PART I – FIRE MANAGEMENT PROGRAM

1.0 The Planning Context

This Section discusses a framework of principles, concepts, processes, relationships, and methods that may be useful in implementing long-term fire management within the Southern Subregion. This framework places planning procedures within a broad, proactive management approach that considers societal values and the protection of biophysical components of ecosystems at the earliest stages of NCCP/MSAA/HCP and fire management program design.

The management approach suggested here is guided by four broad principles:

- Ecosystems are dynamic, evolutionary, and resilient;

- Ecosystems should be viewed spatially and temporally within organizational levels;

- Ecosystems have biophysical tolerances and social limits (society’s willingness to financially support corrective actions); and

- Ecosystem processes are not completely understood and, therefore are not fully predictable.

Clearly, ecosystems are dynamic and change with or without human influence. Existing conditions are a product of natural and human history. Although ecosystems are dynamic, there are limits to their ability to withstand change and still maintain their integrity, diversity, and productivity. Our efforts are guided by an ever-increasing understanding of how large ecosystem patterns and processes relate to smaller ecosystem patterns and processes, however, there are limits to our ability to predict how ecosystems may change. These principles suggest the need for an adaptive approach to fire management, one that can be adjusted in response to new information and changing conditions.

Long-term ecosystem management requires completion of the following tasks:

- Establishing measurable goals and objectives;

- Assessing resources at multiple resolutions and geographic scales;
- Defining a strategy for implementing decisions;

- Formulating and carrying out a monitoring program to evaluate the outcome of decisions; and

- Formulating and implementing adaptive management approaches.

1.1 Land Management Goals

In their 1998 report, Southern Subregion NCCP Science Advisors acknowledged and supported a landscape/natural community focus for fire management as the scale most likely to produce success for this conservation effort. To implement fire management at this scale, it is important to set broad overall goals for land management. It is the intent of the Southern Subregion NCCP/MSAA/HCP Habitat Reserve Management Program (HRMP) Adaptive Management Program (AMP component) that lands set aside in any future Habitat Reserve be managed in accordance with a fire management program that is ecologically based and designed to achieve the following overall goals:

- Ensure the persistence of a native-dominated vegetation mosaic in the future Habitat Reserve.

- Restore and enhance the quality of degraded vegetation communities and other habitat types in a manner consistent with the overall Conservation Strategy for Identified Species and associated natural communities.

- Maintain landscape functions at all identified scales within the Southern Subregion.

- Protect and manage identified target structural characteristics for selected species and their associated habitats.

Overall goals are needed to point the program in the right direction, but they may not provide sufficient guidance in defining target conditions for specific habitats and management activities on individual parcels. The following sections discuss development of a program designed to achieve the above goals by addressing species and community-specific objectives and conditions.
1.2 Keys to Fire Management

Ecosystem management presumes a working knowledge of system function and structure. Unfortunately, the timeline of management actions typically precedes the development of requisite management knowledge. Many habitats are represented in the Southern Subregion with little history of consumptive-resource value. Vegetation types, like coastal sage scrub, received very little attention from researchers until the late 1970’s (see O’Leary et al. 1994). As a result, existing knowledge relies on experimentation as part of an Adaptive management approach to manage the system, particularly in southern California, where habitat fragmentation and continuing urbanization compound already complex management issues. In this situation, Habitat Reserve managers must learn through carefully designed adaptive management actions as a means of compensating for uncertainty.

A structured model for this learning process directs management and monitoring actions to increase the rate of information acquisitions and improve management in iterative steps (Lee 1993).

The fundamental elements of the Fire Management Program are as follows:

1.3 Management Objectives

- Identify appropriate spatial scales and patterns for the long-term management of fire;

- Develop active fire management prescriptions consisting of 1) Management Objectives, 2) Preparing Management Plans and Models for Shrublands (coastal sage scrub and chaparral) and 3) Identifying Uncertainties For Valley Grasslands; and focused on increasing diversity of native plants and promoting community structure and composition favored by target wildlife species;

- Quantify effects of varying fire regimes on selected wildlife species;

- Utilize prescribed fire to reduce unplanned fire events from known ignition corridors;

- Define fire prescriptions that aid in restoring degraded shrublands and riparian areas;
Quantify active restoration techniques for application following fire treatments; and

Develop public understanding and support for active fire management.

1.4 Preparing Management Plans and Models

Based on the best available information and the objectives described above, the second step in the fire management program is to prepare management plans for designated Southern Subregion sub-units. To do this successfully, some concept of how the natural system works is necessary. Researchers at the Riverside Forest Fire Laboratory have brought together university and agency scientists and managers to collect information on the functioning of natural systems and develop working models of those systems.

Because of the complexity of natural systems, even the best models must provide for flexibility as part of an adaptive management approach. Natural community models represent a set of assumptions or hypotheses, based on current knowledge, that are tested through the application of management techniques. Using a combination of the desired future conditions and natural community models, a working fire management plan is developed.

The current stated goals for the Southern Subregion NCCP/MSAA/HCP indicate a desire to protect natural ecosystems, including those that support coastal sage scrub, grasslands, riparian areas and oak woodlands. It is also the desire of Southern Subregion managers to enhance the currently degraded areas of coastal sage scrub plant communities. In addition, existing valley needlegrass grasslands and oak woodland habitats will be maintained and enhanced with the active use of prescribed fire if grazing is not an option. Fire is not a viable option in riparian zones except in some very specific situations. These management goals define a four-part fire management approach:

1. The reduction of unplanned fire events through the use of maintained firebreaks and strategic prescribed burns (Vegetation Management Projects - VMPs);

2. Implementation of a seasonally and frequency-focused fire regime as part of a management/restoration strategy for valley needlegrass grassland;
3. Careful experimentation, in consultation with the Wildlife Agencies, using fire as part of restoration and management in small plots on sites formerly occupied by coastal sage scrub stands; and

4. Implementation of low to moderate intensity ground fires in the oak woodland habitats on the Rancho Mission Viejo property where undergrowth is too thick and dense for cattle. If a wildfire were to occur under these conditions there is a great chance of a stand replacement fire where the entire ecosystem is set back to zero. Goats could be used as an alternative to prescribed fire to reduce understory vegetation beneath the oaks.

Although the four described fire activities will be distributed throughout the Southern Subregion, areas of focus are clearly defined for the landscape. Much of the grassland burning is likely to take place within the eastern half of the Southern Subregion. Prescriptions targeted at coastal sage scrub restoration and management will occur throughout the Southern Subregion. Chaparral/shrub land restoration will most commonly be directed toward the western half of the Southern Subregion. Oak woodland maintenance will be concentrated throughout the Southern Subregion while fire protection strategies will be deployed in both the southern portion and western half of the Southern Subregion.

These four described fire activities will not be implemented on currently undeveloped lands within the Southern Subregion without the concurrence and participation by the landowner or the agency with jurisdiction for the property within the Subregion.

1.4.1 Identifying Uncertainties and Knowledge Gaps in Management Plans

To create an effective fire management program, it is important to identify the gaps in knowledge about the natural system that may lead to uncertainties about the role of fire's effectiveness in meeting the management plan's desired objectives at an early stage.

Scrutiny of management actions, or in-actions, during the implementation of the proposed fire plan may determine the need for more basic research. For example, we may not know what happens to a natural community if prescribed fire is applied too frequently or the fuels are allowed to accumulate due to a no-action decision and a wildland wildfire occurs.
The purpose of identifying gaps in models and knowledge is to translate them into a set of questions (hypotheses) that can be addressed through experimentation, monitoring and research. This experimental approach to management recognizes the limitations of current knowledge about natural communities and encourages and provides opportunities to improve management efforts. As knowledge gaps are identified and hypotheses tested, our models and management decisions will improve, new gaps in knowledge will emerge, and additional questions and hypotheses will be developed and tested.

1.5 Site Description

1.5.1 Location

The 132,000 acre Southern Subregion Study Area is located in the south coast ecoregion of southern California. The Southern Subregion lies entirely within the boundaries of the County of Orange. Thirty (30) percent of the entire Southern subregion (about 40,000 acres) is located within the Cleveland National Forest (CNF) while about 92,000 acres are located outside the CNF.

As shown in Part IV, Figure 3, the Southern NCCP/MSAA/HCP study area includes the southern portion of Orange County from the coast inland to the boundary with the counties of Riverside and San Diego. Along the coast, the Subregion extends from the mouth of San Juan Creek in the City of Dana Point to the San Diego County boundary, in the City of San Clemente. The Subregion is bounded on the west and southwest by the Central and Coastal NCCP Subregion, where a separate NCCP/MSAA/HCP was prepared by the County and approved by USF&WS and CDFG in 1996.

Starting at the southwest corner, the boundary of the Southern Subregion:

- extends from the mouth of San Juan Creek along the Creek inland to Interstate 5;
- northwest along Interstate 5 to El Toro Road;
- north along El Toro Road to the intersection of Live Oak Canyon Road;
- northeasterly on a straight line from the El Toro/Live Oak Canyon intersection to the northern apex of the boundary with Riverside County; and
• along the San Diego and Riverside county boundaries, southerly to the Pacific Ocean.

1.5.2 Land Use Classifications

1.5.2.1 Protected Open Spaces

All undeveloped natural areas, including agricultural and disturbed areas, located within the Southern Subregion study area but outside of the CNF were evaluated during preparation and approval of the NCCP/MSAA/HCP. Within the 92,000-acre portion of the Subregion located outside of the CNF, referred to as the Planning Area, 36 percent (about 33,000 acres) has already been urbanized. Another 6 percent (about 5,500 acres) has been used for agricultural purposes for decades or has been significantly disturbed by other uses. Natural habitats comprise 58 percent (about 53,500 acres) of the remaining non-CNF area (see Part IV, Figure 6 for a map of the currently protected open space areas). The natural communities that are subject to potential development pressure include, but are not limited to, coastal sage and other sage scrub communities, chaparral, oak woodland and forest, riparian, wetlands and remnant annual and perennial native grasslands.

Coastal sage scrub habitat constitutes about 39 percent of the existing natural lands remaining in the Southern Subregion outside of the CNF. A total of 20,994 acres of coastal sage scrub is embedded within the 53,500 acres of natural communities in the 92,000-acre planning area and 16,811 acres of that total is located in Subarea 1 (about 43 percent of the natural lands in Subarea 1). Coastal sage scrub is a naturally fragmented and dispersed plant community embedded within a mosaic of non-coastal sage scrub vegetation communities, including 8,454 acres of chaparral, 16,487 acres of remnant native and non-native grasslands/agricultural lands, 4,805 acres of oak woodland and 2,760 acres of riparian habitat.

1.5.2.2 Land Ownership Map

Part IV, Figure 168 shows major landownership within the Southern Subregion. Land owners include the Audubon Starr Ranch Sanctuary, the Cleveland National Forest, County of Orange (Caspers Wilderness Park, General Thomas F. Riley Wilderness Park, O’Neill Regional Park, Prima Deshecha Landfill and unincorporated open space lands), Rancho Mission Viejo, and the cities of Rancho Santa Margarita, Mission Viejo, San Clemente and San Juan Capistrano. Marine Corps Base (MCB) Camp Pendleton is the southerly landowner and is mentioned throughout this document because of the relationship of MCB Camp Pendleton to the fire history of the Southern Subregion.
1.5.3 Natural Resources

1.5.3.1 Topography and Soils

Elevations in the Southern Subregion run from 5,500 feet in the CNF to sea level. The highest elevation outside the CNF is approximately 1,758 feet on the Starr Ranch property. Topography for the Southern Subregion is characterized by rolling hills and a number of ridge systems that run from north to south from the slopes of the Santa Ana Mountains to the Pacific Ocean. Deep south and west facing canyons dissect the landscape resulting in steep slopes with all aspects (north, east, south and west facing slopes) well represented.

A rich mosaic of soil types exists within the Southern Subregion system. The Gabino gravelly clay loam, Soper loam, and Calleguas clay loam are found within the Southern Subregion at the higher elevations. Myford sandy loam, Capistrano sandy loam, Cieneba sandy loam, Alo Clay and Anaheim clay loam occur in the lower portion of the Southern Subregion. The Cieneba Series is the most well represented soil series within the Southern Subregion.

1.5.3.2 Hydrology

Trabuco Creek, a major tributary of San Juan Creek, San Juan Creek and San Mateo Creek comprise the major drainages in the Southern Subregion. With the exception of a few man-made reservoirs, ephemeral drainages and stream courses characterize the landscape described in this plan. Rainfall and the resulting stream flow tend to be highly episodic in nature. For most Subregion streams, active flow occurs between the months of January and May. The upper reaches of San Juan Creek and San Mateo Creek have year round water.

The soils in the Southern Subregion run from non-erosive to highly erosive (see Part IV, Figure 20). Soil erosion can be greatly accelerated following fire events. Factors effecting the type and amount of erosion include erodibility of the soil, steepness of slope, amount and intensity of rainfall, percentage of plant cover, severity of the fire and the length of time since the last burn. With much overland water flow, erosion potential is a significant planning variable to be considered prior to prescribed fire events. There are numerous examples of moderate gully erosion in the existing grasslands throughout the Southern Subregion.

Within units with high erosion potential, pre and post fire management practices will be recommended.
1.5.3.3 Climate

The Southern Subregion exhibits a characteristic Mediterranean climate with warm to hot summers and mild winters. Rainfall is concentrated in the winter. Summer seasons are characterized by extended periods of sunny weather. A strong marine air influence is experienced throughout the year and helps to lower local temperatures. The Southern Subregion’s location, in relation to other significant topographic features such as the Santa Ana Mountains, may intensify the effect that these air masses have on local temperatures.

Local conditions can strongly effect fire behavior. The unique Mediterranean climate, with its long dry summer, produces many days of great fire potential (McCutchan 1977). Local Foehn winds know as "Santa Ana's" develop in conjunction with high-pressure systems in the Great Basin. When a low-pressure trough is located along the southern California coast, a strong pressure gradient is found across the southern California Mountains (Schroeder and Buck 1970). The resulting strong winds, along with warm temperatures and humidity sometimes lower than 5 percent, produce very serious fire weather.

Average annual rainfall on lands outside the CNF is 12 to 14 inches. The higher elevation portion of the watershed (typically the headwater areas) usually receives significantly greater precipitation due to orographic lifting effects. In addition, rainfall patterns are subject to extreme variations from year to year and longer term wet and dry (drought) cycles. In nearby Laguna Beach, a 74-year average computes annual rainfall to be 30.20 Centimeters (12.08 inches) between 1928 and 2002. Figure N-1 shows the seasonal distribution of precipitation by month for the period of record. Mean annual rainfall from an inland site (Bell Canyon), a newer weather station located within the Southern Subregion and installed in 1994 appears to have more overall rainfall. However, the results (Figure N-2) are not directly comparable due to the different lengths of sampling periods. The majority of this rainfall is winter storm generated from cold frontal systems originating in the Gulf of Alaska. Although infrequent, thunderstorms derived from warm, wet southern air masses often develop during the summer in late July and August. A typical thunderstorm event, resulting in multiple ignitions throughout southern California, occurred in the last days of August 1998.
5.3.4 Vegetation

Five types of upland vegetation are dominant within the Southern Subregion: grasslands (predominately non-native), coastal sage scrub, oak woodland, mixed chaparral, and riparian (see Part IV, Figures 11 through 16). These plant communities
also represent most of the major southern California upland vegetation types and occur in a mosaic pattern across the Southern Subregion. Topography and soils likely drive this vegetation pattern. Non-native grasses generally occupy lowland valleys, with deep soils. Upland areas, with rocky soils, tend to be dominated by chaparral and sage scrub, with significant cover taken up by non-native grasses.

Due to limited spatial extent or lack of relevance to fire planning, additional vegetation types represented within the Southern Subregion’s boundary have not been addressed in this plan. Rare assemblages that may be impacted by fire events will be identified at the unit plan level. Appropriate protection strategies will also be identified at that time.

1.5.3.4.1 Non-native grasslands: Non-native grassland types occurring within the boundary occupy a significant spatial extent of the vegetation. Many of the acres currently identified as non-native grasslands are likely to have been type converted from southern California bunchgrass prairie, native flower fields or coastal sage scrub. Remnant stands of native perennials are most common in areas where soils retain moisture later into spring and early summer, such as in valley bottoms and lower foothills. When Europeans settled and changed these grasslands for raising introduced cultivated crops and domestic livestock, their methods of animal husbandry encouraged cattle and sheep to graze much more intensively than native animals had done. Domestic animals were allowed to graze throughout the year. Fires were either suppressed or set too often (yearly).

Hendry (1931) suggested that red-stemmed filaree (*Erodium cicutarium*), curly dock (*Rumex crispus*), and prickly-sow thistle (*Sonchus asper*) may have preceded Europeans to California. Burcham (1956, 1957) and Robbins (1940) present evidence that suggest major replacement of native herbaceous plants with introduced annuals occurred in stages beginning in the 1850’s and ending by the 1870’s. Both throughout California and locally, the most intensive sheep and cattle grazing ended by the turn of the 20th Century, and the end of the grazing era was coupled with a devastating drought which likely set into permanence the conversion of native perennial grasslands to dominance by exotic annuals.

In a Mediterranean climate, hot and moist conditions are scarce and decomposition rates are slow. This may lead to a negative feedback loop, where excess thatch can follow a year of high production (Huenke and Mooney 1989). This excess thatch and ground litter has significantly altered seedbed micro-environments. Native plants, such as purple needlegrass (*Stipa or Nassella pulchra*), are more likely to successfully germinate in the presence of bare soil (Dyer and Rice 1999). Annual grass conversion has also taken place in coastal sage scrub. The conversion of native shrublands to
annual grasslands is a more contemporary phenomenon, likely caused by elevated fire frequencies (Keeley and Keeley 1984).

In an extensive study of grasslands in Orange County, native perennial grasslands were found to occupy less rocky soils, with higher clay contents (Keeley 1993). Research indicates that shrublands may have been displaced on rocky sites, while native bunchgrass prairies may have been lost on heavier clay soils. In a study of plant succession of central California transition rates between grassland and more woody vegetation was found to be similar on moderately drained sandy clay to clay soils. Still, burned areas experienced significantly lower shrub invasion on silty clay, to clay soils (Callaway and Davis 1993). These results substantiate findings by Wells (1962) that fire in combination with soil types seems to influence the distribution of vegetation types in the Mediterranean climate areas of California.

Grassland composition within the Southern Subregion boundaries closely resembles sites throughout the state. Dominant species include wild oats (Avena barbata), red brome (Bromus madritensis ssp. rubens), soft chess grass (Bromus mollis), foxtail fescue (Vulpia myuros), and red-stemmed filaree (Erodium cicutarium & E. botrys). Additional common broad-leaf weeds include prickly lettuce (Lactuca serriola), tumbleweed (Amaranthus albus) and black mustard (Brassica nigra). Though these communities tend to have low species richness and a high proportion of their composition made up of alien species, with proper management they still provide important habitat for target species. As a result, these sites will be focused targets for fire management.

Examples of native grassland species are thought to include: Purple needlegrass (Nassella pulchra), California brome (Bromus carinatus), blue wildrye (Elymus glaucus), California oatgrass (Danthonia californica), pine bluegrass (Poa secunda), and fescue (Festuca spp.). Opportunistic Native Annuals (Zedler 1995) usually colonize gaps in the grasslands; they appear before canopy dominance by the annual grasses, or in the gaps between bunchgrasses, reaching their maximum population density after fire.

Most grasslands appear dependent on fire to recycle nutrients, which are used up rapidly by grasses (Mutch 1970). The annual growth of above-ground biomass in grasslands dies back during the dry season and decomposition sets in that locks up nutrients needed for productivity. The use of fire supports the recycling process by unlocking the nutrients in the accumulations of cured biomass. Grasslands need the continual and frequent infusion of nutrients that fire provides.
Seed bank composition in California grasslands is highly skewed toward exotic annual species (Champness and Morris 1948; Major and Pyott 1966; Dyer, Fossum, and Menke 1996; Holl et al. 2000; Alexander 2001). Seed production by annual species substantially exceeds the number of seeds necessary to replace the population (Young and Evans 1989), whereas the establishment of perennial species has frequently been shown to be limited by seed availability (e.g., Peart 1989a; Kotanen 1996; Hamilton, Holzapfel, and Mahall 1999). Furthermore, annual seeds in growth chambers have been shown to germinate earlier and under a wider range of temperatures than native perennial seeds (Reynolds, Corbin, and D’Antonio 2001).

The more abundant and earlier-germinating annual grass species can form dense stands and monopolize resources, thereby restricting the growth and survival of native seedlings (Bartolome and Gemmill 1981; Dyer, Fossum, and Menke 1996; Dyer and Rice 1997; Hamilton, Holzapfel, and Mahall 1999; Brown and Rice 2000). As a result, competitive interactions between native and exotic grasses in California have usually been shown to strongly favor the exotic species, especially in recently established native populations (Dyer, Fossum, and Menke 1996; Dyer and Rice 1997; Hamilton, Holzapfel, and Mahall 1999; Brown and Rice 2000). (Corbin et al. 2004).

The dominant native California perennial bunchgrass, Nassella pulchra, has been shown to benefit from fire due to removal of thatch that obstructs light from the grass bunches, increasing tillering and increasing seed weight (Menke 1992). Perennial bunch grasses differ from annual grasses in that they put much of their energy during their first several years into establishing a well-developed root system that would sustain them through regular summer drought. Their roots penetrate deeply into the soil, providing nutrients and water and holding soil particles firmly in place. This decreases the erosive effects of wind and water. Unlike annual grasses, they don't produce seeds the first year, but as the years continue, produce an abundance of seed at maturity. These seeds drop close to the tufted parent plant, and generally, expansion of remnant stands is limited by the limited dispersal capability of these seeds (E. Kellogg, pers. obs.).

Fire timing and intensity can play an important role in affecting the ratio of natives to exotics. In contrast to the fall, in the spring annual seeds are highly susceptible to fire damage (Keeley 2001). Perennials on the other hand can survive fire by resprouting from their base. Thus, if native perennial grasses and exotic invasive annuals co-occur on a site, spring burns can shift the community composition in favor of the native perennial grasses.
Nassella pulchra abundance generally rebounds during the second postfire year, whereas Danthonia californica is slower to recover (Corbin et al. 2004). Although germination of native grasses increased following fire, there was no detectable increase in native-grass abundance in subsequent years. Similarly, Dyer, Fossum, and Menke (1996) found that the establishment of native perennial grass seedlings was about the same in burned and unburned areas, whereas seedling mortality in burned areas was high. By the third year, the cover of native species relative to that of exotic species was not significantly different in burned areas and unburned areas, probably because of the rapid recovery of exotic annual grasses. (Corbin et al. 2004).

The time of year in which controlled burns are performed may have a significant effect on the impact on grassland species composition. D’Antonio et al. (2001) found that the month in which grasslands were burned significantly influenced native perennial grasses, with growing-season burns (e.g., November–June) having significantly more detrimental impacts on native cover than summer or fall burns. Burn season did not have a strong effect on native forbs or exotic annual grasses.

One can then surmise that if stand composition consists of native annual and perennial grasses, burns should be confined to the summer-fall season (prior to November) to avoid damaging the native annual grasses. However, if the stand composition is comprised of exotic annuals and native perennial grasses than spring burning is most effective at eliminating the exotic annual grasses and perpetuating the native perennial grasses. Burns can also be specifically timed to limit invasive species' seed dispersal. Fires targeting medusa head and yellow star thistle before mature plants dispersed their seeds effectively suppressed these species and their soil seed banks (Pollack and Kan 1998; DiTomaso, Kyser, and Hastings 1999).

Bell (no date) reviewed the effect of burrowing pocket gophers (*Thomomys bottae*) on maintaining grassland soils in southern California. He noted their ability to support plant species diversity, reduce runoff erosion, increase water infiltration through soil aeration, mix nutrients (Spencer et al.1985), and rotate seed banks including that of needlegrass and native forbs (Hobbs and Mooney 1985; Reichman and Smith 1985; Anderson 1987). Gopher activity supports pioneer patch dynamics for native forbs. Bell describes gopher diggings as approaching 30 percent of the needlegrass prairie on the Santa Rosa Plateau, and that these diggings can bring up to the surface from five to 50 tons per acre of soil each year (Cox and Allen 1987; Spencer et al.1985). An individual gopher may bring two tons per acre of soil to the surface each year (Downhower and Hall 1966).
1.5.3.4.2 Diegan Sage Scrub: This series is often referred to as coastal sage scrub, which is better thought of as a collection of series. This approach allows stands of comparable composition to be described across a large geographic range. Much effort has gone into detailed mapping of Diegan Sage Scrub sub-associations within the Southern Subregion boundary. For the purpose of this plan Diegan Sage Scrub will be used to describe sage scrub types throughout the Southern Subregion. Dominant shrub species include; black sage (Salvia mellifera) California buckwheat (Eriogonum fasciculatum), California sagebrush (Artemisia californica), California encelia (Encelia californica), chaparral mallow (Malacothamnus fasciculatum), coast prickly-pear (Opuntia littoralis), laurel sumac (Malosma laurina) and coyote brush (Baccharis pilularis).

The shrubs dominant in this vegetation type have evolved adaptive mechanisms to exploit soil moisture in upper soil horizons during cool winter seasons. Most dominant shrubs in this community are winter-active and avoid the summer drought by shedding their leaves (Mooney 1977). Unlike evergreen sclerophyllous chaparral, sage scrub is characterized by malacophyllous subshrubs whose leaves abscise or shed during summer drought and are replaced by a lesser number of smaller leaves (Westman, 1981, Gray and Schlesinger, 1983).

Major factors influencing plant species distribution and composition include evapotranspirative stress, substrate type, soil nitrogen, and air pollution (Westman 1981c). The community composition of coastal sage scrub has been shown to consist of relatively few dominant shrub species, with the majority of species occurring in the herbaceous understory. Westman (1981b) found that of the 375 species encountered during his study, over 50 percent were herbaceous understory species with rare occurrence throughout the community’s geographic range.

Studies of post-fire recovery of coastal sage scrub indicate that community response varies with differences in geographic location, species composition, disturbance history, aspect, fire intensity and fire interval (Wells 1962). Multiple successional pathways may exist following disturbance events in sage scrub. Cooper (1922) indicated that sage scrub might be successional to mixed chaparral types but this view is not shared by in much of the prevailing literature. Many coastal sage scrub species are considered to be “pioneer species,” which are present in early successional stages following disturbances (Mooney 1977, Zedler et al. 1983). However, the coastal sage scrub community can either be “preclimax” to chaparral or a stable climax community, depending on soil moisture (Hanes 1977), soil parent material, aspect, and disturbance history. As moisture-holding capacity of the soil types can vary, soil type can therefore have a significant effect on the distribution of coastal sage scrub species (Westman 1981).
Nitrogen has been shown to be a limiting factor for chaparral plants, and so soil type also plays a part with respect to nutrient availability (Westman 1981).

1.5.3.4.3 Chaparral: Several mixed chaparral types are represented in the Southern Subregion. The most common of these types may be described as chamisal. The term chamisal is applied to chaparral stands in which common chamise (*Adenostoma fasciculatum*) comprises 80% or more of the total shrub cover (Hanes 1977). Chamise chaparral is a dense, interwoven vegetation 1-2m high at maturity without a well-developed understory (Hotrod 1960, Hanes 1971). Stands within the Southern Subregion often have the additional, low frequency occurrence of the following shrubs: eastwood manzanita (*Arctostaphylos grandulosa*), flat-leaved lilac (*Ceanothus crassifolius*), toyon (*Heteromeles arbutifolia*) and black sage (*Salvia mellifera*).

Re-growth after fire may be slower between different chaparral types. Slower response may be due to poor site conditions such as soil depth, soil moisture, nutrient availability (Horton 1960), and the effects of fire intensity on root crown sprouting and response of viable seed in the remaining duff layer. A rich herbaceous flora is often associated with this community type during the first wet seasons following fire events (Horton and Kraebel 1955). Due to the compressed fire return intervals experienced on this site, the Southern Subregion chaparral tends to be more open and of lower stature then other mature stands of this type.

1.5.3.4.4 Riparian: Riparian vegetation in the Southern Subregion varies from well-developed forest types to shrub dominated series. The riparian vegetation occurring on any given site may be a function of disturbance history and/or edaphic (soil type, texture and drainage) conditions. Structural elements vary greatly between riparian series. Utilization of these sites as habitat is often closely correlated with structural change.

Arroyo willow (*Salix lasiolepis*), black cottonwood (*Populus balsamifera*), sycamore (*Platnus racemosa*), mulefat (*Baccharis salicifolia*), coast live oak (*Quercus agrifolia*) and additional willows (*Salix spp*) often dominate riparian scrub. In a shrub land form many emergent trees may be present. These stands may or may not be dominated by a single species.

Despite their relatively small area, a greater diversity of wildlife depend upon riparian plant communities more than any other habitat. Riparian zones provide habitat and forage for migratory and resident birds, as well as natural wildlife corridors for movement of other species, linking habitat types. Other functions are also performed by these drainages that extend beyond their limited area, such as filtering of sediment and contaminants for water quality protection and groundwater recharge.
Riparian dominants almost universally resprout after fire (Zedler 1995), except in conditions resulting in extreme wildfire intensity. Wildfire often top-kills many of the broad-leaved trees in riparian areas when they burn and they may recover very slowly (Davis et al. 1988). Full tree canopy recovery following wildfires in riparian areas may take decades (Davis et al. 1988). While wildfire intensities may approach extreme levels in riparian fuels, resistance to both ignition and suppression is very high. Fuel continuity of understory vegetation and arrangement (vertical fuel ladders) can be very dense. During periods of extreme fire danger and extended droughts, riparian areas can become trouble spots due to the abundance of dense dry fuel. Long-term smoldering fires are common and can become a nuisance to wildlife and firefighters. Cool ground fires that clear out dead plant material and underbrush can be beneficial to the long term retention of riparian areas during wind-driven wildfires. In contrast, intense fires that eliminate willow thickets can adversely affect the reproductive success of least Bell’s vireo if they occur just prior to or during the nesting season. There is some anecdotal evidence that young stands respond more vigorously to wildfire than sites with a dominance of mature vegetation.

The range of naturally viable, fully functioning riparian woodlands does not express itself in any single, simple form, and can include basal sprouts, suckering, response to natural environmental events such as flooding, shade tolerance, and possible disturbance dependency. There are also historic and current management factors that may beneficially or adversely affect the natural viability of riparian stands. Key indicators of stand condition are overall stand structure, unique stand management issues, and a ranking of factors that might be putting stands at risk.

1.5.3.4.5 Oak Woodland: Additional vegetation types of management concern are those sites dominated by coast live oak (*Quercus agrifolia*). Within the Rancho Mission Viejo portion of the Southern Subregion most of the oaks are found in the valley bottoms/drainages/alluvial plains adjacent to riparian corridors. Dense stands of mature trees occur on raised stream banks and terraces.

The remainder of the Southern Subregion coast live oak woodlands occur on north facing slopes on upper terrace floodplains, near rock outcrops and in shaded ravines where there is ample moisture. Engelmann oaks require less soil moisture and thus tend to be found in more upland conditions. Soils are generally sandstone or shale-derived. A mix of tree, shrub and herbaceous species characterize these types. California coffee berry (*Rhamnus californica*), California sagebrush (*Artemisia californica*), poison oak (*Toxicodendron diversilobum*), beardless wild ryegrass (*Elymus triticoides*) and Miners lettuce (*Claytonia parviflora*) may all be common in the
understory of these stands. In addition, a significant portion of the herbaceous layer of these stands may be composed of non-native annual grasses.

Many mature trees, within the Southern Subregion, have survived even high intensity fire events. Post-fire survival of oaks is facilitated by fire resistant, thick bark and massive root systems that allow fast regeneration of lost canopy (Plumb 1980), even with crown fire. Mature coast live oaks recover rapidly from moderate-severity fire and light-severity fire as little effect on them. Basal sprouting is common (Tietje et al. 2001; Pavlik 1991, and Paysen 1993). After the very hot “Old Topanga Fire” of 1993 the majority of oaks in the study recovered 80 percent of their canopy in two years (Dagit 2002). As the canopy regenerated in oak saplings after this fire, the diameter of their trunks increased (Dagit 2002) apparently benefiting from the post-fire nutrient flush after high-intensity fire (Boerner et al. 1998). However, coast live oak seedlings and saplings less than 3 inches (7.6 cm) in diameter may be topped killed by low to moderate severity fire and severe fire kills trees of this size (Dougherty and Riggan 1982, Plumb 1980, Plumb and Gomez 1983). Trees greater than 6 to 8 inches in diameter resist top-kill. The most common fire damage to the trunk is a basal wound resulting in potential cambium death. Large trees may need up to three growing seasons to basal sprout (Plumb and Gomez 1983). Mortality of oaks from fire is greater when there is a shrub understory or adjacent stand of chaparral. Use of fire is frequently recommended for managing understory vegetation. When oaks grow in dispersed stands of low density and low percentages of canopy closure, the associated vegetation is primarily annual and perennial grasses and this presents little danger to the trees in terms of fuel loading. Grassland fires tend to be fast moving and consequently oaks are not exposed to high temperatures of any duration during wildfires burning in grassland fuels. These open oak woodlands can provide firefighters with wildfire suppression control points. In oak stands with dense understory brush, domestic goat grazing (at a rate of 240 goats/acre/day), in conjunction with prescribed fire, has been used to reduce fuel loading and fuel continuity in dense coast live oak chaparral near housing developments (for examples, see Tsiouvaras et al. 1989).

Engelmann oak is relatively rare compared to coast live oak and its very limited range in southern California makes protection a high priority. Engelmann oaks have a reduced ability to recruit seedlings and saplings compared to coast live oak. The research on seedlings and saplings at nearby Camp Pendleton indicates a higher susceptibility of Engelmann oaks to fire than coast live oaks to recruitment impacts and that these seedlings and saplings are more susceptible to mortality following spring fires than fall fires (Lawson et al. 1997).
Since controlled burns are normally conducted in the spring, consideration needs to be paid towards this sensitivity. Mortality of seedlings and saplings occurs differentially according to their height and position in relation to the mature oak canopy. This was observed on Camp Pendleton by Lawson and others (1997), who documented mortality from low to moderate severity fire of small-diameter coast live oaks and Engelmann oaks over five years. Fire mortality of Engelmann oaks and coast live oak (in a woodland with herbaceous and coastal sage scrub species in the understory) was studied for trees less than 3.9 inches (10 cm) in diameter: of 1,214 small trees surveyed, 531 survived 5 years after fire. Both species survived at about the same rate. In the same location on Camp Pendleton, a light to moderate severity fire in an Engleman oak/coast live oak stand enhanced coast live oak seedling establishment. In the two years preceding the fire there was no establishment; in 5 postfire years 1,118 oak seedlings established, of which 1,025 were coast live oak. In contrast, following a severe wildfire in Ventura County severely-burned sites supported no coast live oak germination from acorns the following spring, while adjacent unburned areas produced new seedlings (Davis et al. 1989).

Many studies have shown oak survival to be inhibited by rodent populations in savannas and grasslands. Acorns are cached by wildlife around rock outcrops. Many native wildlife (such as deer and rodents) eat acorns or seedlings. The abundance native wildlife may affect recruitment. Others (scrub jays and ground squirrels) can facilitate the movement of acorns from under mature trees and caching them in areas suitable for germination. In Serrano Canyon in San Luis Obispo County, rodent abundance was found to be a symptom of the conversion from native perennial grasses and forbs to introduced annuals. Overall seed production in the area increased with the annual species. This allowed the population of seed eating rodents to increase proportionately. While perennials produce seed over a relatively longer period each year (well into summer), annuals produce only one crop of seeds each year mostly in one sort season (late spring, early summer). When annuals dominate a savanna, acorns, which mature when annual seed is long gone, become an important food crop for the rodents, and the recruitment of oaks suffers (http://polyland.lib.calpoly.edu/overview/Archives/derome/grasslands.html).

With these well-documented adaptations of oaks in southern California, it is very apparent that fire has played a major historic role in the development of these oak stands. The relationship between tree and shrub recruitment and other stand development issues continues to need to be examined and documented as burning efforts take place in these vegetation types.
1.5.3.4.6 Rare Plants: Thread-leaved brodiaea (*Brodiaea filifolia*) is found in several locations on the Rancho Mission Viejo lands. Please refer to part IV, Figures 30 and 38 for known locations of this species. This species has been listed as a rare plant, yet it appears to respond well to the effects of fire (note that thread-leaved brodiaea is found in grassy areas of Camp Pendleton that are subject to frequent fire). Generally, thread-leaved brodiaea is found in grasslands and at the periphery of vernal pools at elevations of 100-765 meters (Hickman 1993, Reiser 1994, USWFS 2004).

Brodiaea may also be found in association with coastal sage scrub in Los Angeles and San Bernardino Counties (USWFS 2004). Thread-leaved brodiaea is generally found in open areas on clay soils, soils with clay subsurfaces, or clay lenses in loamy, silty loam, or alkaline soils (USFWS 2004). Other native grassland species sometimes found nearby include blue-eyed grass (*Sisyrinchium bellum*) and needlegrass (*Nassella sp.*) (Hickman 1993, Reiser 1994). Thread-leaved brodiaea does not do well in competition with invasive non-native grasses. Native annual grasses tended to be deeper rooted which favored thread-leaved brodiaea.

Also without a mechanism such as periodic fire or grazing in the grasslands the non-native grasses develop a thatch, or mulch layer over the years that is very detrimental to brodiaea. The primary fire-related threat to brodiaea is from mechanical disturbance associated with fireline construction.

Other Rare Plants: The way to ensure protection of other rare plants when they are not specifically identified is to insure that a full range of habitat conditions exist for their specialized requirements, and that rare habitats are protected such as vernal pools, seeps, etc., especially in relatively flat ground where mechanical disturbance is possible. Managing for structural conditions is one way to prevent having to take expensive measures on species recovery (for both plants and wildlife). Identifying locations where mechanical disturbance is to be avoided is the primary reason for mapping rare plants. Fire itself is only a problem if it is too uniform or too extreme in terms of size, intensity (low or high), season, etc. The most at risk plants tend to be the obligate seeders, which are mostly found in chaparral, and the gap-dependent species which require an open but non-weedy condition in either chaparral or coastal sage scrub. Many rare plants have low dispersal capability, and this could be a problem in very large fires that affect all the populations at one time. A very important threat to rare plants is competition from invasive exotic species. In this case fire can be a benefit when used as a management tool.

Plants in fire adapted ecosystems can be divided up by fire response categories. The following breakdown of life histories used for shrubs and trees is based on Zedler (1997,
Classification of herbaceous species is based on Zedler (1995), Keeley (1994), and Keeley et al. (1985). The use of lichens should also be considered.

- Obligate seeders (plants that produce almost exclusively by seed, such as ceanothus spp.)
- Obligate sprouters (reproduce almost exclusively by sprouting, such as sumac,)
- Facultative seeders/sprouters (commonly reproduce by both seed and sprouts, such as chamise)
- Suffrutescents (short-lived perennial plants which are woody only at the base)
- Intermediate to long-lived canopy dominates of coastal sage scrub
- Stem succulents and cacti
- Herbaceous perennials with underground storage structures
- Herbaceous perennials dependent on seed for propagation
- Opportunistic native annuals (plants that die each year and do not need fire for germination, but instead germinate under many conditions)
- Pyrophyte annuals (plants that die each year and only appear after fire because seeds are stimulated to germinate by heat, smoke or charate)
- Lichens

1.5.3.5 Wildlife

For a full description of the wildlife species present with the Southern Subregion refer to Part I, Chapter 3. Many of the species occurring within the Southern Subregion are typical of coastal Southern California, several clines of species (members of population) typical of inland sites, are also present. In addition, connectivity between remaining populations of some of the region’s wildlife species can still be achieved with the Southern Subregion as an important node in a landscape network. Lastly, populations of selected wildlife species within the Southern Subregion are considered rare and declining. Rare species are distributed throughout the Southern Subregion including federally listed species: California gnatcatchers (*Polioptila californica californica*), least
Bell’s vireo (*Vireo bellii*), southwestern willow flycatcher (*Empidonax traillii*), arroyo toad (*Bufo californicus*), Riverside fairy shrimp (*Streptocephalus woottoni*) and San Diego Fairy shrimp (*Branchinecta sandiegensis*) (see Part IV, Figures 26 through 38).

Some of the Southern Subregion’s other notable amphibian and reptile species include the western spadefoot toad (*Scaphiopus hammondii*), the western toad (*Bufo boreas*), Pacific treefrog (*Hyla regilla*), granite spiny lizard (*Sceloporus orcuttii*), side-blotched lizard (*Uta stansburiana*), western whiptail (*Cnemidophorus tigris*), gopher snake (*Pituophis melanoleucus*), California kingsnake (*Lampropeltis getulus*), and the red diamond rattlesnake (*Crotalus viridis*).

Other selected avian species include the northern harrier (*Circus cyaneus*), the red-tailed hawk (*Buteo jamaicensis*), the California quail (*Callipepla californica*), the San Diego cactus wren (*Campylorhynchus brunneicapillus*) the horned lark (*Eremophila alpestris*), the California towhee (*Pipilo crissalis*), the grasshopper sparrow (*Ammmodramus savannarum*) and the western meadowlark (*Sturnella neglecta*). Use of fire on a planned basis will benefit raptors as the amount of protective vegetative cover is reduced, improving the raptors ability to forage for prey. Fire will also benefit seed and insect eating species such as quail, horned larks, towhees, the grasshopper sparrow and the western meadowlark. Since fire will be kept out of riparian areas the southwestern willow flycatcher and least Bell’s vireo will not be adversely impacted. Any temporary disturbance of the coastal sage scrub will adversely impact California gnatcatchers and if nesting sites of tall (4’ to 5’) groupings of prickly pear (*Opuntia littoralis* and *O. oricola*) and coastal cholla (*O. prolifera*) cactus are eliminated the San Diego cactus wren will be adversely impacted.

Significant mammals, which the fire program could conceivably impact, include California ground squirrel (*Spermophilus beecheyi*), valley pocket gopher (*Thomomys bottae*), desert woodrat (*Neotoma lepida*), and dusky-footed woodrat (*N. fuscipes*). Although all of these species can be found in all habitats in the Southern Subregion to some degree, the dusky-footed Woodrat is often the only species found occupying older coastal sage scrub and chaparral stands with high dead to live fuel ratios, in increasing numbers. Wildland fire fighters over the years, including this author, have observed that as coastal sage scrub and chaparral stands age the number of dusky-footed woodrat nests (large mounds, often two to three feet in height and five to six feet in diameter at the ground, comprised of highly flammable vegetative material and small twigs) increases, almost insuring that plant communities complete combustion via mass ignition. During the hot dry summer and fall fire seasons the spread of many wildland fires is aided by glowing embers drifting away from the main fire and landing in woodrat nests resulting in very hot and intense spot fires up to a quarter of a mile or more from...
the main edge of the wildfire, which contribute significantly to the eventual size of the wildfire when finally contained. In a recent story that appeared in the San Diego Union in February 2004, surveys in younger stands of coastal sage scrub had none to one or two rats nests per acre while older stands of coastal sage scrub vegetation had up to 15 dusky-footed woodrat nests per acre, which greatly increases the probability of a stand replacement fire. Wildland firefighters refer to woodrats as “mother natures pyrotechnicians”.

Although many non-burrowing mammals perish during fires, all the above mammal species respond favorably to fire due to the abundance of new sprouts and germinating seed bearing annual and perennial fire followers. The small mammal population explosion following fire greatly benefits the raptor population and other four footed predators.

The degree to which mammals are successful at surviving wildfire depends both on their mobility and the uniformity, severity, size, and duration of the fire (Wright and Baley 1982). Small mammals usually attempt to escape wildfire by using subterranean shelters. Seed-eating rodents generally do well after fire, since they benefit from an open condition and bare mineral soil. Above-ground lagomorphs are more impacted by fire. Those with surface nests, such as the dusky-footed woodrat with its stick house and the California parasitic mouse which lives in woodrat nests, depend on a mature habitat condition.

More mobile, opportunistic large mammals such as carnivores do well, such as foxes and coyotes. They can benefit from increased levels of prey during the recovery phase. Ungulates must find sanctuary in unburned patches or along the periphery of the fire. Large mammal death is generally rare, but possible when fronts are fast moving, wide, and actively crowning with thick ground smoke (Smith 2000).

Depending on the uniformity and severity of a wildland fire, mammals can experience different indirect effects. Given a patchy burn, small rodents, such as woodrats, can recover quickly and exceed pre-burn levels (Schwilk and Keeley 1998 as referenced in Smith 2000). Mule deer (Odocoileus hemionus) generally prefer to forage in recovering burned vegetation as opposed to unburned areas. This is most likely due to an increase in the soil nutrient availability, and thus an increase in the nutritional value of forage within burned sites. In one study, mule deer were estimated to have densities of 25 per square mile in unburned chaparral, while burned areas that had experienced a stand replacing fire had densities of 56 per square mile (Smith 2000).
1.5.3.6 Cultural Environment

**1.5.3.6.1 Pre-Settlement:** Prior to European contact (ca. mid to late 1500’s), the valleys surrounding the Southern Subregion were inhabited by layers of Native Americans of several ethno-historical traditions (McKenna and Hatheway 1988). The earliest known occupants of coastal Southern California are referred to as the San Dieguito Tradition. This first group of coastal residents date from man’s arrival until the establishment of post-glacial environments. The Encinitas Tradition follows with occupations of the Southern Subregion between 6000 to 3000 BC. Sites in the city of Irvine document an emphasis on grass seed procurement, with abundant milling equipment. A high likelihood exists that these people actively used fire to maintain a grass seed resource. The Campbell tradition then occupied these sites from 3000 BC to 700 AD. It is during this period that acorn-processing technology was developed.

The Shoshonean Tradition begins around 500 AD and continues until the Spanish colonization of California. Following the development of the mission system, the term “Gabrielino” was applied to these people and their ancestors. They lived an intensive hunter-gather existences with permanent or semi-permanent villages along coastal estuaries. According to Bean and Shipek (1978), fauna comprised between fifteen and twenty percent of the subsistence resources.

Though the use of managed fire is not specifically documented for the Shoshonean Indians, much evidence exists for tribes throughout California. Several researchers maintain that there is evidence for almost every tribe in the western United States having used fire to modify their respective environments (Lewis 1993, Steward 1955). Reynolds (1959) demonstrates the use of frequent fire by California tribes to increase the yield of desired seeds, drive game, stimulate the growth of wild tobacco, improve visibility and facilitate the collection of seeds. The combination of anthropogenic burning and natural fire likely created a pre-settlement fire regime characterized by frequent fire of variable intensity.

**1.5.3.6.2 Settlement.** The area currently within the Southern Subregion boundaries was originally settled during the late 1760’s and was primarily used for grazing of cattle and sheep (Hudson 1981). Much of the Southern Subregion is contained within the historic boundaries of Rancho Mission Viejo which extended from Camp Pendleton to Cooks Corner. Cattle grazing and water use are documented from the early 1800’s. It is believed the area experienced light to moderate seasonal grazing and infrequent fire. No evidence of managed fire has been documented for this period (1760 – 1800’s).
Starting in 1860 additional land uses developed in and surrounding the Southern Subregion. Orchards were established, sheep grazing was initiated, while limited limestone and coal mining occurred.

1.5.3.6.3 Recent Land Use. Recent history has seen extensive residential and agricultural development throughout the landscape within and surrounding the Southern Subregion. With the exception of the eastern border of the Southern Subregion, the other three edges of the open space landscape within the Subregion have experienced some level of urban-wildland interface. Much of the contemporary development benefits from thoughtful fire planning. Defensible space and appropriate building materials characterize structures built during the last ten years. Still, the high density of human occupation has resulted in a highly altered fire environment. In addition, managed fire has been used successfully and was most recently used in a series of Vegetative Management Projects (VMP’s) in 1985, 1986, 1987 and 1994 (see Figure N-3).

1.6 Natural and Historic Role of Fire

1.6.1 Natural Fire

Little is known of pre-settlement fire events in the Southern Subregion. Many of the assemblages of plants known to occur in the Southern Subregion have documented fire adaptations. With an atmosphere full of oxygen, a surface stuffed with organic fuels and an endless source of ignition, it seems unlikely that these communities would experience significant fire free intervals (Pyne 1995).

The structural and compositional nature of the pre-settlement fuels also indicates frequent fire. Though grasslands may have their origins rooted in soils and climate, the selective forces of fire (Vogl 1974) likely maintained these communities. The widespread appearance of shrublands would also indicate repeated fire events. The spread of chaparral seems to be based chiefly in the Pliocene. The rapid rise of steep slopes, increasingly dry climates strengthened by Santa Ana winds have created extreme fire conditions. Fires from lightning and volcanism effectively eliminated trees and favored seral shrublands.

Contemporary research supports the concept of pre-settlement fire regimes. Oil drilling core samples taken off the west coast in the Santa Barbara Channel graphically display the frequency of wildland wildfires driven by off shore winds and the resulting airborne ash that settled to the channel bottom providing historical records as stratified charcoal
layers. These charcoal deposits indicate that large fires occurred at intervals of every 40 to 50 years at least since the seventeenth century (Mensing et al. 1999).

Wildfires may indeed have burned more often and just not recorded as a charcoal layer in the channel bottom due to no winds, or a prevailing westerly wind. Wildfires that burned under other than an off shore wind condition would not show up as a charcoal deposit.

Analysis of charcoal layers in lake and ocean bottoms has provided ample evidence of past periodic wildfire frequency (Byrne, 1977). Core samples taken from lake bottoms in various parts of the country illustrate similar evidence of frequent wildfires.

Comparing the size and pattern of fire events on either side of the Alta and Baja California borders, Minnich (1990) describes a regime of frequent, low to mid intensity fire events in the absence of fire suppression activities. In contrast Keeley (1999) describes contemporary fire regimes in Southern California shrublands as characterized by too frequent fire. These compressed fire return intervals are thought to be the source of type conversion and the reduction in shrub densities. Modern day fire ecologists most likely considered both theories to be acceptable; however, type conversion and reduction in shrub densities are more directly related to high intensity wildfires. Frequent, large sized low to moderate intensity wildland fire events (prior to the fall Santa Ana winds) of the past very likely reduced the probability of high intensity wildland fires from occurring on the same landscape. And when wind driven wildfires did happen to occur on the same recently burned (zero to fifteen years) landscape they burned with far less intensity, with less residence time causing minimal damage to soils and the organic layer on top of the soil.

At the turn of the century citizens began petitioning the Federal and State Governments to do something about managing the remaining public lands that were the source for devastating wildfires that burned out settlements and the denuded landscapes that contributed to huge sediment flows and severe flooding of down stream properties, towns and cities. The US Forest Service eventually put a 10:00 AM policy into effect. This policy required that all wildfires be vigorously attacked and extinguished by 10:00 AM the following day. This policy resulted in most wildfires being extinguished with the lone exception of the unsuccessful suppression of wind driven wildfires that accounted for all of the acreage burned. Prior to the 1900’s many of the early season fires burned large areas over several months with low to moderate intensity. Since the 1900’s these low to moderate intensity wildfires occurring prior to the Santa Ana winds were quickly and easily suppressed allowing both living and dead vegetative fuels to accumulate.
In the early 1970's land managers began to realize that there were not as many wildfires, but when they did occur they were causing much more soil damage and chaparral stands were being type converted to very flammable stands of non-native grasses. It has taken about 70 years for managers to realize that well-meaning fire suppression policies only postpone the inevitable and were causing unnatural ecosystem changes. These well meaning fire suppression policies were, in fact, allowing wildland fuels to accumulate to unnatural levels because all of the early season, easy to extinguish wildfires were quickly and efficiently suppressed so that when wildfires did occur they were of very high intensities and were very destructive. These realizations have brought us to the present day prescribed fire policy where fire is viewed as natural part of the ecological balance of wildland ecosystems.

Some have stated that the body of recent literature contradicts the premise that early season wildfires created age class mosaics that altered the effects of late season wind driven wildfires. These skeptics hypothesize that Santa Ana wind conditions result in large fires regardless of age mosaics; however, many experienced wildland fighting personnel believe that this theory is unproven and that it contradicts the experience of most wildland firefighting personnel and fire behavior officers on the firelines of southern California. Age class mosaics may not always affect ultimate size (perimeter outcomes), however, they do result in wildfires of less intensity, less residence time and in less damage to the soil, which ultimately results in a reduction in winter storm damage (flooding) following the wildfire event. Combining plant communities such as coastal sage scrub and chaparral as “shrublands” may result in misleading conclusions when analyzing fire perimeter data. Labeling fires as either Santa Ana wind driven wildfires versus fuel driven wildfires, whereas individual wildfires can be classed as both, depending upon which day a particular wildfire is observed also may be misleading. The seven or eight October 2003 wildfires, including the Cedar Fire, for example, burned under both conditions for many days. The Cedar Fire’s size was unprecedented in the historical record. This trend to larger and more dangerous wildfires in the period for which we have records is not a debatable point.

Given the divergent theories concerning the behaviour of wildland fires, the question is, what is the better approach to the management of our wildlands and the unique resources they support than to come down on one side or the other on this debate (see San Diego Union dated May 12, 2004 entitled “Heated Debate”, Scientists split over the wisdom of fire suppression in brushland areas)? It is best to focus on how to manage the risks of extreme fire strategies, such as complete suppression of all of the wildfires that will most certainly lead to a large-scale wildfire event under extreme burning conditions that takes all of the habitat in a single event versus a series of
controlled short-interval prescribed fires that break up vast areas of continuous wildland fuels and whether management objectives can be met under such an approach.

It is even more effective from a planning standpoint to identify the worst possible case regime (scenario) that combines large fires preceded by severe multi-year drought and followed by large winter storms. This is the scenario we are now currently dealing with in the post 2003 fire storm era. The risk of doing nothing (no fire intervention) is unacceptable because 1) we can and did lose thousands of acres of critical habitat in a single event; and 2) we take the chance of losing species (especially wildlife) that depend on an intermediate condition, have short lives, and have limited dispersal capability, and 3) we take the chance of irreversibly degrading biodiversity when we are faced with unmanaged, exotic dominance of the grasslands; and 4) we risk diminishing returns to biodiversity from a single versus a range of structural conditions in the shrublands. Any uniform fire regime carries risks to biodiversity.

The better approach is to define an appropriate range of regimes that addresses objectives for all the functional groups of species, sensitive species and invasives that are the focus of management, and how objectives for these species are to be achieved in a wildland-urban interface. Most of the dominant species have a wide range of fire regimes they can tolerate. We need to identify the role the Southern Subregion should play in a larger regional context and ensure the resiliency of plant and animal communities to the mix of fire-drought-flood-invasion of exotics-El Nino-global warming regimes that we can anticipate. At the community level, the problem is how to manage for conservation of a sustainable range of community conditions, mixed with concentrations of rare endemics, when the scale of disturbance can be a complete stand-replacement wildfire or even a complete watershed-wildfire. Different species may require different management strategies, some of which are in conflict and mutually exclusive. For example, management for a large, frequent and predictable disturbance that creates resource-rich but competitor-poor patches that favor the life histories of herbaceous species and shorter-lived shrubs (Zedler 1994), and the animals that may depend on them are incompatible with longer lived closed canopy shrub communities. Hot fire that benefits chaparral may be harmful in woodlands occupied by endangered species. As communities shift from post-fire recovery and the processes of competition and drought determine the relative abundance of species, are all species of concern preserved? Can the risk of canopy closure that excludes certain species be balanced with that of disturbance-dependent species, within the limits of their life expectancy and their ability to wait out opportunities for regeneration?

There is agreement that mosaic burning is appropriate in the wildland/urban interface (WUI). The Federal Register also identified wildland/urban interface communities as
those where housing was “within the vicinity” of forests and other wildlands, but did not quantify “vicinity”. In its identification of a WUI, the California Fire Alliance (2001) defined “vicinity” as all areas within 1.5 miles (2.4 km) of wildland vegetation, roughly the distance that firebrands can be carried from a wildland fire to the roof of a house. Stewart et al. (2003) combined these into a quantified, operational definition of WUI as follows: “Interface areas have more than one house per 40 acres (16 ha), have less than 50 percent vegetation, and are within 1.5 miles (2.4 km) of an area (made up of one or more contiguous Census blocks) over 1,236 acres (500 ha) that is more than 75 percent vegetated. Intermix areas have more than one house per 40 acres and have more than 50 percent vegetation. (Note: that the term wildland/urban interface for purposes of this NCCP/MSAA/HCP is discussed and defined in Part I, Chapter 4)

Post fire flora is largely the same in coastal sage scrub and chaparral in that the same mix of fire followers occurs. However, dominants in coastal sage scrub are shorter-lived and more facultative with regard to fire. There are more suffrutescents (short-lived perennials) in coastal sage scrub. More of the dominant coastal sage species are wind-dispersed compared to chaparral species. Examples include California sagebrush (*Artemisia californica*); common buckwheat, (*Eriogonum fasciculatum*); and California sunflower, (*Encelia farinose*). An important objective in coastal sage scrub management is to maintain canopy gap-dependent species such as understory forbs and suffrutescents of coastal sage scrub. Are we managing for a loss of these species when we focus totally on gnatcatchers, which prefer coastal sage scrub canopy closure? Canopy gap-dependent species establishment is an example of a process that is not necessarily cued by fire. Very few plants are capable of dispersing to a fully occupied habitat and maturing without relief, at some stage, from the surrounding competition of established individuals (Zedler 1982). Grime (1979) explained some of the reasons for this. Some species appear to require large gaps in which the effect of competition is minimal, while others are capable of establishment in small gaps in which only limited growth is possible before competition with established individuals becomes significant. Capacity to invade depends not only on a plant’s stress tolerance and competitive ability (Grime 1979) but also on dispersal characteristics, which determine the ability of a species to find all the available habitat. During long fire free intervals, gaps created by senescent shrubs are the likely location where these species become established.

Fire control strategies are not necessarily different in coastal sage scrub versus chaparral. However, fire management strategies would differ for targeted fire intervals in coastal sage scrub versus chaparral. No fire suppression approach will likely be able to ensure the sustainability of coastal sage scrub in southern Orange County. Fire intervals in coastal sage scrub (coastal sage scrub) are probably shortening due to the
ubiquitousness of exotic annual grasses. The preponderance of current information suggests that measures to control aggressive exotic annuals may be more successful than fire suppression in protecting coastal sage scrub preserves (Minnich and Scott, no date).

Fire Adaptation Of Coastal Sage Scrub:

While most coastal sage scrub plants sprout poorly after fire, seedlings quickly become established from a preburn soil seed bank (Salvia apiana, S. mellifera) or germinate from seed dispersed by wind (Artemisia californica, Encelia farinose, Eriogonum fasciculatum). Seeds are generally non-refractory, meaning that they are not dependent on fire for germination. Thus constituent shrub species are capable of continual reproduction by seed, unlike chaparral species. Resprouting species flower the first few post-burn years, providing seeds that germinate in subsequent years, and leading to mixed-aged stands (Westman 1981). Recruitment and growth to maturity is extremely rapid (ca. 10-20 yr) for most taxa (Westman 1982), (Minnich and Scott, no date). Coastal sage scrub has lower shrub cover, higher volatile oil content, greater cover by herbaceous (or understory) species, shorter duration of nitrogen-fixing species, and more marked variation in post-fire sprouting patterns than chaparral (Westman 1981).

Coastal sage scrub is probably resilient to a wide range of fire intervals, from 12-40 years, as evident by the mix of resprouting and seeding capabilities of the component species (Malanson 1984). The lack of fire in sage scrub stands is expected to result in a structural simplification as shrubs age and a closing canopy cover diminishes recruitment, an overall reduction in above ground stand diversity (Westman 1981, Westman 1982, O'Leary 1989, Malanson and O'Leary 1982, De Simone and Zedler 1999), low numbers of above ground annuals and a general absence of nitrogen-fixing organisms (Westman 1982, DeBano and Dunn 1982). Canopy dominants such as lemonade berry would be expected to expand, while California sagebrush may decline in cover due to plant aging. Concentration of dominance by a small number of coastal sage scrub species is at its maximum in older stands (Westman 1981).

Low intensity fires stimulate sprouting of dominants, while hot fires suppress crown sprouting and consequently promote herbaceous flora (Westman 1981). Resprouting shrubs recruit seedlings immediately after fire but also recruit in gaps of unburned stands. Gap creating agents vary (including shallow soils, steep slopes), but animals, especially small mammals, can be important in creating and maintaining gaps (DeSimone and Zedler 1999). In coastal areas, most sage scrub species resprout from below ground root crowns, although there can be substantial seedling germination
(White 1995). This is not the case in inland areas where there is little or no regeneration from sprouting and virtually all recovery is dependent upon seed germination.

Post fire resprouting in coastal sage scrub subshrubs tends to be more successful in younger, rather than in older shrubs and at coastal rather than inland sites (Keeley 1998). Fuel bed characteristics of coastal sage scrub differ from mixed evergreen chaparral both in terms of fuel loading and fuel arrangement. The volatile oil concentrations of coastal sage scrub species are considerably higher than mixed chaparral, which creates a higher reaction intensity per unit of fuel during pyrolysis.

Fuel moisture is lower in coastal sage scrub than chaparral during the summer, but the fuel in coastal sage scrub is less continuous, has lower canopy volume and lower tissue density (Mooney 1977). Prevailing summer onshore winds and coastal fog provide higher relative humidity's that mitigate some of these qualities. California sage brush, *Artemisia californica*, leaves are considered to be highly flammable under dry conditions, as are black sage, *Salvia mellifera*, common buckwheat, *Ergonom fasciculatum* and chamise, *Adenostoma fasciculatum*. Fine fuels, such as the leaves of common buckwheat, and grasses contribute to flammability of these plants and are the most responsive to changes in fire weather. Under hotter, drier, windier conditions (late September-November), coastal sage scrub can create fast spreading and very high intensity wildfires. Once the fine fuels (1 hour and 10 hour time lag fuels dry out, they can sustain very explosive wildfires (Andrews 1986, 1989, Montague, pers. communication). A time lag in fuels terminology refers to a National Fire Danger Rating System ratings classification regarding dead fuels (dead plant material). Subclasses are assigned based on the speed, or time lag, with which these fuels gain and lose moisture content (Biswell 1989), which is roughly related to stem diameter and density. In such fuels, moisture content is controlled primarily by precipitation, relative humidity and temperature.

*Fire Adaptation in Chaparral*

Throughout its range, chaparral occurs in a matrix of coastal sage scrub and oak woodland. Herbaceous species are restricted to canopy openings and, as a result, are uncommon in mature chaparral but can dominate briefly after a fire. Post fire emergence of these species is largely from dormant seeds in the soil, as well as from bulbs, rhizomes and tubers that are stimulated by the fire event. A second pulse of annual herbs often occurs within five years of a fire (S. Keeley 1977), probably corresponding to the first above average annual rainfall. As the community ages, it becomes increasingly dominated by a few species of tall, vigorous crown sprouters.
(Lloret and Zedler 1991, Van Dyke et al. 2001). In areas where oaks or other tree species are located in proximity to mature chaparral, the chaparral shrubs can act as “nurse plants” for tree seedlings, which may eventually overtop and kill their hosts by shading them out (Callaway and D’Antonio 1991). Fire interacts with processes of drought mediated canopy development, production and mortality to affect the stability of community composition (Riggan et al. 2003). Chaparral is generally believed to be resilient to fire return intervals ranging from between 20 and 150 years, with average natural return intervals of 50 to 70 years at least in inland situations (Minnich 1983, Davis and Michaelson 1995, Conrad and Weise 1998, Mensing et al. 1999). The degree to which fire can influence community structure in Mediterranean shrublands depends most strongly upon the fire interval (Hobbs and Huenneke 1992, Keeley 2001, Zedler et al. 1983). The natural fire interval in southern California chaparral is believed to be greater than 20 years (Keeley 2001).

Keeley (2001) observed that chaparral is particularly immune to alien invasions because exotic herbaceous growth forms cannot establish under the closed canopies of native shrubs. Non-native species are not common in intact chaparral stands, but shallow soils, erosion or other disturbance can provide exotics with a foothold.

A fire aided invasion process has been documented by Minnich (Minnich and Dezzani 1998), who examined aerial photographs of the mountains in western Riverside County taken between 1931 and 1995. In this area, which burned more frequently than the county wide average, the images showed that chaparral and sage scrub were gradually displaced by alien grasses as the fire rotation interval fell from 30 years to 8 years. Minnich noted that chaparral coverage in the study area dropped from 1,137 acres to 109 acres made up of “fragments fortuitously skipped by fire”, while coastal sage scrub habitat fell from 2,577 acres to 1,137 acres.

Fire intensity can also affect stand composition because of the diverse cues through which vegetation may respond to fire; blazes of different intensities or degrees of smoke production may result in different plants dominating the post fire recovery. In intact chaparral, most wildfires are naturally high intensity and these fires facilitate the regeneration of the natural stand by scarification of the seed coats of the seeds stored in the soil. In areas where chaparral is disturbed from high fire frequency, low intensity burns can fail to produce the heat needed to scarify seeds in the seed bank or to destroy seeds of some non-native annuals when their water content and carbohydrate storage condition make them most vulnerable to being killed (Moreno and Oechel 1991). Nitrogen oxides, which are also important components of air pollution, are the chemicals in smoke responsible for germination of some chaparral species.
Species that resprout vigorously and also produce seedlings after fire are the most wide spread in chaparral, but seedlings alone seldom contribute to substantial numbers of individuals that ultimately become members of the mature chaparral community (Hanes 1977; Zedler 1995). The primary advantage of root crown sprouting is that these large, well developed root systems allow these shrubs to gain competitive dominance of the site within a few years following the fire. The resprouts will eventually out compete the seedlings for light, moisture and nutrients as the canopies of the resprouts expand, causing the gradual elimination of seedlings (Horton and Krabel 1955; Hanes 1971; Vogl 1981; Keeley 1981).

Despite the advantages of resprouting following a fire, certain chaparral shrubs reproduce exclusively by seeds and lack the capability to resprout (ceanothus spp). Other species that have well developed capabilities for resprouting still produce millions of seeds annually, which would seem wasteful from the standpoint of a plant's allocation of energy. There appears to be certain environmental conditions within the chaparral community in which the capability to reproduce by seed becomes important. For example, seed reproduction becomes essential through periods of extreme environmental stress, such as extended droughts, landslides and in colonizing areas devoid of resprouting individuals (Vogl 1981). Keeley and Zedler (1978) hypothesized about evolutionary selection pressures under both short and long fire free periods that favor dominance by different species groups. During a short fire cycle, there are fewer dead shrubs prior to the fire, thus more potential resprouts. Subsequent fires are less intense, so there is less fire induced mortality, and fewer openings for seedlings to become established. The result is low selection pressure for obligate seeders, or plants that depend upon fire for regeneration by seed and do not recover by sprouting (ceanothus spp).

Sprouting species are at a disadvantage during long fire free intervals. The intensity of fire produced by the accumulated fuel load will reduce the number of individuals that can successfully resprout. Also, there are fewer resprouts after a burn because more shrubs were already dead prior to the burn. The result is larger openings for seedlings and high selection pressure for the obligate seeding strategy. The successful species will be the ones with the most seeds per unit area in the soil seed bank when the eventual fire does occur.

Under the scenario of pre-historic fire regimes, the obligate seeding strategy becomes extremely advantageous. Obligate seeding species will gradually become eliminated under high fire frequency or short fire return interval scenarios. Certain species of ceanothus, for example, may be eradicated because the post fire population requires at least seven years to produce viable seed and multiple years or even decades to
establish enough seed in the soil seed bank so that original population levels will be replaced after the eventual fire. With some fire free periods of 150 years or more, the limits of obligate seeders and other organisms resilience is unknown. The length of time that seeds remain viable is unknown for most chaparral species (Tyler and Odion 1996). Further study of seed bank longevity is needed to understand the risk to species of concern.

Under long term absence of fire, chaparral composition will shift towards taller canopy dominants which are in the vigorous crown sprouter group (such as oak and toyon), while ceanothus and other obligate seeders will drop out. Van Dyke et al. (2001) found that more than half of the herbs growing above ground were absent in samplings 25 years apart in maritime chaparral of Monterey County. The remaining herbs were restricted to the few remaining canopy gaps and the understory was bare except for litter and seedlings of trees, which need shade to germinate and then a canopy opening to fully establish new individuals. In general, herbs are expected to be low in number and restricted to canopy openings where dominant species die from aging, or on shallow soils or finer textured soil types that restrict shrub growth resulting in well spaced dominant plants with lots of openings. If fire parameters do not favor regeneration of obligate seeders, or if seedlings emerge under adverse environmental conditions after fire and die from drought or competition, stand dominance may shift to chamise. Animals with sedentary life cycles that are dependent on herbaceous or suffrutescent shrubs of a more open habitat condition may be at risk.

1.6.2 Historical Uses of Fire

Aboriginal use of fire is often invoked as the disturbance maintaining open grasslands and oak savannas. Many authors support the view that Indian burning was frequent and widespread (Cooper 1922, Jepson 1910). Although Indians lived and utilized the area within the Southern Subregion, the extent and frequency of Indian burning is unknown, however, it is very likely that Indians did use fire. Although evidence is not clear, it is quite likely that countless fires were set during the settlement period. Early California newspapers document grassland and chaparral fires, which burned for weeks and months at a time.

During contemporary land management, in addition to the wildfire history, a small number of vegetative management burns have occurred within the Southern Subregion (see Figure N-3). The Orange County Fire Authority, California State Parks, the Cleveland National Forest and the Marine Corps Base at Camp Pendleton have all implemented successful prescribed burns in grassland and shrubland vegetation. These efforts were intended to both reduce fire hazards and improve habitat quality.
1.7 The Fire Environment

1.7.1 Fire Weather

The fire season in Orange County usually starts in May and ends in November, although critical fire weather can occur year round (Orange County Historical Records). Significant fires have been recorded in December and January (California Department of Forestry and Fire Protection).

Several synoptic weather conditions produce high fire danger. One is a cold front passage followed by winds from the northeast quadrant. Another is produced by high pressure systems in the Great Basin. This Great Basin high produces the foehn-type wind along the west slope of the Coast Ranges, known as "Santa Ana Winds". Peak "Santa Ana" wind occurrence is in November with a secondary peak in March, however, over time Santa Ana winds have been recorded in every month of the calendar year with the exception of August.

A third high fire danger situation occurs when a ridge or closed high aloft persists over the western portion of the United States. At the surface, this pattern produces very high temperatures, low humidity, and air-mass instability (Schroeder and Buck 1970).

1.7.2 Fire History

Historic fire data indicates that large wildland fires have been a frequent visitor to the Southern Subregion lands. Most of the Southern Subregion lands have experienced a wildfire one or more times in the past 50 years. Since the 1940s, the California Department of Forestry and Fire Protection (CDF) and later the Orange County Fire Authority (OCFA) have documented all wildland fire events for the entire county. Figure P-8 depicts wildland fires by decade for the Southern Subregion. Most of these fire events were of human origin. The majority of ignitions have been associated with roadways, arson and person-related activities. Exceptions include the Santiago Canyon Fire of 1998, where multiple lighting strikes caused this fire.

1.8 Fuels

The Southern Subregion’s vegetation, topography and disturbance history has created a mosaic of fuel types. Frequent disturbance has created low volume fuel beds throughout portions of the Southern Subregion. Open grasslands in the eastern portion of the Southern Subregion are an example of this respective fire type. In areas where fire and grazing has been excluded, fuel loads have reached moderate to high levels.
Where fuels have burned within the last 10 years, fuels can be generally characterized as low volume with a high percentage of fine, herbaceous fuels.

A variety of fuel classes are represented in the Southern Subregion. Although most fuels occur in the 1-hr and 10-hr size class, 100-hr and 1000-hr fuels\(^1\) do exist in the Southern Subregion's dense brush, riparian and tree fuel models located within the interior units. Two grass fuel models occur (Fuel Model 1 and Fuel Model 2), as well as three shrub fuel models (Fuel Model 4, Fuel Model 5 and Fuel Model 6) and one tree (hardwood) fuel model (Fuel Model 9). Part III of this Fire Management Plan discusses these fuel models in detail.

### 1.9 Fire Effects

Fire is recognized as directly influencing the physical and chemical properties of soils. Many of the soils on the Southern Subregion are poor in some plant nutrients. Large portions of these nutrients are contained in actively growing plant stems. Mineral and nutrient cycling in fire type ecosystems is dominated by periodic ashing (Zinke 1977). Most nutrients are deposited on the soil surface where they are readily taken up by plants. Some portion of existing nitrogen will be volatilized. That remaining in ash is highly available in the form of ammonia nitrogen, or after nitrification, as nitrate nitrogen (DeBano et al. 1977).

Fire in shrubland communities may affect the soil infiltration rates. Large temperature gradients in the upper few centimeters of soil layer may cause vapor and gases containing hydrophobic substances to move downward in the soil profile where they condense on soil particles (DeBano 1966). Hydrophobicity may facilitate dry ravel and wet hillslope erosion processes.

Fire is effective in reducing on-site fuel loading including foliage, stems and woody portions of plants. Consumption rates may be high for litter and humus layers of soil. In

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\(^1\) The moisture content of vegetative fuels is critical, since it directly controls the combustibility of dead and living vegetative fuels. The capacity of fuels to vary in fuel moisture content depends upon the type and condition of the fuels. Cured or dead fuels are grouped as follows: 1. Fine fuels-grass, leaves, tree moss and loose leaf litter. These fuels can absorb or lose moisture very rapidly and their flammability can respond very quickly from no flammability to very high flammability in a very short period of time. These fuels are known as 1-hour fuels, ¼ of an inch in diameter or less; 2. Light fuels-small twigs and stems both living and dead. These fuels dry our very quickly and can become highly flammable. These fuels are known as 10—hour fuels and run from ¼ inch up to 1 inch in diameter; 3. Medium fuels-sticks, branch wood and medium deep duff. These fuels gain and lose moisture more slowly and cumulatively. They also add to wildfire intensity. Where they constitute the key fire carrying fuels of an area, their moisture content is more significant in the suppression of the fire than that of the more variable fuels. These fuels are known as 100-hour fuels. They take longer to gain and lose fuel moisture. This delay in gaining and losing fuel moisture is known as lag time. Fuel sizes run from 1 inch to 3 inches in diameter; 4. Coarse and heavily compacted fuels, logs, deep duff and peaty material. These fuels change moisture content very slowly and cumulatively over long periods of time. These fuels are known as 1,000-hour fuels and run from 3 inches in diameter and up to include large logs. When these fuels reach a state of low fuel moisture and high flammability very serious and destructive fires can occur.
addition, fire may create large amounts of dead organic matter by killing but not consuming vegetation (Wright and Heinselman 1973). This may be true within the Southern Subregion's riparian areas but is not the case on dry hillside slopes covered with dense chaparral.

Fire impacts on individual plant species and communities are often significant. Heat shock, presence of charite and change in photoperiods may all stimulate seed germination (Keeley and Keeley 1987). Post-fire response can include both vegetative reproduction and stimulation of flowering and fruiting (Malcolm 1977). Combustion of above ground biomass will alter seedbeds and temporarily eliminate competition for moisture, nutrients, heat, and light (Wright and Heinselman 1973).

On a community level, fire may also influence successional pathways through varying frequency and/or intensity.

Wildlife populations and their distribution are often regulated by fire. Fire can increase food resources for both grazers and browsers. Mass production of species, such as oaks, is also affected by fire. Insect populations, an important food source for birds and some mammals, may also be regulated by fire. Of particular consequence to wildlife is the mosaic of vegetation on the landscape. Vegetation type, structure and age class distributions are generally controlled by fire (Wright and Heinselman 1973).

Populations of insects, fungi and pathogens are also influenced by fire. Regulating stand age, sanitizing of plants and production of charcoal, which can stimulate ectomycorrhizae, can all impact the above-mentioned variables (Wright and Heinselman 1973).

1.9.1 Plant Community Response To Fire

Each plant community responds differently to wildland fire depending upon fire intensity and frequency. The key plant communities of concern in this document with regard to fire response are coastal sage scrub, grassland, oak woodlands and riparian. The literature cited in Section 1.6 and in the following sections describes both inland and coastal plant communities. The authors are describing vegetative conditions in Ventura, Los Angeles, Orange and San Diego counties, therefore, the term “coastal” and “inland” will vary depending upon location. “Coastal” can best be thought of as the summer season marine fog line that extends from the ocean to the coast range foothills. In the case of the Southern Subregion this would be that area that lies between the coastline and the Santa Ana Mountain range [coastal means the moist marine air still has a significant influence (at least occasional fog) on plant fuel moisture content. This
distinction is important. An estimated 94% of the existing population of the California gnatcatcher resides below 250m in elevation (Atwood, 1990) and sub-populations in the maritime zone appear to be more stable than those in interior coastal and transition zones).

1.9.1.1 Coastal Sage Scrub: Three floristic associations of coastal sage scrub are recognized in southern California: Venturan, Diegan, and Riversidian. Venturan coastal sage scrub occupies coastal sites in Santa Barbara, Ventura, and Los Angeles counties, while the Diegan association occupies coastal sites in Orange and San Diego counties. The Riversidian association occupies more inland sites in Los Angeles, Riverside, San Bernardino, and San Diego counties (Axelrod 1979; Kirkpatrick and Hutchenson 1977; Westman 1981c). Plant species common to these associations include California sagebrush (*Artemisia californica*), black sage (*Salvia mellifera*), white sage (*S. apiana*), purple sage (*S. Leucophylla*), California buckwheat (*Eriogonum fasciculatum*), brittlebush (*Encelia farinosa*) and California encelia (*E. californica*).

A complex composite of factors, particularly fire, was found to be significant in the maintenance of these habitat types (Wells 1962). With rapid crown sprouting and small wind dispersed seeds, sage scrub communities are often fire successional to chaparral at lower elevations (Cooper 1922; Wells 1962). Traditionally fires were likely to have burned hot, consuming most of the above ground biomass. Stand development periods seem to be short, with some longer period during which insufficient fuel is available to carry fire. Undisturbed stands may begin to open up their canopy in 20-25 years following periods of drought or competition for available nutrients and moisture. As canopy gaps begin to form, potential arises for invasion of annual grasses. It should be noted that major factors influencing sage scrub species distribution and composition include evapotranspirative stress, substrate type, soil nitrogen, and air pollution (Westman 1981c).

It appears that re-sprouting of dominant shrub species, particularly on coastal sites, enhances community succession in coastal sage scrub following fire (Mooney 1977). Keeley and Keeley (1982) studies within the Santa Monica Mountains estimated that 70 percent of pre-fire shrub populations re-sprouted the first year following fire, covering one-third of the ground surface by the end of the second season. The first year after a fire, herbaceous annuals dominate, with very little recruitment of perennial herbs from seed. Sprouts of dominant shrubs grow rapidly in height and dominate the site within several years. Shrub seedlings tend to be observed primarily two years after fire, possibly as a result of seed produced from first-year sprouts (Hanes 1971; Westman 1982).
Compared to coastal stands, inland stands of sage scrub recovered more slowly from fire and show greater change in post-fire composition because of reliance entirely upon seedling recruitment. Re-sprouting of dominant shrub species rarely occurs. Recovery of pre-fire species composition and cover on inland sites is therefore completely dependent upon an existing native seed bank or seeds brought onsite by wind or wildlife species (Myers and Ellstrand 1984). Generally, annual herbs dominate during the first year after a fire, but tend to decline in subsequent years as shrubs attain greater cover. Perennial herb understory species, which may grow from re-sprouts, show low recruitment for the soil seed bank.

Unlike herbaceous annuals, the overall diversity of perennial understory herbs remains constant the first few years following fire. New species continue to recruit into recovering sage scrub, reaching a peak at 5 to 10 years after a fire. After the 5 to 10-year peaks in species diversity, there is a general decline in perennial understory herb species. This may be attributed to dominate shrub species increasing in cover, thereby shading out the understory herbs (Keeley and Keeley, 1984).

On coastal sites, Malanson and O’Leary (1985) suggested that early post-fire recovery on sites dominated by suffrutescent shrubs such as deer weed (Lotus scoparius) may suppress seedling establishment, thereby favoring re-sprouting as the dominant recovery strategy. Because suffrutescent shrubs are more abundant during shorter fire intervals, seedling establishment may be favored during less frequent fires or during fires of greater intensity.

The resilience of a particular stand of sage scrub largely depends on the re-sprouting vigor of dominant shrub species. Sites dominated by vigorous re-sprouters tend to be more competitive than sites with both weak and strong re-sprouters, although intense fires can kill even strong re-sprouters. Sites with both strong and weak re-sprouters are more likely to experience permanent alteration (Westman and O’Leary 1986). Malanson and O’Leary (1982) noted the variable nature of re-sprouting within species and suggested that it may be attributed to differences in rooting depth, carbohydrate storage, location of adventitious buds, size of plant, soil moisture conditions at the time of fire, and fire intensity.

O’Leary and Westman (1988a) compared post-fire herbaceous growth between sites with different disturbance regimes. It was found that herb species richness was similar for all sites immediately after the fire. The sites on the coast that were not affected by air pollution or close proximity to grazing rapidly returned to pre-fire levels of herb diversity in conjunction with rapid recovery of shrub re-sprouts. However, a coastal site adjacent to grazing tended to become dominated by introduced annual grasses with
poor recovery of dominant shrubs. The introduction of competitive annuals on sites in proximity to grazed areas occurs due to the condition of the adjacent grazed area and the presence of annual grasses, seeds caught in the hair of cattle transported to the site from other areas and seed material in waste deposited on the ground.

Factors such as slope, aspect, and substrate type appear to have an effect on distribution patterns of herbs in sage scrub after fire. These variations in response may be driven by temperature and available moisture associated with insolation differences on opposing slopes. In addition, differences in soil temperature on opposing slopes may affect seed survival and also act as a factor in determining species distributions (O’Leary 1988).

Comparative analyses of fire response of two different sub-associations of Venturan sage scrub characterized by different soils and aspect revealed that species richness, cover, and equitability on north-facing slopes were higher than on south-facing slopes. This was attributed with relatively mesic conditions under which the particular sub-association develops (O’Leary 1990). Malanson’s (1984) simulations of shrub response to fire interval and intensity indicate that long-term fire trends are unlikely to have caused the sharp boundary between these two sub-associations of coastal sage scrub.

Because of the close association of sage scrub and chaparral throughout their geographic range (Gray and Schlesinger 1980), fire management strategies for these two plant communities have largely been the same. Studies have indicated, however, that the successional processes, and therefore, fire intervals for these two plant communities may be different. Unlike chaparral, coastal sage scrub shrubs are able to establish by seed and re-sprout on a continual basis in the absence of fire. Thus, a stand of coastal sage scrub may be typically mixed aged. This indicates that the optimum fire interval for coastal sage scrub may be different than chaparral, which may also explain the need for longer fire-free intervals in chaparral plant communities (Malanson and Westman 1984; Malanson 1985).

While it may be true that fire intensity in chaparral increases with longer fire intervals, a decline in leaf litter in coastal sage scrub over time can decrease fire intensity at longer fire intervals. Although the total annual litter fall in coastal sage scrub is similar with that of chaparral the low productivity, soft wood, green tissue, and open vegetation structure of coastal sage scrub may favor more rapid decomposition and prevent large fuel accumulations beneath the plants (Grey and Schlesinger 1981). Fire intensity in coastal sage scrub is likely more dependent on weather conditions than on stand age (Malanson and O’Leary 1985). More intense fire suppresses crown sprouting and consequently promotes herb flora.
Malanson (1985) utilized a fire behavior computer model to analyze demographic competition of five coastal sage scrub species under different fire intervals. Short fire intervals of 10-20 years may greatly reduce or eliminate some species, while longer fire intervals allow for the maintenance of species diversity. Unusually short fire intervals produce anomalous vegetation responses in coastal sage scrub (Zedler, 1983). After 2 years, many shrub species failed to re-sprout or re-seed.

The Point Loma Ecological Reserve, which is managed by the Department of Defense, United States Navy and the National Park Service (NPS), Cabrillo National Monument, has not had fire in this coastal sage scrub/chaparral ecosystem since 1871 or a fire free interval of 134 years. Many of the coastal sage scrub and chaparral species are dropping out, specifically ceanothus (*Ceanothus verrucosus*). Yet soil samples taken throughout the Reserve contain viable seed from plants no longer observable on the Reserve. When short duration intense heat is applied to these samples, species germinate that were once well represented within the Reserve.

In this case, a high intensity fire disturbance is needed to restore the Point Loma Reserve back to its pre-settlement vegetative condition (Zedler 1995). This example illustrates that in this case coastal sage scrub species diversity appears to be declining in this long fire free interval, which contrasts with some of the literature findings presented above. This is not to say that we are advocating short-term fire intervals in coastal sage scrub, however, the thinking that fire should be excluded for the longest fire free interval possible (well beyond 25 years) until a major wind driven wildfire burns through all age classes and habitats under the most extreme conditions is equally unacceptable.

Fire may be necessary for the annual and perennial gap species to be expressed and sustained and for both flora and dependent fauna species. We don’t know about all of these interdependencies, so a conservative stewardship approach that ensures some open condition that supports these species is appropriate. Animals with sedentary life cycles that are dependent on herbaceous or suffrutescent shrubs of a more open habitat could be at risk from a prolonged absence of fire as the shrub canopy fills in. The risk is also to dispersal limited, short lived species such as some butterfly species.

The main point here in bringing up potential risks of long fire intervals is to acknowledge that there are unknowns about plant communities and fire adaptation and that there is a risk of completely eliminating a natural disturbance regime from a conservation area.

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2 Pre-settlement is mentioned here because this is a management mandate of the NPS which is to manage the unique landscapes assigned to the Service in their pre-settlement condition for interpretation and education of future generations. There is absolutely no inference here that all coastal sage scrub in the Southern Subregion would be managed in a pre-settlement state.
based on the limited amount of inference we can make from available data. The risks of excluding fire completely may outweigh the risk of including the planned use of fire in a very objective and controlled manner. Again, we are attempting to manage wildlands and all the diversity present on them where fire has been a major disturbance force under a variety of conditions (not just fall Santa Ana wind driven wildfires) for thousands of years.

1.9.1.2 Grasslands: After fire, cover, density, and seedling establishment of purple needlegrass (*Nassella pulchra*) often increases as a result of increased soil temperature, light intensity, and nutrient release, and decreased standing litter (Ahmed 1983, Brown 1982, Dyer 1993, Dyer 1996, and Langstroth 1991). Regeneration occurs from tillers at the soil surface, fragmentation of bunches, and/or by seedling establishment. Needlegrass stands that experience severe fire have larger decreases in individuals’ basal area and foliage height in the 1st postfire year but are more likely to increase by fragmentation. These patterns are more pronounced with short-duration grazing, particularly in early spring (Dyer 1993 and Langstroth 1991).

Annual grasses have larger seeds than purple needlegrass and are better adapted to establishing in litter layers. For this reason, fire can increase purple needlegrass seedling establishment, particularly in old stands where litter accumulation is highest (Ahmed 1983, Dyer 1993 and Langstroth 1991). Adult individuals are also benefited by reduction of competition from annual grasses.

Even though fire during periods of rapid growth can be detrimental to purple needlegrass, it is generally more damaging to nonnative annuals (Ahmed 1983 and Bartolome 1981). Some studies, however, have found fire and/or grazing effects on cover, density, or seedling establishment of purple needlegrass were highly variable or insignificant, suggesting a large influence of climate on purple needlegrass' response to fire (Dyer 1996 and Hatch 1999).

Fire regimes: There is little direct physical evidence of the historical extent of purple needlegrass and less about historic fire frequencies in the communities where it occurs. Most agree, however, that purple needle grass' abundance was historically greater, and fire exclusion has been a factor in its decline (Bartolome 1981, Brown 1982 and Langstroth 1991). In the coastal scrub, chaparral, and oak woodland, fire frequency declined in the early 1900s with restrictions against burning; in grasslands fire frequency declined in the 1840s when heavy grazing and intermittent drought reduced fuels (Brown 1982 and Greenlee 1990).
Before Spanish settlement, California prairie was used by tule elk, pronghorn antelope, and mule deer, but grazing was intermittent enough to allow dominant grasses to regrow and support fire (Brown 1982). In many areas where purple needlegrass and non-native annuals now coexist, purple needlegrass and native annuals were historically mixed. Here, the interaction of fire and grazing likely reduced competition from annual grasses, reduced woody species encroachment, and improved purple needlegrass regeneration (Dyer 1997 and Langstroth 1991).

One study of vegetation dynamics in coastal sage scrub, chaparral, and coast live oak woodland near Santa Barbara found that without fire or livestock grazing, coastal sage scrub was replaced by oak woodland at a rate of 0.3% annually. Grassland to coastal sage scrub transition occurred at a rate of 0.69% per year, and oak woodland reverted to grassland at a rate of 0.08% per year. On burned areas without livestock grazing or on unburned sites with livestock grazing, rates of transition of grassland to coastal scrub and coastal scrub to oak woodland were lower.

On areas burned without grazing or grazed without burning the rate of oak woodland reversion to grassland was higher than on areas with neither burning nor grazing (Callaway 1993).

In chaparral and coastal scrub, early postfire vegetation is dominated by native and nonnative annuals. Herbaceous vegetation is greatest in areas where fire eliminates nonsprouting shrubs (Keeley 1990 and Mensing 1998).

Purple needlegrass and other perennial grasses are more abundant after fire in coastal sage scrub than in chaparral. Fire repeated in less than approximately 3-year intervals often causes the herbaceous sere to persist (Keeley 1990). Conversion of purple needlegrass grassland to coyote bush/ripgut brome communities has been observed with 24 years of fire exclusion (Langstroth 1991).

Purple needlegrass is present in oak and pine woodlands and in the early seral stages of mixed evergreen forests, redwood and coast Douglas fir forests. Generally, purple needlegrass and other herbaceous species are present in later successional oak woodlands only in intercanopy areas. Closed stands have up to 5 inches (12.7 cm) of oak litter that essentially eliminates grass growth (Plumb 1983).

1.9.1.3 Oak woodland: Coast live oak (Quercus agrifolia) is exceptionally fire resistant, more so than other California oak species. Evergreens are often better able to conserve nutrients than deciduous species and are favored in fire prone environments (McDonald 1981). Low intensity surface fires have little effect on mature coast live oak.
Saplings and seedlings generally recover quickly from low to moderate intensity fire (Dagit 2002, Plumb and McDonald 1981). Because of mortality among small diameter oak trees, frequent fire limits coast live oak invasion of grasslands (Mensing 1998). Mature oaks are more likely to be damaged by fall fires than early season fires. Severely burned oaks are vigorous sprouters (Plumb and Gomez 1983).

Acorns on the soil surface are killed by low intensity fire, where animal-buried acorns usually survive moderate intensity fire, sometimes resulting in high rates of post fire establishment (Davis, Keller, Parikh, and Florsheim 1989; Lawson, Zedler, and Seiger 1977).

Fire frequency largely defines the extent of coastal sage scrub, chaparral, and oak woodland; in these habitats decreasing fire frequency tends to favor the development of coast live oak. One study of vegetation dynamics in coastal sage scrub, chaparral and oak woodland near Santa Barbara, California found that without fire or livestock grazing, coastal sage scrub was replaced by coast live oak woodland at a rate of 0.3% annually. Again without fire, grassland to coastal sage scrub transition occurred at a rate of 0.69% per year, and oak woodland reverted back to grassland at a rate of 0.08% per year. On burned areas without livestock grazing, rates of transition of grasslands to coastal sage scrub and coastal sage scrub to oak woodland were much lower and the rate of oak woodland reversion to grassland was higher (Callaway and Davis 1993). Sites without shrub increase are generally south facing and/or on shallow soils (Griffin 1977). Generally, grass is present in open stands while closed canopy stands have up to 5 inches of oak litter, which prohibits the presence of grass (Plumb and Gomez 1983).

Fire exclusion in coastal sage scrub and mesic chaparral communities allows coast live oak to increase in density and reduce understory diversity and abundance (Mensing 1998; Van Dyke, Holl and Griffin 2001). Van Dyke and Holl recommend prescribed burning in coastal sage scrub to maintain scrub species and associated herbaceous species and to slow coast live oak expansion.

Flammability of coast live oak, coastal sage scrub and chaparral communities with a coast live oak component is of particular concern because of their high fuel loadings and proximity to developments in interface areas. Some fire-excluded chaparral habitats have fuel accumulations of 30 to 40 tons per acre (Hecht-Poinar, Costello and Parmeter 1987).

Fuel modification zones in the immediate areas of development provide the best measure of protection for structures encroaching into wildland areas. Recommendations for property protection include: planting trees away from the
structures so at maturity the tree crowns are no closer than ten (10) feet to the structure, trimming up low branches up to six (6) feet from the ground, eliminating all shrubbery from beneath the canopy of the planted trees, selecting less flammable native plants for landscaping (see the Orange County Approved Plant List in Appendix A) and using non-flammable construction materials (East Bay Municipal Utility District 1992; Franklin 1997).

Domestic goat grazing (at a rate of 240 goats per acre for 1 day) in conjunction with prescribed fire, has been used successfully to reduce fuel loadings and fuel continuity in dense coast live oak, coastal sage scrub scrub/chaparral near housing developments (Tsiouvaras, Havlik, and Bartolome 1989).

1.9.1.4 Riparian: Coast live oak associates and willow (*Salix spp.*) associates in riparian areas include white alder (*Alnus rhombifolia*), California sycamore (*Platanus racemosa*) and Fremont cottonwood (*Populus fremontii*) all of which sprout vigorously after fire. Severe high intensity fires were apparently historically rare in riparian habitats. Currently most fire in riparian zones is accidental and of high severity, causing relatively high rates of top-kill and basal sprouting of all these species (Barro 1989; Davis, Keller, Parikh, and Florsheim 1989). Riparian zones comprise the least number of acres by vegetation type representing between 2 and 5% of the total acres in the Southern Subregion, yet they contain the greatest mix of biodiversity richness. Fires in riparian zones should be suppressed at the smallest size possible. Please refer back to Section 1.5.3.4.4 Riparian, for a more complete description of the riparian zones and fires affect on them.

Fire managers in southern California utilize fire control lines adjacent to coast live oak riparian woodlands and willow dominated riparian areas when planning broadcast burns of adjacent grassland, coastal sage scrub and chaparral vegetation as well as for containing wildfires. The control lines are burned out to create a fuelbreak between the riparian zone and the adjacent fuels targeted for treatment with prescribed fire (Dougherty and Riggan 1982).

1.9.2 Wildlife Response To Fire

Because so little has been published on the response of wildlife to fire in coastal sage scrub, this summary relies heavily on the fire ecology literature as it relates to other habitats, especially chaparral. Of the papers cited here, only Price and Waser (1984), Moriarty et al. (1985), and Stanton (1986) pertain specifically to coastal sage scrub. Writing in 1969, Udvardy stated that the literature on special adaptations toward fire resistance in animals is scarce, and the effect of recurring fires on their distribution has
not yet been assessed. This account will address several general topics before discussing selected taxonomic groups in more detail.

The role of disturbance in creating early successional habitats, and how it relates to the overall health of wildlands, is receiving increasing attention in the conservation biology community (e.g., Litvaitis 1993). Under natural conditions, fire is the most common disturbance in many plant communities, including coastal sage scrub. The positive role of fire in maintaining a mosaic of habitats has often been emphasized (Fox and McKay 1981, Quinn 1982, Willan and Bigalke 1982, Pyne 1984). The latter author noted that the variable intensities of fire ensure that a variety of biotic ensembles, a mosaic, persists. The following summary was also provided:

“Free burning fire, it is argued, is a primary mechanism for ensuring complexity, variety, and ultimately stability in natural systems.”

The indirect influence of fire (primarily the temporary loss of habitat) has long been recognized as being far more important than direct impacts (Leopold 1933). Still, considerable attention has been given to the fate of wildlife during fires. The negative observations of Chew et al. (1959) are the exception. They found 43 dead mammals and two dead birds in 1.7 acres following a Malibu, California chaparral fire and suggested that the fire’s toll on wildlife was enormous. Howard et al. (1959), Stoddard (1903), Komarek (1969) and Biswell (1989) especially downplay the loss of life (birds and animals) due to wildfire, based largely on their experiences with controlled burns, which typically burn with less intensity and slower rates of spread than wildfires. Leopold (1933), Lawrence (1966), Catling et al. (1982), and Pyne (1984) took more moderate positions, suggesting that few birds and mammals die in wildfires, but acknowledging that under certain conditions, many animals may die.


Komarek (1969) provides considerable information concerning animal responses to fire, and included a lengthy appendix detailing specific species’ reactions. Most of his observations were in the southeastern Untied States, in the general vicinity of the Tall Timbers Research Station in Tallahassee, Florida. He observed ants relocating their nests (including eggs and larvae) from burned areas to unburned vegetation within an hour after burning.
Komarek (1969) found frogs seeking moist areas to avoid fire and heard the spring chorus of certain species resume soon after a fire passed by a breeding pond. Similarly, he found little evidence of lizards or snakes killed by fire. He watched hispid cotton rats (*Sigmodon hispidus*) herding and carrying young to safety ahead of fires, and never found dead young in burned nests.

Recent surveys have shown large numbers of snakes being killed during coastal prescribed burns. Thirty-five dead snakes were collected in a 25-acre burn (Fisher, unpublished data). The majority of animals collected were western rattlesnakes (*Crotalus viridis*). Fisher has proposed that fire directly affects the heat sensors of these animals and increases the chance of individuals being killed. Still a large number of live snakes were observed in the same units, following fires. This can be attributed to those individuals that survived in rocky outcrops or took refuge in ground squirrel, fox and rabbit burrows.

In a controlled experiment, Howard et al. (1959) measured the lethal temperature for several chaparral rodents at 138-145°F. Burrows a few inches deep were sufficient to insulate animals from these temperatures as fire burned on the surface. Lawrence (1900) examined this issue further, finding that three (3) inches of depth was probably enough to survive heat and increased vapor pressure in burrows. He suggested that post-burn predation is probably a more restrictive factor on small birds and mammals than the fire itself. Still, Wirtz (1995) found species requiring brush for cover and/or food, like the California gnatcatcher (*Polioptila californica californica*), are most severely impacted by fire, and require the longest time to recover to pre-fire densities.

Some animals are drawn to active fires. Biswell (1989) reported that birds have been observed to fly in back of a fire and begin feeding almost immediately. Raptorial birds and predatory mammals exploit birds and small mammals fleeing fires, while flycatchers, swallows and others aerial feeders prey on displaced insects (Stoddard 1963, Komarek, 1969). Other species, especially ground feeders, such as mourning doves (*Zenaida macroura*), northern flickers (*Colaptes auratus*), American robins (*Turdus migratorius*), bluebirds (*Sialia sp.*), sparrows, and finches may forage on burned areas immediately following fire (Stoddard 1963). Komarek (1969) noted many instances of birds and mammals consuming ash following fire, presumably as a dietary supplement.

Quinn (1982) lamented the fact that the study of insects in chaparral and other Mediterranean-type ecosystems has been largely ignored, especially in view of the strong influences insects have over plant communities.
Force (1982) conducted a four-year post-fire study in chaparral of the San Gabriel Mountains and found that pollen-nectar feeders and predatory insects can be very abundant beginning in the first spring after a burn. Phytophagous insects (other than flower feeders) and parasitic insects more slowly establish in the burn. Fourth year insect richness and diversity showed a dramatic increase after an overall three-year decreasing trend.

Hogue (1993; page 46) discussed “fire beetles” (genus *Melanophila*, family Buprestidae) and smoke flies (genus *Microsania*, family Platypezidae), noting that both are attracted to heat and smoke and may arrive on a burning plot before the flames recede. Lawrence (1966) found insects to be particularly susceptible to predation by California quail (*Callipepla californica*), California towhees (*Pipilo crissalis*), and western meadowlarks (*Sturnella neglecta*) following fire.

Birds, as a group, have received the most attention in wildlife fire studies. The benefits of fire in game management have been especially well covered. In attempting to summarize the effects of wildland fire on wildlife, Pyne (1984) stated that, in general, there tends to be a slight increase in avifauna and relatively constant number of mammal species following fire. The size of individuals tends to increase in both birds and mammals due to the abundance of both seeds and insects on burned over areas for the first several years following a fire.

Shrubland bird populations are generally known to decline as an immediate response to wildfires (Smith 2000). This can result from emigration to unburned patches and direct fatalities as result of overexposure to the fire, or to predation following the fire event. Population levels can also directly affected if fire disrupts nesting areas. If burns are patchy the initial losses related to the fire itself are often counter-balanced by population increases during the vegetation recovery in subsequent years. Research in chaparral after the first year of a stand-replacing fire suggests that avian species richness in the burn was 70% to 90% of that in the unburned adjacent sites (Moriarty et al.1985).

Patch size has been shown to be the single most important predictor of native plant species richness (Alberts et al. 1993). It follows that patch size dynamics of burns and the availability of unburned islands greatly influences the use of unburned sites by shrub-dependent bird species, such as in burned coastal sage scrub (Beyers and Wirtz 1997). Researchers have found a negative correlation between a number of species and the proximity of urban edges or level of fragmentation of a habitat (Alberts et al. 1993, Andren et al. 1985, Andren and Angelstam 1988, Santos and Telleria 1992). Bolger (1997) found that the abundance of edge- or fragmentation-reduced species appears to be depressed within 650 to 1650 ft from an edge, and the abundance of
edge/fragmentation enhanced species is elevated up to 3,280 ft from an edge, depending on the species.

Sage sparrows (*Amphispiza belli*), wrentits (*Chamaea fasciata*), black-throated sparrows (*Amphispiza bilineata*), bushtits (*Psaltriparus minimus*), mountain chickadee (*Parus gambeli*), acorn woodpeckers (*Melanerpes formicivorus*), white-headed woodpeckers (*Picoides albolarvatus*), other sapsuckers, western screech owls (*Otus kenicottii*), northern sawwhet owls (*Aegolius acadicus*) are all canopy dependent species which would decline in a post fire environment where the majority of the shrub cover is consumed. Canopy dependent species may have severe short-term and long-term effects because of possible type conversion in some habitats (for example, from pine forest to oak woodlands in the Cuyumaca area of the 2003 San Diego County Cedar Fire).

Wrentits and California thrashers (*Toxostoma redivivum*), are very sedentary and are easily killed by wildfire. California gnatcatchers prefer the cover and structure provided by certain mature, unburned coastal sage scrub vegetative types. California thrashers and gnatcatchers will recolonize burned coastal sage scrub sites four to five years after the fire and do not reach maximum densities until twenty years post fire (Cody 1998).

In contrast, the lazuli bunting (*Passerina amoena*) feeds on the first annuals after fire and is considered a fire follower species. It was never detected at any locations within the October 2003 Cedar Fire perimeter during the five-year San Diego Natural History Museum surveys for the San Diego County Bird Atlas Project (Phil Unitt, Cuyumaca Conference 2004), but was observed in the first post fire period. Also, Lawrence’s goldfinch (*Carduelis lawrencei*), mountain quail (*Oreortyx pictus*) and wild turkey (*Meleagris gallopauo*) increased. A 2,000% increase in black-chinned sparrows (*Spizella atrogularis*) was observed and swallows, swifts, sparrows and flycatchers were more abundant in burned over chaparral in the first post-fire period. Finally, this rebound effect was especially dramatic on Costa’s hummingbird (*Calypte costae*). This increase resulted from heightened levels of fire followers such as poodle-dog bush (*Turricula parryi*), beardtongue (*Penstemon spectabilis*), and woolly blue-curls (*Trichostema lanatum*) that promote feeding and nesting. Pre and post surveys in areas consumed by the 2003 Cedar Fire in San Diego County tend to bear out Pyne’s above “general” summary of fire effects on avifauna. Avifauna species richness has increased as Pyne states, however, the mix of species is different in that avifauna that benefit from the temporary openness of the burned over conditions are very abundant while canopy dependent species have declined.
It is very unusual for raptors to suffer mortality due to the direct impact of fire (USFS 2003). Adult individuals can escape fire, however, fire could directly reduce raptor populations if the fire destroys nesting trees. Low intensity fires probably have little negative impact on raptors. Most raptors are unaffected or positively benefited when occupying burned habitat. Burned areas provide little cover for prey species and raptors readily take advantage of this vulnerability. Additionally, because prey species populations often increase after fire, raptors continue to benefit. Coopers hawk (*Accipiter cooperi*) populations have been documented as benefiting from fire (Dodd 1988 as referenced in Smith 2000). Nonetheless, fires that destroy potential and existing nesting trees could impede reproduction of raptors when alternative nesting sites are scarce (USFS 2003).

Studying chaparral in the Sierra Nevada, Lawrence (1966) found that many species were severely exposed to predation in the bare ash following fire, and most small mammals and brush dwelling birds decreased rapidly; predatory birds and mammals were found to increase. Further, with time, brush-dwelling species declined as forbs and grasses increased, while grass-dwelling species increased. No species were eliminated altogether. This finding points out that nothing is static. Many species benefit with disturbance and then slowly decline in numbers as the species they replaced begin to increase as the site returns to pre-fire disturbance conditions.

Komarek (1969) pointed out that birds and mammals are often attracted to a “greening” burn site, where they feed on tender shoots unavailable elsewhere.

Without specifying habitats, Biswell (1989) claimed that one could expect an increase in bird numbers the first year after fire, especially seed-eating birds. Lawrence (1966) and Wirtz (1982) present the results of two relatively long-term studies of bird response to fire in chaparral. Lawrence found mourning doves and western meadowlarks to be among the earliest users of burned areas at his Sierra Nevada study site, and the degree of habitat recovery in the first year following fire was sufficient to allow accelerated reproductive rates in these species. He documented an overall increase in nesting bird density following fire, especially among seed-eating birds. Increased numbers of predators following the fire included sharp-shinned hawk (*Accipiter striatus*), Cooper’s hawk, red-tailed hawk (*Buteo jamaicensis*), American kestrel (*Falco sparverius*), great horned owl (*Bubo virginianus*), and common raven (*Corvus corax*).

Wirtz (1982) found both species richness and species diversity to increase in the 42 months following fire at his study site in the foothills of the San Gabriel Mountains. No increase was noted in the number of omnivorous birds or birds that take insects from
the air (flycatchers), but increases were noted in the number of insect and seed-eating birds (towhees, quail and meadow larks). These differences were most pronounced in the first year following the fire. Species that glean insects from vegetation and insect and fruit-eating species exhibited a decrease in the use of burned areas.

Moriarty et al. (1985) and Stanton (1986) compared bird communities on a burned coastal sage scrub site and control site in Pomona, California. The initial study showed greater species richness on the control site, but similar numbers of individuals on both sites, due in large part to the presence of ground-feeding finches. Substantial similarity between the two sites was evident within one year of the fire. Wrentits, California thrashers, and California towhees were more common on the control site, while mourning doves, scrub jays (*Aphelocoma coerulescens*), house finches (*Carpodacus mexicanus*), lesser goldfinches (*Carduelis psaltria*), and American goldfinches (*C. tristis*) were more common on the burned site.

The follow-up study by Stanton (1986) was completed less than three years following the fire. Reduced species richness was again found on the burned site, with similar numbers of individuals on the two sites. Most species preferred the control, with the following species among the exceptions: American kestrel, Say's phoebe (*Sayornis saya*), western kingbird (*Tyrannus verticalis*), yellow-rumped warbler (*Dendroica coronata*), lazuli bunting, house finch, and lesser goldfinch. Greater heterogeneity of habitat was offered as the explanation for greater bird use of the control site, and it was suggested that coastal sage scrub might not fit the general pattern of increased bird use following fire in chaparral.

Two sensitive birds that utilize coastal sage scrub plant communities, the coastal population of cactus wren (*Campylorhynchus burnneicapillus*) and coastal California gnatcatcher show some negative correlation to fire. Rea and Weaver (1990) point out that fire is apparently the primary limiting factor in the distribution of cactus in southern California. In coastal California, cactus wrens are restricted to coastal sage scrub with large cactus clumps. On Camp Pendleton, in San Diego County, Tutton et al. (1991) found that 80 percent of known coastal California gnatcatcher locations were in areas that had not burned in at least 16 years. Artemisia dominated coastal sage scrub attracts the highest numbers of gnatcatchers as documented in a study on the Fallbrook Naval Weapons Station located immediately to the east of Camp Pendleton (Kellogg 2002) with many coastal sage scrub variants receiving no use by gnatcatchers. Biologists from Dudek and Associates have noted in personal conversations with this author that an area of coastal sage scrub vegetation devoid of gnatcatchers in Carlsbad, CA unintentionally burned several years ago. As the site recovered Artemisia was well represented on the burned area and gnatcatchers began to occupy this site.
where no records of occupation existed prior to the fire. In coastal sage scrub areas unburned in many years with a poor representation of Artemisia, fire may actually increase habitat suitability for gnatcatchers provided Artemisia becomes the dominate plant in the coastal sage scrub post fire recovery. This relationship of Artemisia’s response to fire in areas where Artemisia is no longer dominate can be tested for in the experimental plot design discussed in Chapter 5 without executing an untested landscape scale approach.

Beyers and Wirtz suggested the threatened California gnatcatcher could potentially be adversely affected by prescribed burning in coastal sage scrub. The authors recommend maintaining both mature and burned scrub areas to reduce fuel loads while mitigating impacts to shrub-dependent species such as the California gnatcatcher. If connectivity exists between burned and unburned habitats, prescribed fire can be used without adverse impact. (USFS Fire Effects database)

The coastal California gnatcatcher (CAGN) is an obligate coastal sage scrub species. USFWS estimated that only about 2,500 pairs exist in the United States as of the Listing date (March 30, 1993). The range of this non-migratory songbird extends from the coastal areas of southern California including Los Angeles, western Orange, and San Diego Counties, south to El Rosario in Baja California, Mexico. Within this range, CAGN is found in the coastal areas or in areas where the temperature regime is moderated by the marine influence. Coastal sage scrub on slopes less than 25% appears to be the preferred habitat for the nesting CAGN, although dispersal often occurs through other nearby habitats such as riparian and chaparral. Nesting territories range in size from two to 30 acres, which both the male and female defend zealously. (50 CFR Part 17; March 30, 1993).

In surveys in 2000 on the Naval Weapons Station at Fallbrook, Varanus Biological Services (2002) showed that gnatcatchers prefer coastal sage scrub habitat dominated by California sagebrush (Artemisia californica) (subtype I), California sagebrush/California buckwheat (Eriogonum fasciculatum) (subtype II), and white sage (subtype III) irrespective of their representation on the installation. In 2000, 67% of California gnatcatcher locations were within habitat that had not burned in over 25 years. However, gnatcatchers did begin moving into burned habitat 6-10 years after a fire and used all age classes beyond the previously burned area. According to the available fire history, 55 (51%) of the observations were within habitat that was unburned. This is similar to results from Camp Pendleton (MCBCP 1998) which found 64% of 1994 gnatcatcher locations within unburned habitat (1973–1997 fire history).
Shrub structure and cover may be more important than how old the habitat is, though the two are obviously related. Atwood (1990) and Bontrager (1991) both suggest that vegetation within gnatcatcher territories is approximately one meter in height, but also that California gnatcatchers are more abundant in areas where herbaceous patches are intermixed with shrubs (Atwood and Bontrager 2001). The species requires approximately 50% shrub cover and over 3.3 feet (1 m) in shrub height for nesting territories. Under optimal conditions, this stand structure is commonly achieved in 4 or 5 years after fire, or as long 15 years. Using Bontrager’s estimate of 60-70% shrub cover within gnatcatcher territories, Cario and Zedler (1995) estimated that gnatcatchers should prefer habitat on Camp Pendleton that is approximately 8-15 years old (Figure 15 in Cario and Zedler). Based on data from long-term ecological trend monitoring (LTETM) plots, Cario and Zedler state that shrub cover can reach 70% in some coastal sage scrub plots four years after a fire and continual increases can eventually lead to some areas with 90% shrub cover. They explain the use of these high density areas by gnatcatchers by pointing out that older coastal sage habitats often have a higher density of dead shrubs which may keep cover within acceptable levels for gnatcatchers (Cario and Zedler 1995). Other factors such as the intensity of the fire, grazing levels, and available seed bank will also influence how quickly vegetation returns after a fire. Furthermore, Cario and Zedler made these extrapolations without actually studying gnatcatcher occupation and without distinguishing among subtypes of coastal sage scrub that gnatcatchers prefer; so, more work is needed to validate their determinations. Finally, in later work done on Camp Pendleton (Tierra Data 2002) which assessed coastal sage scrub cover in locations believed to be of similar age based on fire history, actual shrub cover was found to vary widely. Recovery after five years was consistently greater than 40% total shrub cover; however, some sites exceeded 60% cover in two to three years, while others remained at 50% cover after 20 years.

Excessive fire frequency in coastal sage scrub can promote dominance of shrublands by laurel sumac, which lowers the value of the habitat for California gnatcatchers. Also, there is a smaller risk of losing representation from fire-following annuals with extremely long fire intervals.

“Efforts to conserve bird and small mammal biodiversity in coastal sage scrub should not focus exclusively on rare species or on locations with the highest species richness, but instead should focus on a diverse suite of species that are representative of the range of variation in communities found in coastal sage scrub habitats” (Chase et al. 2000) Chase, Mary K., William B. Kristan III, Anthony J. Lynam, May V. Price, and John T. Rotenberry. 2002. Single species as indicators of species richness and composition in coastal sage scrub birds and small mammals. Conservation Biology 14, 2: 474-487.
Estimate of net gain to gnatcatchers by establishment of firebreak perimeter estimated at Fallbrook Weapons Station:

In the most recent ten years, about 1,289 acres of coastal sage scrub have burned, including all subtypes of this community, from fires coming onto the Detachment from elsewhere. (In the previous decade, a similar acreage of coastal sage scrub burned from fires coming from elsewhere.) If 50% of this could have been saved, this would have been about 644 acres that did not burn over that 10-year period, or 64 acres per year saved if burns were spread evenly over 10 years. Considering that after about seven years the burned habitat may be occupied again by gnatcatchers anyway, the actual savings per year would be about 70% of 64 acres, or 45 acres. This is a savings of 450 acres of habitat potentially suitable for gnatcatcher occupation over 10 years. Recognizing that this is an imprecise calculation with many uncertainties including numbers of burns, severity, and size, it is still much greater than an estimated permanent loss of 12 acres (new firebreaks to be maintained as bare soil), and temporary loss of at most 186 acres (newly burned fuelbreak affecting coastal sage scrub, excluding scrub islands intentionally left unburned) for a perimeter buffer area of all subtypes of coastal sage scrub. In the previous decade (1983-1993), the additional coastal sage scrub available for CAGNs would have been about 420 acres (excluding fires related to control burns), using the same logic. [based on assumption that a firebreak in combination with a fuelbreak would be 80% more effective than a firebreak alone at stopping the flank of a wildfire, and 20% more effective at stopping the head of a wildfire (Green 1977; Montague, pers. comm.).]

Through the development of pre-suppression and suppression tactics aimed at reducing fire frequency and size of wildfires, the Fallbrook Fire Plan is expected to result in an older age structure for coastal sage scrub across the Station. Age classes will be determined every five years in conjunction with CAGN surveys. The Fire Department will use pre-suppression tools, including prescribed fire, to accomplish these results in order to buffer potential catastrophic loss of gnatcatchers and their associated habitat.

In the case of the 1993 Laguna Fire approximately 13,000 acres of natural vegetation in the San Joaquin Hills burned including 6,800 acres of coastal sage scrub. Only 470 acres of coastal sage scrub within the burned area was left unburned or only lightly burned. Prior to the fire an estimated 127 pairs of California gnatcatchers and 282 pairs of cactus wrens occupied this area.

The fire resulted in the loss or displacement of many of the resident California gnatcatchers and cactus wrens. However, the coastal sage scrub plant community reestablished itself quickly and within two years the numbers of gnatcatchers and
Cactus wrens began to increase over post fire populations. In 2003, almost 10 years after the Laguna Fire, the numbers of California gnatcatchers were almost back to pre-fire population levels and cactus wrens have also increased, but at a much slower rate (Harmsworth Associates 2003).

As pointed out above, the range for habitat recovery can be from 4 to 15 years, which is a result of both on site factors (slope, aspect, soils, etc.) and weather cycles, however, in this case population recovery began in the second year after the Laguna Fire due to rapid habitat recovery. This slower response for cactus wrens is attributed to a lack of suitably sized cactus plants which were hit hard by the fire and recover at a much slower rate than do the coastal sage scrub species. Cactus wrens require large clumps of cactus, which keeps predators away from their nests, which are in the open but in the interior reaches of the cactus plant.

Similar findings are being observed in the 1996 and 1997 burned areas in the Upper Chiquita Canyon Conservation Easement. A new wildfire occurred in May 2002 in the southern portion of the Conservation Easement. In spite of the recent droughts, the 1996 burned area is coming back to a mixed sage plant community and the 1997 burn is coming back to a coastal sage scrub plant community. Gnatcatchers nested in the 1997 Coto Wildfire burned area for the first time in the spring of 2003. In the May 2002 Antonio Wildfire, population numbers declined as follows: before the fire gnatcatchers were estimated at 104 locations and after the fire were detected at 42 locations. Cactus wrens dropped from being detected at 65 locations to being detected at 30 locations. Population numbers are adequate to permit the re-colonization of the May 2002 burned area as soon as the habitat recovers (Harmsworth Associates 2003). Coastal sage scrub tolerates a wide range of fire intervals, from 12–40 years. While these shrublands are fire-adapted, below a certain threshold of fire frequency, resilience is inversely related to the fire return interval – this threshold is approximately 15 years in chaparral and 5-10 years in coastal sage scrub (Keeley 2000). Conversion to annual grasses has been reported at fire return intervals of five to 10 years, particularly at drier inland locations (Timbrook et al. 1982, Callaway and Davis 1993, Riggan et al. 1994, Minnich and Dezzani 1998, O'Leary 1995b). This may be due to interaction with other disturbance types such as grazing or drought or the ready establishment of exotic annual herbs which can often support high fire frequencies (Minnich and Dezzani 1998). The Southern Subregion wildfire history records (see Fire History Maps) indicate this area of the May 2002 Antonio Wildfire last burned in the 1958 Steward Wildfire for a fire free interval of 44 years. This long fire free interval supports the Harmsworth prediction for successful coastal sage scrub habitat recovery and subsequent re-colonization of the 2002 Antonio burn by both gnatcatchers and cactus wrens occupying adjacent habitat.
The results of a number of studies, primarily on rodents, are hereby summarized. Crowner and Barrett (1979) identified three major factors influencing reduced rodent numbers following fire: 1) reduced cover, 2) increased predation, and 3) reduced food availability. Kaufman et al. (1990) noted two additional behavioral factors: forced emigration and direct reduction in reproductive output.

In brush habitat in the east San Francisco Bay region, Cook (1959) found that rodents were apparently limited in the first year following a fall fire by a lack of cover, as seed was abundant by early spring. After initial “annihilation,” brush-dwelling mice showed a population increase, exceeding that of a control site throughout the second year following the fire.

No small mammals were trapped immediately following a fire on a Sierra Nevada chaparral site studied by Lawrence (1966), and no marked animals from the burned area were captured in adjacent habitat. Three months following the fire, marked animals were again trapped on the burned site, confirming the survival of a resident population. The loss of adults and the degree of habitat recovery in the first year following the fire apparently stimulated the reproductive rate of brush mice, producing more young than the control site. The average ratio of body weight to body length of brush mice (*Peromyscus truei*) was reduced in the first year following the fire, but was nearly equal to control animals in the second and third years following the fire.

Blankenship (1982) found no significant difference in rodent weights between burned and unburned sites in montane chaparral in San Diego County.

Rodent species richness, biomass per hectare, and species diversity reached levels equal to, or exceeding, those in 16 to 20 year old chaparral within 15 to 24 months post-fire on Wirtz’s (1982) study site in the San Gabriel Mountains. Heteromyids (Kangaroo rats and pocket mice) and California meadow mice (*Microtus californicus*) contributed significantly to early post-fire series; woodrats (*Neotoma spp.*) and white-footed mice (*Peromyscus sp.*) contributed significantly in older stands. Because of above ground nesting, woodrats are particularly susceptible to fire, and may not recolonize burned areas for 1-2 years following fires. It is assumed that some refugia always remain, due to the normally patchy nature of burns.

A study in coastal southern California coastal sage scrub by Price and Waser (1984), suggests brush-dwelling species declined following disturbance by fire while Erickson (1993) suggested fire may have been beneficial in opening up habitat for the pacific pocket mouse (*Perognathus longimembris pacificus*) in coastal southern California coastal sage scrub.
In conclusion, our wildland ecosystems are not static, nor are the wildlife populations that inhabit these systems. Withholding fire benefits some plant and animal species and denies others and conversely, fire benefits many plant and animal species while adversely impacting others. Many species that have adapted to southern California over thousands of years need and, in fact, require fire to perpetuate themselves although in some cases that relationship is not so obvious.