorn-eating animals such as gophers and ground squirrels. Natural variations in precipitation, mast years, and seed predation are other possible factors. The closed-canopy coast live oak woodlands are less threatened. Regeneration does not appear to be a problem, and coast live oak is a vigorous crown sprouter after fires (Barbour 1988, Stephenson and Calcarone 1999).

Other factors affecting the ecological integrity of oak and walnut woodlands include land development, agricultural expansion, groundwater pumping and lowering of the water table, road building, unauthorized off-highway vehicle use, and changes in the fire regime. Many oak and walnut woodlands are located in urban-interface areas, and while many local jurisdictions have ordinances that regulate the removal of trees, fragmentation and habitat losses continue and affect woodland management on National Forest System lands (Giusti and Tinnin 1993).

***Nonnative Undesirable Species*:** The presence of nonnative annual grasses in native grasslands has been shown to increase oak seedling mortality by limiting the availability of soil moisture. Introduced annual grasses grow rapidly during spring and deplete surface water much earlier in the season than the displaced native perennial grasses. Oak seedlings exposed to rapid declines in soil moisture experience water stress and display reduced growth. In controlled experiments, valley oak seedlings grew significantly larger in association with native purple needlegrass than they did with nonnative wild oats (Stephenson and Calcarone 1999).

Livestock Grazing: Livestock production has long been the principal economic activity in foothill woodlands. Many of the oak woodland and savanna habitats on National Forest System lands are within livestock grazing allotments. Grazing allotments are particularly common in oak savanna types and encompass 60% of Engelmann oak woodlands and 87% of blue oak woodlands. Most of the private land near National Forest System lands is consolidated in private livestock ranches (Stephenson and Calcarone 1999).

Intensive livestock grazing has been shown to reduce survival of oak and walnut seedlings because livestock consume and trample acorns and seedlings and cause soil compaction. Livestock grazing is also associated with the spread of nonnative undesirable grasses and forbs. However, moderate levels of grazing with carefully managed duration and timing can effectively reduce the cover of nonnative undesirable species and improve stands of native grasses, which may increase oak recruitment (Stephenson and Calcarone 1999).

Land Development: In recent decades an increasing number of large foothill ranches have been subdivided and converted into ranchette-style housing developments. This trend is expected to continue and perhaps intensify in the coming decade, particularly in San Diego, Riverside, Ventura, and Santa Barbara Counties (Stephenson and Calcarone 1999). While individual patches of oak woodland are often conserved in this pattern of development, wildlife habitat becomes increasingly fragmented and isolated, and the ratio of edge to interior habitat increases greatly.

For many wildlife species, the change in habitat quality because of land development or agricultural conversion can be as important as the loss in habitat quantity. Patterns of rural land development such as ranchettes tend to preserve individual trees or patches of oak woodland, savanna, and grassland. However, disruption of the understory, increasing habitat fragmentation, and the increasing ratio of edge to interior habitats reduce the habitat value for many species of wildlife.

Management Considerations: The trend towards increased development of foothill woodlands has several clear implications for public land management. As the urban interface expands and increasingly surrounds public wildlands, demand for recreation and other facilities increase and the ability to manage fire on the landscape becomes more constrained. Moreover, the decline of high-quality oak woodlands on private lands increases the importance of such habitats on public lands for wildlife and recreational uses. Land development and the decline in livestock ranching in the region is also gradually reducing the amount of livestock grazing on public land. (Stephenson and Calcarone 1999.)

The removal of oak woodlands on private lands and the poor representation of these habitat types on public lands underscore the importance of conserving oak and walnut woodland and savanna habitats on National Forest System lands. For example, 88% of Engelmann oak habitat in southern California is located on private land next to the fastest growing urban landscape in the country. On private lands, the rapid expansion of agriculture and urban development on lands surrounding the Los Padres National Forest is causing a substantial reduction in the extent of valley oak woodlands. For example, oak savannas are being cleared for large vineyard operations in the foothills and valleys of Santa Barbara County. More than 2,500 oaks in the Santa Ynez and Los Alamos Valleys have been felled during the last 2 years, and Santa Barbara County predicts vineyard acreage to increase 300% in 10 years. (Stephenson and Calcarone 1999.)

Encouraging private landholders to protect Engelmann oak is key to conservation. In addition, it is important for public land management agencies to pursue acquisition of lands containing Engelmann oak when they become available. In recent years, major progress has been made in conserving this species through the purchase of key areas by Riverside County (on the Santa Rosa Plateau), San Diego County (on Volcan Mountain), CalTrans, and the Cleveland National Forest (Roberts and Rutherford Ranches). The primary management concern for Engelmann oak woodlands on public lands is maintaining sufficient regeneration. (Stephenson and Calcarone 1999.)

Six special-status wildlife species and nine special-status plant species, including three that are federally listed, are associated with oak woodlands, savannas, and grasslands on National Forest System lands in southern California. Seven of the plant species occur on the Los Padres National Forest, and two are found on the Cleveland National Forest. Mature oak woodlands with features such as nest cavities, coarse woody debris, thick duff, and downed logs provide important habitat elements for special-status owls and salamanders. (Stephenson and Calcarone 1999.)

In general, relatively little is known about the population status and trends of special-status species associated with oak woodlands, savannas, and grasslands on National Forest System lands. Because of outdated mapping, a better determination of the current distribution, extent, and trends of California black walnut is needed. In addition, surveys of the size class distribution and reproductive status of oak woodlands and savannas would provide valuable information for management decisions regarding project level impacts, wildlife habitats, and restoration needs. Further controlled experiments on oak reproduction and fire relationships may also provide insights into the role of land management decisions on the long-term viability of oak woodlands on National Forest System lands.

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Pebble Plain

Description: Pebble plain habitat supports one of the most threatened and biologically rich plant communities within the San Bernardino National Forest and adjacent lands. Plant species occurring in this habitat include three Federally threatened, eight Forest Service Sensitive, and six Watch List plant species. Most of these 17 plant species are locally restricted to the eastern San Bernardino Mountains or the Big Bear area (USDA Forest Service 2002b). The habitat also provides the host plant requirements necessary for five species of rare butterflies, three of which are endemic, and are known only from pebble plain habitat on the Forest (USDA Forest Service 2002b).

Pebble plain is a unique habitat type consisting of distinct open patches of rocky inclusions within lower montane forest and woodland vegetation often dominated by Jeffrey pine (*Pinus jeffreyi*), single leaf pinyon (*P. monophylla*), and junipers (*Juniperus occidentalis* ssp. *australis, J. osteosperma*). These treeless, deep clay deposits support an assemblage of plants reminiscent of an alpine flora. This rare plant community consists of small cushion-forming plants, tiny annuals, grasses, and succulents. The plants are all well spaced, low growing, and sun tolerant, but exact floral composition varies between sites. The substrate consists of clay soil (up to 53%) mixed with a "pavement" of quartzite pebbles and gravel that are continually pushed to the surface through frost action (Holland 1986, Neel and Barrows 1990, USDI Fish and Wildlife Service 2001).

Remnants of an extensive Pleistocene lake bed (Krantz 1983), pebble plains are part of the Mojave crustal block uplifted during the Quaternary. Pebble plains have a naturally fragmented distribution on the landscape because the edaphic conditions required for their establishment occurs in isolated patches (Stephenson and Calcarone 1999, USDI Fish and Wildlife Service 2001).

Distribution and Abundance on National Forest System and Adjacent Lands: Pebble plain habitat has very limited distribution, occurring only in the northeastern San Bernardino Mountains. It occurs at elevations of 6,000–10,000 feet (1829–3048 meters) in areas receiving less than 25 inches of annual rainfall (USDA Forest Service 2002b).

Prior to 2002, there were reports of 379–546 acres (153–221 hectares) of pebble plain habitat, with 60-94% occurring on public lands (Neel and Barrows 1990, Stephenson and Calcarone 1999). This acreage was calculated using the first habitat definition described by Derby in 1979. Her definition was based on the presence of both the Bear Valley sandwort and the southern mountain buckwheat and oxidized clay soils with a saragosa quartzite component (Derby 1979). This definition promoted conservation efforts on the most classic examples of pebble plain habitat, but those containing only the Bear Valley sandwort and the southern mountain buckwheat.

Since that time, subsequent work by Krantz (1983) determined that outlying pebble plain habitat east and west of the Big Bear Valley fit the general soil description of the habitat but did not always support populations of both or either of the Bear Valley sandwort and the southern mountain buckwheat. The habitat does, however, support 15 other rare plant species as well as 4 rare butterfly species.

In 2002, to 1) promote protection of all rare species occurring within this habitat, 2) encompass the range of habitat characteristics on the Forest, and 3) to enhance habitat for the recovery of the listed species, San Bernardino National Forest botanists created a broader definition for pebble plain habitat (USDA Forest Service 2002b). The habitat is now characterized using a point system based on the presence of clay soils and strong and weak plant species indicators. Disqualifiers include a closed canopy and extensive litter cover. Using this broader definition, pebble plain habitat spans across approximately 4,000 acres on Federal, state and private lands (USDA Forest Service 2002b).

To facilitate the management of pebble plain habitat on the Forest, clusters of habitat were assigned to a "complex" within a defined geographic area. Some well known complexes include the Big Bear Lake, Coxey Meadow, Gold Mountain, Holcomb Valley, North Baldwin Lake, Sawmill, South Baldwin Ridge, Sugarloaf Ridge, and Arrastre Flats (Neel and Barrows 1990). In 2001, a survey and mapping effort identified additional habitat. The Broom Flat Complex was expanded and new complexes were assigned in Fawnskin, Rattlesnake, and Snow Valley (USDA Forest Service 2002b).

Ecological Processes:

Environmental: Pebble plain habitat provides a relatively harsh environment for plant growth. The combination of clay soil, frost heaving, extreme annual and daily temperature fluctuations, high light intensity, and desiccating winds is thought to prevent the establishment of tree species on pebble plains. Many of the endemic plants show several physical adaptations including low-stature growth habit, high reflectivity of all vegetative parts, and leaf succulence (USDA Forest Service 2002b). Vegetation growth and establishment following disturbance are slow. Because of the high clay content in the soil, the habitat is especially vulnerable to damage from vehicles when the ground is saturated. Deep ruts are created in the soil that directly affect the vegetation and alter the surface hydrology of pebble plains. Ground disturbance has also contributed to the increase of nonnative undesirable plant species. Cheatgrass (*Bromus tectorum)*, red stemmed filaree *(Erodium* cicutarium*)* and peppergrass (*Lepidium* sp.) are the most frequently observed nonnative species encroaching on to the habitat.

Ecological: Populations of pebble plain endemic plants are naturally threatened because of their limited habitat, and isolation from populations in other pebble plain habitats (Stephenson and Calcarone 1999, USDI Fish and Wildlife Service 2001). Only basic ecological information is known. Ciano (1983) found that the larger the size of the pebble plain and the closer proximity to other plains, the greater the species diversity. Studies on the genetic diversity of Bear Valley sandwort (*Arenaria ursina*), within and between the Sawmill, Arrastre Flat and Gold Mountain complexes, found a high degree of genetic variability between 6 populations (Ciano 1983). Pollination and genetic studies completed on southern mountain buckwheat (*Eriogonum kennedyi* var. *austromontanum)*, ash gray paintbrush (*Castilleja* *cinerea*) and silver-haired ivesia (*Ivesia argyrocoma)* found that little genetic material was exchanged between pebble plain occurrences (Freas 1988). Pollen transfer was observed at distances of less than four meters. Using seed traps, Freas found that seed dispersal was limited to about 5 meters from a pebble plain edge, however seed movement by animals was not addressed in this study.

Wildfire: Although it is not known to what extent fire has shaped the pebble plain community, the high percentage of rock cover in the habitat suggests it may not have played a significant role (USDA Forest Service 2002b). The interior of pebble plain habitat is largely immune from high intensity burning during wildfire due to the large percentage of bare ground, rock cover and the limited amount of fuel (USDA Forest Service 2002b). Instead, fire is usually carried around the margins of the plain. This is due to differences in soil type, increased litter from pinyon and juniper trees and the presence of sagebrush (*Artemesia nova*), and matchweed (*Gutierrezia sarothrae*). There is concern that the presence of cheatgrass (*Bromus tectorum*) could promote the spread of fire across the plain and increase the fire return intervals (USDA Forest Service 2001b). Krantz (1981) suggests that pebble plain habitat makes a natural fuelbreak and could be incorporated into a fuelbreak system if fuelbreak maintenance did not involve use of herbicides or grass seeding. If pebble plain habitat was proposed as part of a fuelbreak system, Forest botanists recommend that ground disturbance be avoided and that tree thinning along the habitat perimeter is not performed to prevent unauthorized off-road vehicle use into the habitat.

The effects of fire on three pebble plain species have been observed to date. Krantz (1981) observed that *Arenaria ursina* showed a vigorous response after the Heartbreak Fire, which occurred prior to 1981. He observed "many plants surviving, fruiting and seeding in good densities." Krantz also documented the post fire response of *Eriogonum kennedyi* var. *kennedyi* after the 1976 Coyote Fire within the Coxey Pebble Plain Complex. Plants infrequently resprouted from their caudexes and he observed little regeneration from seedlings (Krantz 1981). Post-fire effects to Kennedy's buckwheat (*Eriogonum Kennedyi* var. K*ennedyi),* were monitored for 2 years following the September 1999 Willow Fire with the same results. Occurrences of Kennedy's buckwheat, burned at high intensity, did not re-sprout after fire nor was regeneration from seed observed during the 2-year monitoring period (USDA Forest Service 2002a). This may have implications for the rare butterfly species that host entirely on this plant. Post-fire response of *Arabis parishii* was also monitored in this study. Parish's rock cress, which burned at low intensity, resprouted from its base and flowered the first year after the fire.

Floral and Faunal Relationships: There is a hemiparasitic relationship between 2 federally threatened plant species occurring in the habitat and ecological studies are needed. The ash gray paintbrush (*Castilleja cinerea*) is hemiparasitic on the Bear Valley buckwheat (*Eriogonum kennedyi* var. *austromontanum)* and 2 other more common buckwheats (*Eriogonum kennedyi* var. *kennedyi, E. wrightii* var. *subscaposum*). The paintbrush is also known to host upon sagebrush (*Artemesia nova, A. tridentata*). Within modeled habitat of the ash gray paintbrush, the conservation strategy includes protection of the occurrences, its host plants and their habitat.

There are also a number of rare butterflies, endemic to pebble plain habitat in the San Bernardino Mountains, which use *Eriogonum kennedyi* varieties as their host plant and studies are needed. See the updated butterfly species accounts for additional information. In the interim, any activities affecting the host plants or stages of the butterfly life cycles should be considered prior to implementation (USDA Forest Service 2002b). This includes the potential effects to host plants during pebble plain monitoring, and collection of host plants or their seed.

Factors that Influence Ecological Processes:

Prehistoric acreages of pebble plain habitat are not known. An estimated 150 acres (61 hectares) of pebble plain habitat was lost by creation of the Big Bear Lake reservoir in the 1800s. Historic gold mining, and previous cattle grazing and rock collection affected the habitat. Subsequent urbanization of Big Bear Valley and associated high impact land use also resulted in habitat loss (USDA Forest Service 2002b). Ongoing disturbances which further reduce the extent and quality of pebble plain habitat include: roads, small mining operations, recreational activities and Special Use Permits, development, and unauthorized grazing. Fire suppression activities also have the potential to affect habitat.

Habitat protection measures including barrier installation to delineate trails and developed sites, placement of boulders to impede vehicle access, and restoration of disturbed sites have been completed to reduce impacts on the Forest. These measures are monitored and modified as needed. An increase in public educational opportunities and submittal of a mining withdrawal are additional measures completed to protect habitat. None of the pebble plain complexes, however, are fully protected from ongoing disturbance (USDI Fish and Wildlife Service 2001) due to several authorized activities and impacts from unauthorized use (USDA Forest Service 2002b).

Roads: Activities associated with road use have resulted in direct loss of individual pebble plain plants through crushing by vehicles, horses, mountain bikes, or foot traffic. One pebble plain, in the Sawmill Complex, has been completely devegetated as a result of heavy vehicle use (Stephenson and Calcarone 1999, USDA Forest Service 2002b, USDI Fish and Wildlife Service 2001). The indirect effects of roads on pebble plain habitat include an influx of nonnative undesirable plant species; dust from dirt roads, which reduces photosynthesis and reproduction of pebble plain plants; and interruption of natural sheet water flow across the habitat. In addition, the roads within or near pebble plain complexes provide access for unauthorized vehicle travel through pebble plains habitat, often resulting in the formation of unauthorized roads, and loss and degradation of pebble plains plants (Stephenson and Calcarone 1999, USDI Fish and Wildlife Service 2001).

In the late 1980s, the Forest began an analysis of road densities within pebble plain habitat. As a result, eleven roads within the Arrastre Complex (SBNF 1988) and several unclassified roads in the Sugarloaf Complex were decommissioned. To encourage habitat recovery, several sections of the decommissioned roadbeds within the Arrastre Complex were mechanically ripped. These locations provide sites for the study of pebble plain habitat restoration methodology. Road decommissioning in pebble plain habitat continued through the early 1990s with several more roads becoming decommissioned or rerouted. In 1999, the Forest took another close look at classified roads affecting pebble plain habitat and decommissioned seven additional roads or spurs (USDA Forest Service

1999b). Currently, due to the large number of unclassified roads within threatened habitats in the Big Bear area, the Forest is conducting an environmental analysis to determine if additional roads through habitat could be closed and rehabilitated.

As roads are decommissioned, they are usually sub-soiled to reduce compaction, barricaded, and signed as closed. To disguise roadbeds and promote vegetative cover, local seed is collected from adjacent land, propagated, then out-planted or direct seeded on to sites leading to pebble plain habitat.

To deter additional unclassified road development, monitor effectiveness of protection measures, and promote environmental education, a full-time resource patrol officer was hired for the Big Bear area of the Forest. This has resulted in timely fence repairs, rapid disguise of unclassified roads as they are developed and an increase in public environmental awareness.

Road maintenance activities (e.g., grading, cleaning and repair of drainage structures) have also resulted in the removal of pebble plain plants. To reduce effects of road maintenance, the Forest implemented a road maintenance plan in 1999, that modified maintenance activities within habitat (USDA Forest Service 1999).

Mining: Historic gold mining in the Holcomb Valley area during the late 1800s greatly affected pebble plain habitat. Although the scale of gold mining has been reduced, gold mining activities conducted by small miners continue to occur thoughout several of the pebble plain complexes. Several Plans of Operation for mining on the Forest may currently be affecting pebble plain species (USDA Forest Service 2001b). Prospecting, the search for mineral materials in anticipation of finding a discovery, is more widespread and thus the major concern due to lack of restrictions governing this use. Mining operation activities currently causing the greatest effects to habitat include the development of access roads, digging, storage of spoil materials, and dry washing.

In 2001, the Forest submitted an application to the Bureau of Land Management (BLM) to withdraw portions of the Forest from mineral location and entry under the mining laws. The lands specified in the application for withdrawal total approximately 44,575 acres and constitute habitat for 12 threatened, and endangered plant species occurring in pebble plain, carbonate, and meadow habitat, and for the endangered southwestern toad. At the time the application was submitted, all known pebble plain habitat was included in the proposal. Additional habitat that has been recently surveyed and mapped will also be proposed for inclusion in the withdrawal. On October 29, 2001, the BLM published the Notice of Intent in the Federal Register initiating a public comment period and starting a 2-year segregation affecting public lands proposed for withdrawal. This temporary segregation period will terminate by October 29, 2003. The Forest is currently conducting an environmental analysis on this proposal. Should the requested withdrawal be finalized, all lands specified in the application, or some portion thereof, would be closed to mineral location and entry for a period of 20 years, subject to valid existing rights (USDA Forest Service 2001b).

Collection of the beautiful iron-stained rock from the Gold Mountain Pebble Plain Complex has occurred for use in construction of rock walls, riprap, and chimneys in the Big Bear area for decades. In locations where collection activity had the potential to affect pebble plain habitat, permits for common variety minerals have been discontinued.

Recreation: Recreational activities have also degraded pebble plain habitat. Developed sites were constructed on or near pebble plains at Aspen Glen Picnic Area, Holcomb Valley Campground, Juniper Springs Campground and the Doble Trail Camp. Although impacts could not be eliminated, measures to reduce effects are in place. Portions of the Sugarloaf Trial, Pine Knot Trail, Green Valley Trail, the Pacific Crest Trail, and the Gold Fever Interpretive Trail were constructed within pebble plain habitat. Barriers and signs have been installed in all of these locations to delineate the trail corridor.

Special use events (Mountain Man, Motorcycle Trials, Team Big Bear Races, filming, etc.) previously located within habitat were relocated to other sites or the proposal was changed to eliminate effects. As requests for additional events are submitted, proposals are analyzed for effects to pebble plain habitat. If effects will occur, the Forest works with the proponent to relocate the project. If effects cannot be mitigated, the proposal may be denied or consultation with the USFWS may be initiated.

Prior to 1998, the Forest allowed dispersed target shooting at locations throughout the Forest. The non-wooded openings provided by pebble plain habitat often became sites for target shooting, causing impacts to habitat from vehicle use and trash. In 1998, the Forest completed an analysis to find suitable locations for shooting areas that were fire safe and did not contain sensitive plant or wildlife habitat. Several dispersed sites were designated and garbage was removed from most of the habitat previously used for target shooting.

Development: Pebble plain habitat is affected by recreational cabins under Special Use Permit from the Forest Service in the Snow Valley and Metcalf Tracts. Parking areas were delineated in 1999 to reduce effects of parking on the habitat. Cabin owners were also advised to plant native plant species in their yards to reduce effects to habitat from non-native plants.

Other Special Use Permits that affect habitat include the Rim Nordic Ski Area, Snow Valley Ski Area, Onyx Communication site access road, and the former Snow Forest Ski Area. Several utility companies have corridors in habitat. Measures to reduce impacts to habitat in these locations are currently in place.

Unauthorized Grazing and Wild Burro Territory: There are no active grazing allotments on the Forest that include pebble plain habitat. Pebble plain habitat is affected in the Broom Flat Complex, however, from trespass grazing on the Rattlesnake Allotment. Pebble plain habitat within the Rattlesnake Complex may also be affected. Recent construction of drift fence to prevent cattle access has been ineffective, as cattle are often reported within both complexes. The Forest continues to work with the permittee to contain the cattle. Elimination of cattle trespass is a Forest objective.

Burro territory is present within the North Baldwin, Gold Mountain, South Baldwin/Erwin Lake, Broom

Flat and Rattlesnake pebble plain complexes. The Forest completed formal consultation with the USFWS regarding effects from burros, and a monitoring plan remains in place today (USDA Forest Service 1996). In 1997, the Forest removed 100 burros from residential areas after several burros were hit along the highways. Since that time, effects to pebble plains on the east side of Big Bear Valley have been reduced.

Fire Suppression: Fire suppression activities that disturb or compact the soil have the highest potential to impact pebble plain habitat. Driving on habitat, construction of dozer lines, use of habitat for fire camps or staging areas, and Burned Area Emergency Rehabilitation treatments are of the greatest concern. Hand line construction, mop-up activities, use of alkaline water drops and aerial retardants are additional activities that may affect habitat, but to a lesser degree. To reduce the potential for fire suppression activities to impact habitat, the Forest implemented a fire suppression plan for this habitat. Fire personnel are trained to identify habitat and to use suppression techniques that reduce or prevent soil disturbance. A notebook with habitat maps and suppression plans was also created and is updated and distributed to fire personnel annually.

Management Considerations: In 1990, The *Pebble Plain Habitat Management Guide and Action Plan* (Neel and Barrows 1990) was developed by the San Bernardino National Forest and the Nature Conservancy to provide management direction for long-term conservation of pebble plain habitat and the rare plants associated with it. Goals included conserving pebble plain habitat over a broad geographic range, reducing fragmentation and encouraging compatible uses. The guide also prioritized measures to promote habitat protection.

In 2002, a draft revision of the 1990 Guide was completed. Goals were extended to promote conservation of pebble plain habitat throughout its range, to aid in the recovery of 3 federally listed plants and to improve conditions for Forest sensitive species occurring in this habitat. New objectives include: 1) the need to heighten public awareness of pebble plain habitat through an environmental education program, 2) promotion of partnerships with agencies, universities, and other entities to conduct conservation activities, including research, education, interpretation and protection, 3) maintenance or improvement of all pebble plain habitat on Forest land to the maximum extent possible, 4) recovery of federally listed species, 5) improvement in methods for restoration of degraded habitat, 6) minimization of threats associated with forest uses, 7) future land acquisition of pebble plain habitat, and 8) designation of a Research Natural Area to promote ecological and restoration research (USDA Forest Service 2002b).

Maintenance of all protection measures previously described in this document is also recommended in the 2002 Habitat Management Guide.

Revegetation of previously disturbed habitat is also needed to restore and enhance pebble plain habitat. The Forest continues to restore the North Baldwin Pebble Plain, previously impacted by unauthorized trespass with a front-end loader. To date, erosion control measures, collection and propagation of seed from pebble plain plants on site, and out-planting and direct seeding have been completed. Monitoring and additional efforts to restore vegetation to the site using direct seeding are currently underway (USDA Forest Service 2001b). Over the long term, this project should benefit pebble plains plants by increasing their populations (USDI Fish and Wildlife Service 2001).

The former Snow Forest Ski Area is another pebble plain affected by prior use. The habitat is currently being evaluated and measures to restore the site will be recommended. Restoration at this site will become an ongoing project for the next decade.

Development of more effective measures to restore pebble plain habitat is needed. Methods to increase success of out-planting of potted material on site are needed, as prior success has not been high. Methods to increase germination after direct seeding are also needed. The Forest is dedicated to working with botanical gardens and other agencies to increase knowledge of ecological restoration methods upon recommendation from the USFWS (USDI Fish and Wildlife Service 2001).

Section 7(a)(1) of the federal Endangered Species Act (ESA) directs federal agencies to utilize their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

The USDI Fish and Wildlife Service (2001) developed conservation recommendations for listed species in pebble plain habitats on the San Bernardino National Forest, including the measures described below.

Quantify the extent of pebble plains plant habitat and determine the proportion of that extent that is affected or potentially affected by roads, trails, or other land use activities.

Participate in the development and implementation of any ongoing and future habitat and species conservation programs in Bear Valley and other communities.

Protect and restore all remaining pebble plains habitats and associated physical features. The restoration program should include involvement of species and restoration ecology experts to develop and refine methods of restoring pebble plains plants on disturbed surfaces.

Emphasize efforts to control or remove nonnative undesirable plants and animals to the maximum degree possible, especially nonnative undesirable grasses in areas occupied by listed plants.

Restrict, to the maximum extent possible, unauthorized human and vehicular activities in pebble plains and perform routine maintenance on fences, boulders, and other protective barriers for pebble plains habitats.

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Riparian Habitats

Description: Riparian habitats are typically characterized by linear woodlands or forests that occur along stream channels and adjacent areas with relatively moist soils. Riparian forests and woodlands contrast sharply with surrounding uplands by typically having greater canopy cover dominated by deciduous broad-leaved trees, multi-layered canopies, and overall increased vegetation density and species richness. Riparian habitats are extremely productive and important habitats for wildlife; they provide cover, shade, ameliorated microclimate, water sources, and nesting and foraging habitats. Most upland wildlife species rely on or use riparian habitats during some part of their life cycle.

Riparian habitats reach their peak as distinct ecological communities along mid- to large-order streams at elevations below 4,000 feet (1,219 meters) in the foothills and valleys.

There are many different riparian plant associations, but foothill riparian woodlands generally fall into one of three broad categories on the basis of dominant tree species: (1) Fremont cottonwood (*Populus fremontii*)/willow (*Salix* spp.), (2) California sycamore (*Platanus racemosa*)/coast live oak (*Quercus agrifolia*) and (3) white alder (*Alnus rhombifolia*) (Stephenson and Calcarone 1999).

Other common trees associated with riparian habitats on the four southern California National Forests include black cottonwood (*Populus balsamifera*), California bay (*Umbellularia californica*), big-leaf maple (*Acer macrophyllum*), Oregon ash (*Fraxinus latifolia*), black oak (*Quercus kelloggii*), and coast redwood (*Sequoia sempervirens*). Aspen (*Populus tremuloides*), which occurs in riparian habitats in other mountains of California and is one of the most wide-ranging tree species in North America, is virtually absent in southern California (Barbour 1988, Stephenson and Calcarone 1999). In many low-elevation areas, nonnative undesirable species such as arundo (*Arundo donax*) and tamarisk (*Tamarisk* spp.) have invaded riparian habitats and replaced native species.

Occurrence on National Forest System and Adjacent Lands: Riparian habitats occur along perennial and seasonal streams on all four southern California National Forests. Because riparian ecosystems are linear and often narrow features on the landscape, they are difficult to accurately map across large areas using remote sensing techniques such as aerial photograph or satellite imagery interpretation. The USDA Forest Service's existing vegetation maps do not fully capture the distribution of riparian habitats; therefore, quantitative estimates of the extent of the habitat on National Forest System lands in southern California are not available (Stephenson and Calcarone 1999).

Two disjunct stands of quaking aspen are present on National Forest System lands on the San Bernardino National Forest. The two aspen occurrences comprise fewer than 50 acres combined. Both stands are in the San Bernardino Mountains: one on Fish Creek in the San Gorgonio Wilderness Area and the other on upper Arrastre Creek (Thorne 1988). The aspen groves occur on decomposed granite in canyon bottom riparian areas. The Arrastre Creek grove is along a seasonal stream and occurs with mainly upland plants species such as sagebrush scrub, pinyon (*Pinus quadrifolia*), and Jeffrey pine (*P. jeffreyi*) forest. The Fish Creek grove also occurs along a stream within Jeffrey pine and subalpine forest (Stephenson and Calcarone 1999). Aspen is a clonal organism; the San Bernardino Mountain populations are almost genetically identical to each other (Thorne 1988).

Ecological Processes: The dynamics of streamflows and the proximity of groundwater control the extent and character of riparian, wetland, and aquatic habitats. Seasonality, volume, duration, and year-to-year variability of streamflows all greatly influence the structure and composition of ecological communities in the channel and floodplain. Groundwater fluctuations have a similar effect on communities associated with springs, seeps, and ephemeral water bodies.

The historic flow pattern in southern California streams reflects the region's climate of long, dry summers and short, wet winters. Stream discharge peaks in the winter and early spring and declines dramatically in the summer months. Low flow in summer can result in an interesting situation, in which the headward parts of streams may be dry, the middle portion wet, and the low portion dry during the summer months. The middle and lower portions of streams, typically at elevations below 3,000 feet (914 meters), support a higher number of aquatic and riparian species. However, streams that flow through bedrock canyons often have perennial flow because deep pools are fed by groundwater and the bedrock serves as a barrier to infiltration (Stephenson and Calcarone 1999).

The high variability of precipitation and runoff in southern California also produces large flood events, which periodically scour channels and redistribute sediment and bedload. Maximum discharge periods in high-elevation stream reaches tend to be more heavily influenced by the timing of spring snowmelt (Stephenson and Calcarone 1999).

Factors that Influence Ecological Processes: No other habitat type in the southern California National Forests has been as drastically altered by human activities as has freshwater riparian habitat. Ecological processes in riparian systems have been fundamentally altered by the development and operation of water storage and diversion structures, the invasion of nonnative undesirable species, development, including construction of roads and their associated drainage systems and crossings, and, to a lesser extent, livestock grazing, recreation, and mining.

Low-elevation streams face greater threats than high-elevation streams because water flows are much more likely to be diverted or altered, the adjacent terraces are commonly farmed or developed, and there is a greater abundance of nonnative undesirable species. The overall proportion of the surrounding watershed that is developed and roaded also increases at lower elevations (Stephenson and Calcarone 1999).

Water Storage and Diversion: Instream water storage and diversion have resulted in a dramatic reduction in the extent and distribution of native freshwater riparian habitats in this region. Approximately 95-97% of low-elevation floodplain riparian habitat in southern California has been eliminated, and all of the major streams that originate in the mountains contain dams or diversions at some point along them. In addition, many small streams and springs are dammed or diverted for water supply and flood control. Subsurface waters are also heavily tapped, lowering water tables and base flows of many springs and stream reaches (Stephenson and Calcarone 1999).

Most remaining riparian habitats function under a highly modified hydrologic regime with upstream dams regulating streamflows. More recently, strong regulatory policies on "no net loss" of wetlands have helped to check this decline, and the extent and quality of riparian habitats on public lands is relatively stable (Stephenson and Calcarone 1999).

Dams remove riparian habitat directly by inundation, but a greater effect is the amount of habitat degraded downstream by alteration of natural hydrologic and sedimentation processes. Typically, dams have the effect of reducing the magnitude and frequency of flood flows, increasing base flow, stopping the downstream transport of sediment, and altering instream water temperature.

The reduction in the magnitude and frequency of flood flows removes a key disturbance process in floodplain riparian habitats. Many riparian trees, such as willows and cottonwood, are short-lived and regenerate on floodplains following flood and sedimentation events. Major floods remove riparian vegetation by scouring and altering channel morphology but also deposit moist sediments necessary for regeneration and fish spawning.

The decrease in scouring flood events, coupled with an increase in base flow, improves conditions for riparian tree and shrub growth along stream banks, but these less-disturbed habitats have fewer opportunities to regenerate, and many forests have become senescent. Existing riparian woodlands on many stream reaches below dams have experienced an increase in vegetative cover, which can effectively channelize streams and transform them into habitats less suitable for some native aquatic and riparian species.

The interruption of the sediment supply by dams results in the water having greater erosive force, which causes channel incision. Channel incision lowers the water table and increases the vertical distance to the floodplain. The result is that any stream-reaches below dams are characterized by an almost complete lack of sand and fine gravel and by a succession of very deep scour pools floored with boulders and mud (Stephenson and Calcarone 1999). Sandy levees and bars suitable for plant growth are reduced and only coarse sediments remain. Many reaches lack suitable gravels for anadromous fish spawning. As stream incision progresses to a base elevation, lateral erosion occurs and stream banks supporting riparian vegetation are undercut.

The timing and duration of water releases from reservoirs can also greatly influence downstream habitats for riparian wildlife species. For example, large, sudden releases of water, particularly in the summer months, can quickly scour away a whole year's reproductive effort for native species such as arroyo toad (*Bufo microscaphus californicus*), red-legged frog (*Rana aurora*), pond turtle (*Clemmys marmorata*), and California newt (*Taricha torosa*). Potential spawning beds are reduced when sand and gravel bars are removed and new sediments do not replace them. Cooler instream water temperature favors many introduced species such as brown trout, but has a detrimental effect on native warmwater fish. Conversely, low-level year-round flow regimes facilitate the spread of exotic predators (e.g., bullfrog, sunfish, bass, blue-gill, catfish, and Asian clams) into downstream areas that historically were dry in late summer (Stephenson and Calcarone 1999).

An exception to the high level of modification to natural riparian systems is on the Los Padres National Forest. Most of the streams on the coastal side of the northern Santa Lucia Ranges have natural hydrologic regimes with unimpeded flow, and their largely undeveloped upper watersheds have helped maintain a high level of ecological integrity. Important streams include the Big Sur River, the Little Sur River, and San Carpoforo Creek. The aquatic habitats have few nonnative undesirable species, and support southern steelhead spawning and populations of red-legged frog, foothill yellow-legged frog (*Rana boylii*), tiger salamander (*Ambystoma californiense*) and California giant salamander (*Dicamptodon ensatus*) (Stephenson and Calcarone 1999).

Nonnative Undesirable Species: Next to alteration of streamflow, the biggest factor threatening the health of native riparian ecosystems is the spread of nonnative undesirable species. Reservoirs and other artificial aquatic habitats facilitate the introduction of a wide variety of nonnative undesirable aquatic species into the surrounding streams. Collectively, introduced species are causing a serious decline in the capability of riverine aquatic habitats to support native species (Stephenson and Calcarone 1999).

Brown-headed cowbird (*Molothrus ater*), which parasitizes the nests of native birds, and European starling (*Sturnus vulgaris*), which displaces native cavity-nesting birds, are causing declines in habitat capability in many areas. Bullfrogs, green sunfish, bass, blue-gill, catfish, and Asian clams are spreading in many streams, causing large impacts on native populations of amphibians, aquatic reptiles, and fish. Beavers were introduced into the Arrastre Creek aspen grove and subsequently reduced this grove by an estimated 25-50%. The nonnative undesirable plant species arundo and tamarisk are also spreading, displacing native vegetation and causing a decline in surface water availability in some low-elevation streams (Stephenson and Calcarone 1999).

Arundo is a large, perennial grass that has become widespread in many states. It was intentionally introduced to California in the 1820s in the Los Angeles area as an erosion-control agent in drainage canals. Arundo forms dense thickets primarily in riparian areas and drainage channels or where there is a shallow water table. On the four southern California National Forests, arundo occurs in foothill areas, primarily along large streams but also in other areas where there is pooled water. It occurs in more than 50 watersheds and reaches peak abundance along major rivers in the coastal basins such as the Ventura, Santa Clara, Santa Ana, Santa Margarita, San Luis Rey, and San Diego River systems (Stephenson and Calcarone 1999).

Arundo is mostly restricted to elevations below 2,000 feet (610 meters) and has not spread into the mountains or up the steep, narrow canyons in lower montane areas. Arundo requires well-developed soils to become established. Arundo tends to have a competitive advantage in riparian areas where there is a modified hydrologic regime. It has rapid growth rates, growing 2.1 to 4.9 times faster than native willow species. This rapid growth is sustained by the consumption of large amounts of water, causing a decline in the availability of surface water (Stephenson and Calcarone 1999).

Once established, arundo often forms monocultural stands that physically inhibit the growth of other plant species. These stands provide neither food nor cover for most native species of wildlife. Only a small number of bird species have been observed using arundo for nest sites, and dramatic reductions (50% or more) in the abundance and diversity of invertebrates were documented in arundo thickets compared with invertebrate populations found in native willow/cottonwood vegetation. Arundo also provides less shade in riparian areas than does native vegetation, causing increased water temperatures and lower oxygen concentrations, which in turn negatively affect fish and other aquatic species. Arundo thickets are highly flammable and are known to carry wildfire along riparian corridors (Stephenson and Calcarone 1999).

Tamarisk is widely distributed in southern California in both coastal and desert-side drainages. There are at least four species of tamarisk invading native riparian habitats; these have been documented in at least 60 watersheds on southern California National Forests, especially in foothill and desert streams with deep alluvial channels (e.g., arroyos). Tamarisk occurs in a number of lower montane drainages as well, but seems to spread slowly in narrow bedrock channels (Stephenson and Calcarone 1999).

Like arundo, tamarisk uptakes large quantities of water from the soil, effectively lowering the water table and reducing the amount of available surface water. In some areas, tamarisk has reduced or eliminated water supplies for bighorn sheep, pupfish, salamanders, and desert palm groves. Tamarisk also provides poor forage and nesting sites for wildlife. The scale-like leaves are unpalatable to grazers, and birds favor native riparian vegetation over tamarisk (Stephenson and Calcarone 1999).

Tamarisk is often called salt cedar because it exudes salts from its leaves. These salts accumulate in the soil, making the area less hospitable to native plants. Like arundo, tamarisk appears to be most successful in drainages with unnatural or reduced discharge, where it can replace native riparian forest species that no longer find suitable reproductive habitats. Because tamarisk is sometimes planted as an ornamental on private lands, invasions into riparian habitat are likely to continue because the tree produces enormous quantities of seed dispersed by wind and water (Stephenson and Calcarone 1999).

Livestock Grazing: Cattle tend to stay in riparian habitats for prolonged periods of time because of the available forage, browse and water. Prolonged grazing can result in devegetation of stream banks, prevention of seedling establishment, and degradation of the bed and bank and water quality from trampling and soil erosion. Current management practices are directed to reducing the amount of time livestock are present in sensitive habitats such as riparian areas (Stephenson and Calcarone 1999). Livestock grazing in riparian areas has been substantially reduced, resulting in some dramatic improvements in vegetation condition.

Recreation: Concentrated recreation use is causing localized habitat damage from devegetation, bank trampling, littering, and pollution in some areas. Foothill riparian areas are cool, pleasant places, so recreation pressure is inevitable in the vicinity of large urban populations. However, habitat degradation tends to be localized in a few popular, easily accessible areas (Stephenson and Calcarone 1999).

Development: Land and road development within watersheds typically alters natural hydrology and can cause channel incision. Development decreases the infiltration capacity of watersheds and increases channelized runoff. Roads tend to channelize water into ditches, often increasing or altering the amount of water that reaches the stream at critical points. These changes increase peak storm runoff, transport of water pollutants, and sediments from cleared lands, but decrease groundwater recharge and base flow.

Land ownership patterns and threat factors differ dramatically with elevation. While 74% of stream miles above 3,000 feet (914 meters) are on public lands, this proportion drops to 50% between 1,000–3,000 feet (305 and 914 meters), and down to 17% below 1,000 feet (305 meters) (Stephenson and Calcarone 1999).

Mining: Suction dredging and sand and gravel mining directly affect riparian and riverine aquatic habitats. Sand and gravel mining occurs in foothill streams where there are well-developed alluvial deposits. Most of these operations are on private lands, but they affect habitats and species movement along the riparian corridor. Sand and gravel mines completely alter the stream channel and usually result in the creation of deep pools that can prevent instream migration (Stephenson and Calcarone 1999).

Suction dredging utilizes high-pressure water pumps to vacuum a mixture of streambed sediment and water and pass it over a sluice box mounted on a floating barge. Denser particles (including gold) are trapped in the sluice box, and the remainder of the entrained sediment is discharged into the stream. Large tailing piles remain where dredges have operated for a long period of time. Suction dredging can effect aquatic habitats by increasing turbidity, altering channel morphology and bottom substrates that serve as spawning areas for fish, and directly causing the mortality of fish and amphibian eggs and larvae in the sediment vacuuming process. The level of impact varies by site and depends largely on the site's ecological integrity. Suction dredging impacts are most concentrated along low-elevation reaches of major streams (Stephenson and Calcarone 1999).

Management Considerations: Given the significance and rarity of hydrologically intact low-elevation streams, those occurring on public lands should be given special attention. Especially important are stream reaches in a relatively unmodified state with largely undeveloped watersheds, and lacking instream water storage and diversions and nonnative undesirable species. In such areas it may still be possible to maintain historic disturbance regimes and the natural range of variability and populations of native species. Hydrologically unregulated sections of these streams, such as those in the coastal northern Santa Lucia Ranges, are likely to be the best remaining examples of intact low-elevation aquatic ecosystems in the central and southern California coastal region. Thus, they represent the best opportunities for maintaining intact aquatic ecosystems (Stephenson and Calcarone 1999).

Habitat Restoration: The restoration potential for riparian habitats is largely controlled by the ability to restore the natural hydrologic and sediment regimes. Below existing dams, however, there is little ability to recreate large flood and sediment transport events because water storage, flood control, and recreation are usually priorities for the management and operation of such facilities. The importance of dams and diversions in preserving domestic water supplies, controlling downstream flooding, and power production ensures that most instream structures will continue to operate. The amount of land development, the size, type, and maintenance history of road networks, and the effects these have on hydrology, are also important considerations. Nonetheless, there are many opportunities where rather minor changes in the management road networks or water releases could greatly improve habitat capability for species of concern. In many cases, riparian systems with a natural or somewhat natural hydrology regime are very resilient to historic disturbances.

At Piru Creek on the Los Padres National Forest, for example, spring and summer discharges from Pyramid Lake used to fluctuate on a daily or weekly basis from 0–150 cubic feet per second (cfs) (0-4.25 cubic meters per second [cms]) and arroyo toad clutches and larvae were often stranded or swept away. In 1992, a shift to constant releases during the spring/summer period resulted in a large increase in larval arroyo toad survival (Stephenson and Calcarone 1999).

Another example is to release large spring scouring flows during wet years to mimic natural flood-related disturbance events. Resource management agencies should pursue opportunities to work with water and flood control agencies to address flow release issues (timing, duration, and volume of water releases) on a site-specific basis. In particular, efforts should be made to limit the practice of rapid, large changes in the volume of spring and summer water releases that adversely affect special-status and native species (Stephenson and Calcarone 1999).

The USDA Forest Service has developed a database that provides baseline information on watersheds and the major streams occurring within them. Information compiled for each stream includes linear miles by elevation intervals, percent on public land, occurrence of roads along the stream corridor, land use intensity, degree of streamflow alteration, and level of infestation by nonnative undesirable species. Information on each watershed includes the percentage of the entire watershed that is on public land, road densities in the watershed, and occurrences both of species of concern and of nonnative undesirable species. This information can be used to develop an index of riparian system integrity and restoration potential on National Forest System lands in southern California (Stephenson and Calcarone 1999).

A number of private and public organizations (including the USDA Forest Service) have formed alliances (e.g., Team Arundo) to coordinate efforts in controlling the spread of arundo. Considerable progress has been made in the refinement of techniques for effective arundo eradication (Stephenson and Calcarone 1999).

Conservation: All four southern California National Forests have endorsed the *Interim Riparian Management Guidelines* (USDA Forest Service 1999). The USDA Forest Service has also proposed to implement additional measures to obtain information to recover, minimize, or eliminate potential impacts on the listed riparian species. A detailed map of riparian plant communities would be extremely useful for management purposes. These additional actions include special-status species surveys, habitat assessments, monitoring, removal of unnecessary system and nonsystem roads from riparian habitats, maintenance, repair or removal of upslope roads where inadequate drainage systems are affecting downslope riparian habitats, and interpretive and education activities (USDI Fish and Wildlife Service 2000).

All four southern California National Forests are also undertaking reasonable and prudent measures to avoid or minimize incidental take of federally listed riparian species. These measures include avoiding or minimizing impacts on existing riparian habitat that could result from routine trail and road use and maintenance, use of developed recreation sites, special use permit activities, grazing activities, administrative sites, and road construction (USDI Fish and Wildlife Service 2000).

The USDI Fish and Wildlife Service has provided detailed terms and conditions to implement reasonable and prudent measures with which the USDA Forest Service must comply during the undertaking of management actions in order to be exempt from the take prohibitions in Section 9 of the ESA for federally listed riparian species. Section 7(a)(1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information. The USDI Fish and Wildlife Service (2000) recommended the following conservation actions for riparian habitats and associated listed species:

- Additional enforcement activities.
- Evaluation of the relocation of developed facilities.
- Construction of viewing platforms or other low-impact uses in and around developed recreation sites.
- Production of additional public information and educational/interpretive materials and programs.
- Temporary relocation of native species during ground-disturbing projects.
- Minimizing the invasion of nonnative undesirable plant species.
- Consideration of the closure of all, or portions of, existing developed sites or areas, when other actions fail to provide needed protection to listed riparian species.
- Consideration of the feasibility of conducting an aquatic exotic species removal at the Province level.
- Conducting Province-wide monitoring surveys that would provide additional information in determining the extent and amount of impacts that activities are having on listed riparian species.

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Serpentine Outcrops

Description: Serpentine outcrops are a habitat in which the composition and structure of the plant community is strongly determined by the mineral composition of the soil (Kruckeberg 1984). Serpentine soils are derived from serpentinite, a mineral recognized by its waxy texture and colors that range from green to blue to red. Serpentinite is a type of ultramafic rock, so called because of the high concentration of *mafic* minerals such as magnesium, iron, nickel, chromium, and cobalt (Stephenson and Calcarone 1999).

Serpentine soils typically have a low concentration of calcium, limiting the ability of plants to uptake essential soil nutrients. Serpentine soil often has a clayey texture and a higher water-holding capacity than adjacent soils. Soil texture, pH, and other factors can also limit nutrient uptake. Accordingly, serpentine soil is considered impoverished of nutrients and supports only those plants adapted to or tolerant of its unique chemistry (Marschner 1995).

Most of the research on serpentine habitats has focused on plant physiology and nutrition or on rare plants and serpentine endemics (Martens 1989). In general, however, there is limited work describing serpentine-associated vegetation. Variations of grassland, chaparral, oak woodland, and lower montane mixed-conifer forest all occur on serpentine, but species richness, vegetation density, and plant growth and stature are lower than on adjacent soils (Kruckeberg 1984, Hanes 1988, Vogl et al. 1988, Stephenson and Calcarone 1999). Chaparral on serpentine has relatively large intershrub spaces that are poorly vegetated or have some cover of grasses and herbs but little leaf litter (Hanes 1988). Serpentine habitats also support unique species associations and endemic species (Kruckeberg 1984).

Serpentine habitats are often recognized by a conspicuous shift in vegetation type (Martens 1989). In areas dominated by grasslands, serpentine tends to support chaparral. In chaparral areas, serpentine often supports sparse grassland vegetation. Oak woodlands typically shift to chaparral or grasslands on serpentine. Coniferous forests and mixed-evergreen forests generally become more open but retain conifer dominance. Extreme serpentine habitats are referred to as *barrens* because they support little or no vegetation. Less toxic sites can support up to 215 species and varieties of plants and at least nine butterfly taxa (Stephenson and Calcarone 1999).

Some tree species, such as Sargent cypress (*Cupressus sargentii*) and knobcone pine (*Pinus attenuata*), occur on a variety of substrates, but are reliable indicators of serpentine soil because they assume greater dominance as a result of their ability to tolerate serpentine soils. Because of their ability to tolerate serpentine soils, thereby avoiding competition from other plant species, some species have elevational or geographic range extensions on serpentine soils (Axelrod 1988).

Distribution and Abundance on National Forest System and Adjacent Lands: In California, serpentine outcrops occur sporadically in bands throughout the Coast Ranges and the foothills of the Sierra Nevada. Serpentine outcrops occur on National Forest System lands primarily in the Santa Lucia Ranges and the southern Los Padres region on the Los Padres National Forest. A small, isolated patch of serpentine habitat occurs on the northern Santa Ana Mountains on the Cleveland National Forest (Stephenson and Calcarone 1999).

An estimated 31,470 acres (12,736 hectares) of serpentinite-derived soil occur on the Los Padres National Forest. The Cachuma Saddle area and Figueroa Mountain, both in the San Rafael Range, contain serpentine chaparral and woodland, including groves of Sargent cypress. Serpentine woodland and Sargent cypress occur again at the 1,334-acre (540-hectare) Cuesta Ridge Botanical Area in the southern Santa Lucia Range. Further north, the Chew's Ridge area in the Santa Lucia Ranges supports good examples of serpentine grassland and woodland (Stephenson and Calcarone 1999). Serpentine grassland is also found in the Pine Ridge area of the Santa Lucia Ranges (Kruckeberg 1984).

Ecological Processes: Vegetation communities on serpentine soils tend to be relatively sparsely vegetated or dominated by plant species adapted to periodic fire occurrence, such as chaparral, closed-cone pines, and cypress; in either case, such communities are resilient to fire occurrence. Changes in the intensity or frequency of fire, however, can alter species composition. Several stands of closed-cone pines and cypress, for example, have been reduced on the four southern California National Forests because of more frequent fire occurrences in recent decades (Dunn 1987, Stephenson and Calcarone 1999). High-intensity fires during severe fire weather, which are typically human-caused, can limit postfire regeneration from stump sprouting or from the seed bank and shift species composition, favoring grasses and herbs (Stephenson and Calcarone 1999). Postdisturbance revegetation on serpentine tends to be slower than that of chaparral on other soil types because of the slower growth rates characteristic of serpentine communities (Hanes 1988).

Factors that Influence Ecological Processes: Historic mining has removed some areas of serpentine habitat. Serpentine is an indicator of economic metals. Mercury, chromium, nickel, magnesite, asbestos, talc, soapstone, and jadeite are all found in association with serpentine and other ultramafic outcrops. A number of historic and active mines occur in the Santa Lucia Ranges, and potential exists for mining activities to adversely affect serpentine habitat on the Los Padres National Forest (Stephenson and Calcarone 1999).

In general, the relatively sparse vegetation cover results in reduced grazing pressures and fuel and timber management practices in serpentine habitats. In several areas, however, unauthorized off-highway vehicle use and other recreation uses have removed vegetation and caused soil erosion in serpentine habitats (Stephenson and Calcarone 1999).

Management Considerations: Serpentine habitats are naturally small in size and have an inherently fragmented and disjunct distribution pattern. In addition, plant species typical of serpentine habitats exhibit high levels of local adaptations and between-population variability, so management or restoration of serpentine habitats should focus on conserving site-specific resources. Consequently, in situ conservation of serpentine habitats and populations of special-status species is recommended.

Ten special-status plant species on National Forest System lands in southern California are indicators of or endemic to serpentine soils. Of these, nine are USDA Forest Service Region 5 Regional Forester's Sensitive Species that occur on the Los Padres National Forest. The population status of most of these plants is unknown or stable, and they are mostly considered to have low vulnerability on National Forest System lands. More information is needed on the floristic composition of serpentine outcrop plant communities, on the population status of sensitive plants endemic to serpentine soils, on wildlife use of serpentine outcrops, and on specific management measures for serpentine outcrops on National Forest System lands in Southern California (Stephenson and Calcarone 1999).

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Vernal Pools

Description: Vernal pools are small, isolated, rain-filled depressions that hold standing water for a period of time following winter and spring precipitation. These seasonal wetlands typically form in small depressions in flat grasslands and are underlain by an impermeable substrate that prevents the infiltration of soil moisture. Most vernal pools have a small surrounding watershed that is not connected to riparian drainages. As temperature increases and precipitation decreases in the spring, surface water is lost through evaporation. By late spring, most vernal pools have dried, and through the summer they typically appear as dry, unvegetated shallow depressions.

Classification of vernal pool habitats is generally based on subtle variations in hydrology, such as the size, depth, temperature, seasonality, and duration of surface water. Geographic location, substrate, and the surrounding vernal pool watershed largely control the hydrologic regime.

A suite of plant and invertebrate animal species is endemic to vernal pools in California. These species are adapted to the specialized hydrology and seasonal growing conditions in vernal pools. Species composition of vernal pools varies with hydrology, geographic location, surrounding habitat types, and the degree of isolation. Water chemistry, especially alkalinity, total dissolved solids, and pH, are some of the most important factors in determining the distribution of and habitat suitability for invertebrate species (Eng et al. 1990, Eriksen and Belk 1999). Many endemic plant species are annuals that germinate below standing water or in saturated soil on pool margins, then quickly flower and set seed as water levels drop. Similarly, the life cycle of several small invertebrate species is tied to the few weeks or months each year when the pool has water; these species persist most of the year as hardened cysts.

The vernal pools on the four southern California National Forests have not been fully surveyed or described. Despite the lack of site-specific information, it is possible to infer some likely habitat parameters based on the species identified on some of the pools. Identified species include vernal pool fairy shrimp (*Branchinecta lynchi*) (federally listed as threatened), Conservancy fairy shrimp (*B. conservatio*), and longhorn fairy shrimp (*B. longiantenna*).

Pools occupied by vernal pool fairy shrimp are often alkaline pools with grass or mud bottoms and clear to tea-colored water. Vernal pool fairy shrimp also inhabits ephemeral drainages, rock outcrop pools, ditches, stream oxbows, stock ponds, vernal pools, vernal swales, and other seasonal wetlands. In southern California, vernal pools may occur in potreros: small, isolated grasslands on fine-textured or clayey soils in a chaparral-dominated landscape.

Vernal pool fairy shrimp average 41 days to reach maturity; consequently, they normally inhabit vernal pools in which water remains ponded for more than 6 weeks (Eng et al. 1990, Eriksen and Belk 1999). Occupied habitats range in size from rock outcrop pools as small as 1 square yard (0.84 square meter) to pools as large as 11 acres (4.5 hectares); the potential ponding depth of occupied habitat ranges from 1.2 to 48 inches (0.03 to 1.22 meters) (USDI Fish and Wildlife Service 2001).

Vernal pool fairy shrimp and other fairy shrimp species have also been observed in other types of depressions where water ponds. In addition to swales, fairy shrimp species can occupy artificial habitats that are partially or completely unvegetated; these include railroad toe-drains, roadside ditches, abandoned agricultural drains, ruts left by heavy construction vehicles, and depressions in fire breaks (Eng et al. 1990). Vernal pool fairy shrimp have also been found in water pooled in sandstone outcrops. Vernal pool fairy shrimp are not found in riverine, marine, or other permanent waters (50 Federal Register [FR] 48136-48153, September 16, 1994).

Longhorn fairy shrimp is only known from alkaline and grassy-bottomed vernal pools and pools in sandstone rock outcrops (Eng et al.1990, Eriksen and Belk 1999). Potential habitat may also occur in other depressions that hold water of a similar volume, depth, and area, and for a similar duration and seasonality as vernal pools. In general, longhorn fairy shrimp is found in pools lacking other fairy shrimp species, although the taxon has been reported occasionally to co-occur with vernal pool fairy shrimp (Eriksen and Belk 1999).

Distribution and Abundance on National Forest System Lands and Adjacent Areas: Current information suggests that vernal pools on National Forest System lands are limited to the Los Padres National Forest (Stephenson and Calcarone 1999). An estimated 751 acres (304 hectares) of occupied federally listed fairy shrimp habitat, including vernal pools, may occur on National Forest System lands (USDI Fish and Wildlife Service 2001). The known or reported locations include a small pond in grassland habitat on Cuesta Ridge on the Santa Lucia Ranger District, a few potreros in Cachuma Canyon on the Santa Barbara Ranger District, and two vernal pools in montane meadows on the Mount Pinos Ranger District.

There are no known occurrences of longhorn fairy shrimp on National Forest System lands in southern California. The nearest known occurrence is in Soda Lake on the Carrizo Plain of San Luis Obispo County (Eriksen and Belk 1999). There is potential for longhorn fairy shrimp to occur in potrero habitats in the Sierra Madre Mountains on the Los Padres National Forest (Stephenson and Calcarone 1999, USDI Fish and Wildlife Service 2001).

Vernal pools and similar seasonal wetland habitats constitute a naturally rare habitat type. While they occur throughout California and are most abundant in the Central Valley, nowhere do they occupy large, contiguous areas of the landscape. Because of mountainous topography, vernal pools and vernal pool complexes are especially rare in the Santa Lucia, Transverse, and Peninsular Ranges.

Ecological Processes: Ecological processes and patterns of species composition in vernal pool habitats are controlled by hydrology, degree of isolation, and dispersal mechanisms. Variations in precipitation, heat, and wind influence the amount, seasonality, duration, and temperature of surface water in the small, closed basins. The populations of individual species in any given pool can vary with hydrology.

Most species associated with vernal pools, however, have developed adaptations, such as long-lived seeds or hardened cysts, to withstand high levels of interannual variability in habitat quality or multiple years of inadequate habitat conditions.

The primary historic dispersal method for some vernal pool species likely was large-scale flooding, which allowed species to colonize different individual vernal pools and other vernal pool complexes (USDA Fish and Wildlife Service 2001). This dispersal mechanism no longer exists in most watersheds because of the development of water storage and diversion structures (Stephenson and Calcarone 1999).

Currently, the main mechanism of distributing fairy shrimp populations is consumption of fairy shrimp cysts (resting eggs) by predators, especially waterfowl, which subsequently expel viable cysts in their excrement at other sites (Wissinger et. al. 1999). The membranous layers of the cyst are functionally impervious to enzymes in the predators' digestive system. If conditions are suitable, these transported cysts may hatch at the new location and potentially establish a new population. Cysts can also be transported in mud carried on the feet of animals, including waterfowl and livestock that may wade through vernal pools. (Eriksen and Belk 1999.)

Factors that Influence Ecological Processes: At least 90% of historic vernal pool habitat in California, and up to 98% in southern California, has been directly lost because of conversion to agricultural uses and urban development. Although developmental pressures continue, only a small fraction of remaining vernal pool habitat in California is protected in reserves or on public lands. Many individual vernal pool species are now threatened by stochastic extinction as a result of the small and isolated character of remaining populations (USDI Fish and Wildlife Service 2001; 50 FR 48136-48153, September 16, 1994).

Activities that change the ponding duration, alkalinity, and pH of vernal pools can adversely affect some species. These activities include damaging or puncturing the underlying hardpan layer, placement of fill in the vernal pool, introduction of nonnative undesirable plants, and destruction or degradation of upland habitats that contribute runoff to vernal pools (USDI Fish and Wildlife Service 1996, Eriksen and Belk 1999).

The introduction of nonnative undesirable fish into vernal pool habitats also threatens the survival of native invertebrates. Natural or agricultural flooding can introduce fish into otherwise isolated vernal pool habitats. Opportunistic fish, such as mosquito fish, consume fairy shrimp and can eliminate populations (USDI Fish and Wildlife Service 1996).

Potential ongoing threats to vernal pool habitats on National Forest Service lands include livestock grazing, off-highway vehicle and mountain bike use, runoff and sedimentation into vernal pools from projects within the drainage area, invasion of brushy plant species or nonnative undesirable plant and animal species, and disturbance associated with general recreational use. Excessive sedimentation can fill pools, and invasion of brushy or nonnative undesirable plant species can degrade water quality and alter the natural cycles of filling and drying of pools. Other activities that adversely affect vernal pool habitats include mosquito abatement measures, pesticide/herbicide use, gravel mining, and contaminated stormwater runoff. Most of these impacts can be particularly damaging when pools contain water. (USDA Fish and Wildlife Service 2001.)

Livestock Grazing: Livestock grazing can affect vernal pool habitats by trampling, soil compaction, increased sedimentation, and increased eutrophication from livestock defecating in the pools. In general, livestock use in vernal pool areas has not been shown to cause any substantial effects on populations of fairy shrimp, although individual animals may be lost to trampling. In some instances, presence of livestock may benefit vernal pool organisms by assisting in dispersal or by preventing nonnative undesirable grass species from dominating the vegetative cover in pools (USDA Fish and Wildlife Service 2001).

Some of the vernal pools occupied by listed fairy shrimp species on National Forest System lands are located within active grazing allotments; these pools have been fenced to exclude cattle from the pools. No other activities are likely to affect these ponds with the exception of occasional dispersed use by hunters, who are unlikely to walk through the pools (USDI Fish and Wildlife Service 2001).

Fire: Prescribed burning or fuelbreak construction typically would not directly affect fairy shrimp because the relatively flat grassland or scrub habitats where vernal pools occur are not areas typically targeted for fuel management. In general, vernal pool plant communities can readily recover following fire, especially if the fire occurs when the pool is dry and after seeds have set on annual plants. Seeds from vernal pool plants can survive fire occurrences because the moister conditions and low biomass in the pools result in less complete and less intense fires. Maintenance of the soil seed bank promotes relatively rapid regeneration during the subsequent wet season. Perennial vernal pool plant species can regenerate vegetatively following a burn. However, vernal pool habitats and species are vulnerable to any ground-disturbing suppression and prevention activities, especially when the soil is moist, and to drafting of water for fire suppression activities (Bauder and Weir 1991, Pollak and Kan 1998, USDA Fish and Wildlife Service 2001).

Management Considerations: The *in situ* conservation of vernal pools and their associated watersheds should be a high management priority. The historic and ongoing loss of vernal pool habitats and the small number of extant vernal pools protected in reserves or on public lands increases the importance of conserving vernal pool habitats on National Forest System lands.

Vernal pools have a naturally isolated and fragmented distribution. On National Forest System lands, the degree of separation from other vernal pool communities is particularly extreme. Isolated populations are more susceptible to inbreeding depression, which can result in local extinction or reduced species fitness. Because of the limited and disjunct distribution of vernal pools, as well as the even more limited distribution of associated federally listed species, any reduction in vernal pool habitat could have a substantial adverse effect on the availability of this habitat type and on populations of federally listed species. (USDI Fish and Wildlife Service 1996).

Currently, basic information is lacking on the type, size, distribution, and species composition of vernal pool habitats on National Forest System lands. Vernal pools and their watersheds should be mapped and evaluated to facilitate the management and conservation of their integrity and their natural hydrologic regimes (Stephenson and Calcarone 1999).

Prescribed burning in vernal pool watersheds can be effective tool for managing nonnative undesirable species and promoting restoration of native plants. Burning after native vernal pool species seeds have matured but before the maturation of nonnative plants could result in the reduction of nonnative seeds from the seed bank and the increase of native plant species populations (Bauder and Weir 1991, Pollak and Kan 1998).

In consultation with the USDI Fish and Wildlife Service (2001), the four southern California National Forests have adopted new management direction measures to minimize effects on vernal pool habitats occupied by federally listed fairy shrimp species. The new measures include those listed below.

Prohibit activities (e.g., construction, road maintenance and use, water diversion) that alter the hydrology or cause sedimentation of key and occupied vernal pool habitats.

Include resource advisors as part of fire incident command teams to make recommendations on how to protect listed fairy shrimp and vernal pools, and to brief crew supervisors and equipment operators on locations of listed fairy shrimp in suppression areas.

Prohibit establishment of staging areas, helibases, base camps, fuel breaks, or other areas of human concentration and equipment use within listed species' key and occupied habitats.

Prohibit drafting or dipping water from vernal pools.

Emphasize prevention of establishment and spread of nonnative undesirable plant and animal species in listed species' habitats.

Include provisions in contracts and permits for use of National Forest System lands and resources as necessary to prevent the introduction and spread of nonnative undesirable plants in listed species' key habitats.

If listed fairy shrimp are detected in a recreation area, take steps to avoid or minimize negative impacts on the species.

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