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Chapter 6: Fire in Western Shrubland, Woodland, and Grassland Ecosystems

Western shrubland, woodland, and grassland ecosystems lie west of the eastern humid temperate zone, which begins a short distance east of the100th meridian. Shrublands include sagebrush, desertshrub, south western shrubsteppe, Texas savanna, and chaparral-mountain shrub ecosystem types. Woodlands include south western ponderosa pine, pinyon-juniper, and oak types that at times can be considered either forests or woodlands. The woodland/forest dichotomy can depend on phase of stand development and on the realization of natural site conditions that can form savannas with tree overstories. Grasslands include plains, mountain, desert, and annual grassland ecosystems (table 6-1).

Understory Fire Regimes

Major Vegetation Types

Southwestern United States ponderosa pine consists of two varieties: (1) interior ponderosa pine found over most of Arizona and New Mexico, and (2) Arizona pine found in the mountains of extreme south western New Mexico and south eastern Arizona, and extending into northern Mexico (Little 1979) (fig.6-1, 6-2). Based on stand physiognomy (as in Paysen and others 1982), many stands of this vegetation type can be considered woodlands (relatively open grown), and many are classical closed forests. Differences may be due to inherent site conditions or to expressions of a developmental phase; fire frequency seems to play an important role as well.

Fire Regime Characteristics

Fires were frequent and of low intensity. Light surface fires burned as intervals averaging less than 10 years and as often as every 2 years (Dieterich 1980; Weaver 1951). The short fire-interval was caused by warm, dry weather common to the Southwest in early summer, the continuity of grass and pine needles, and the high incidence of lightning. Two fire seasons usually occurred each year, a major fire season after snow melt and just before the monsoon season in midsummer

Table 6-1-Occurrence and frequency of presettlement fire regime types by Forest and Range Environmental Study (FRES) ecosystems, Kuchler potential natural vegetation classes (1975 map codes), and Society of American Foresters (SAF) cover types. Occurrence is an approximation of the proportion of a vegetation class represented by a fire regime type. Frequency is shown as fire interval classes defined by Hardy and others (1998) followed by a range in fire intervals where data are sufficient. The range is based on study data with extreme values disregarded. The vegetation classifications are aligned to show equivalents; however, some corresponding Kuchler and SAF types may not be shown.

			Fire regime types						
			Understory		Mixed		Stand-replacement		
FRES	Kuchler	SAF	Occur[a]	Freq[b]	Occur	Freq	Occur	Freq	Nonfire
Ponderosa pine 21	SW ponderosa pine[c]	Interior ponderosa pine 237	M	1a:2-10	m	1			
	Arizona pine forest KO19		M	1a:2-10	m	1			
	Pine-cypress forest KO09	Arizona cypress 240			M	1,2	m	1	
Pinyon-juniper 35	Juniper-pinyon KO23	Rocky Mountain juniper 220			M	1			
	Juniper-steppe KO24	Western juniper 238			M	1			
		Pinyon-juniper 239			M	1			
		Arizona cypress 240			M	1			
Southwestern oaks[d]	California oakwoods KO30	Canyon live oak 249			M	1			
		California coast live oak 255			M	1			
		California black oak 246			M	1			
		Blue oak-digger pine 250	M	1	M	1			
	Oak-juniper KO31	Interior live oak 241			M	1			
Shinnery 31	Shinnery KO71	Mohrs oak 67			M	1			
Texas savanna 32	Ceniza shrub KO45				M	1			
	Mesquite savanna KO60	Mesquite 68, 242			M	1			
	Mesquite-acacia savanna KO61				M	1			
	Mesquite-live oak savanna KO62	Western live oak 241			M	1			
	Juniper-oak savanna KO86	Ashe juniper 66			M	1			
	Mesquite-oak savanna KO87				M	1			
Sagebrush 29	Sagebrush steppe KO55						M	2a:20-70	
	Juniper steppe KO24	Rocky Mountain juniper 220					M	2a	
	Great basin sagebrush KO38	Western juniper 238					M	2a:20-70	
	Wheatgrass-needlegrass shrubsteppe KO56						M	2a	
Desert shrub 30	Mesquite bosques KO27	Mesquite 68,242					M	1,2a	
	Blackbrush KO39						M	1,2a	
	Saltbrush-greasewood KO40						M	1,2a	
	Creosotebush KO41						M	1,2a	
	Creosotebush-bursage KO42						M	1,2a	
	Paloverde-cactus shrub KO43						M	1,2a	
	Cresotebush-tarbush KO44						M	1,2a	
SW shrubsteppe 33	Grama-tobosa KO58						M	1,2a	
	Trans-pecos shrub savanna KO59						M	1,2a	
Chaparral-Mountain shrub 34	Oak-juniper woodland KO31						M	1,2a	

(con.)

Table 6-I-Con.

FRES	Kuchler	SAF	Fire regime types: Understory Occur[a]	Understory Freq[b]	Mixed Occur	Mixed Freq	Stand-replacement Occur	Stand-replacement Freq	Nonfire
	Mountain mahogany-oak scrub KO37						M	1,2a	
	Transition of KO31 & KO37						M	1,2a	
	Chaparral KO33						M	1,2a	
	Montane chaparral KO34						M	1,2a	
	Coastal sagebrush KO35						M	1,2a	
Plains grasslands 38	Grama-needlegrass-wheatgrass KO64						M	1	
	Grama-buffalograss KO65						M	1	
	Wheatgrass-needlegrass KO66						M	1	
	Wheatgrass-bluestem-needlegrass KO67						M	1	
	Wheatgrass-grama-buffalograss KO68						M	1	
	Bluestem-grama prairie KO69						M	1	
	Mesquite-buffalograss KO85	Mesquite 68,242					M	1	
Desert grasslands 40	Grama-galleta steppe KO53						M	1,2a	
	Grama-tobosa prairie KO54						M	1,2a	
	Galleta-threeawn shrubsteppe KO57						M	1,2a	
Annual grasslands 42	California steppe KO48						M	1,2a	
Mountain grasslands 36	Fescue-oatgrass KO47						M	1	
	Fescue-wheatgrass KO50						M	1	
	Wheatgrass-bluegrass KO51						M	1	
	Foothills prairie KO63						M	1a	
	Cheatgrass[c]								

[a]M: major, occupies >25% of vegetation class; m: minor, occupies <25% of vegetation class
[b]Classes in years are 1: <35, 1a: <10, 1b: 10 to <35, 2: 35 to 200, 2a: 35 to <100. 2b: 100 to 200, 3: >200.
[c]This type was not defined by Kuchler.
[d]Added subdivision of FRES.

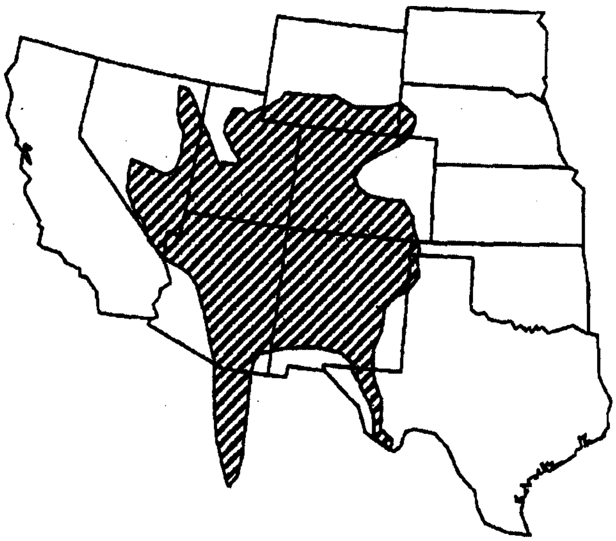

Figure 6-1-Southwestern ponderosa pine distribution.

and a secondary season in the fall. Once a fire started, the forest floor was generally consumed, but the damage to trees was highly variable. Low intensity surface fires predominated and were probably large where dry forests and adjacent grasslands were extensive such as on the gentle topography of high plateaus in Arizona and New Mexico. Damage to trees was highly variable but mortality to overstory trees was generally minor.

Fuels

The structural and compositional changes in Southwestern ponderosa pine over the past 100 years or more have been repeatedly documented (Biswell and others 1973; Brown and Davis 1973; Cooper 1960). What was once an open, parklike ecosystem, maintained by frequent,low-intensity fires, is now a crowded, stagnated forest. In addition to stand changes, general fire absence has led to uncharacteristically large accumulations of surface and ground fuels (Kallander 1969;

The natural accumulation of pine needles and woody fuels is exacerbated by the slowd ecomposition rates characteristic of the dry, Southwestern climate (Harrington and Sackett 1992). Decomposition rate (k) (Jenny and others 1949) is the ratio of steady state forest floor weight to the annual accumulation weight. Harrington and Sackett (1992) determined k values of 0.074, 0.059, and 0.048 forsapling thickets, polestands, and mature old-growth groves, respectively. Decomposition rates this slow, which Olson (1963) considers quite low, border on desertlike conditions. Humid, tropic conditions would have k values approaching 1.0 where decomposition occurs in the same year as the material is dropped on the ground.

Fuel loading estimates can be obtained from predictions based on timber sale surveys (Brown and others 1977; Wade1969; Wendel 1960) and using Brown's (1974) planar intersect method for naturally accumulated downed woody material. Forest floor weights

Figure 6-2--Typical Southwestern ponderosa pine fuels near Flagstaff, Arizona.

have been studied extensively in Arizona and New Mexico; results show high variability between sites. Ffolliott and others (1968, 1976 ,1977), Aldon (1968, and Clary and Ffolliott (1969) studied forest floor weights in conjunction with water retention on some Arizona watersheds. These and other works included prediction equations relating forest floor weight to stand basal area (Ffolliott and others1968, 1976, 1977), age (Aldon 1968), height and diameter (Sackett and Haase1991), and forest floor depth (Harrington 1986; Sackett 1985).

The forest floor consists of a litter (L) layer, recently cast organic material; a fermentation (F) layer, material starting to discolor and break down because of weather and microbial activity; and the humus (H) layer, where decomposition has advanced. The loosely packed L layer and upper portion of the Flayer provide the highly combustible surface fuel for flaming combustion and extreme fire behavior during fire weather watches and red flag warnings (fig.6-3). The lower, more dense part of the Flayer and the H layer make up the ground fuel that generally burns as glowing combustion.

Forest floor fuels (L, F, and H layers including woody material 1 ≤ inch diameter) were sampled in 62 stands in Arizona during the 1970s in Arizona and New Mexico (Sackett 1979). Throughout the Southwest, unmanaged stands of ponderosa pine had from 4.8 tons/acre (10.8 t/ha) in a stand on the Tonto National Forest to more than 20 tons/acre (45 t/ha) in a stand on the north rim of the Grand Canyon National Park. The next two heaviest weights (18.3 and 18.0 tons/acre) also occurred on the north rim of the Grand Canyon. Mean forest floor loading for the entire 62 stands measured was 12.5 tons/acre (28.0 t/ha). When woody material greater than 1 inch diameter was added, the average increased to 21.7 tons/acre (48.6 t/ha).The heavier material does not have much to do with extreme fire behavior, except as a spotting potential; these fuels do contribute to localized severity when burned. A range of forest floor fuel loadings is summarized in table 6-2.

Of the 12.5 tons/acre (28.0 t/ha) average of forest floor fuel load found in the Southwest, about 1.0 ton/acre (2.2 t/ha) was L layer material, 3.8 tons/acre (8.5 t/ha) was in the F layer, and 6.1 tons/acre (13.7 tha) was H layer. Small diameter woody material and other material comprised the remaining 1.8 tons/acre (4.0 t/ha). The large woody material that accounted for 42 percent of the total fuel loading, consisted of 1.4 tons/acre (3.1 t/ha) of material 1 to 3 inches (2.5 to 7.6 cm) in diameter, 5.0 tons/acre (11.2 t/ha) of rotted woody material 3+ inches in diameter, and 2.8 tons/acre (6.3 t/ha) of sound wood 3+ inches in diameter. See Sackett (1979) for complete summary.

Not only is there wide variation from site to site in the Southwestern ponderosa pine ecosystem, but vast differences exist within stands with respect to overstory characteristics (Sackett and Haase 1996). Experience indicates four separate conditions: sapling (doghair) thickets, polestands, mature old growth (yellow pine) groves, and open areas in the groves

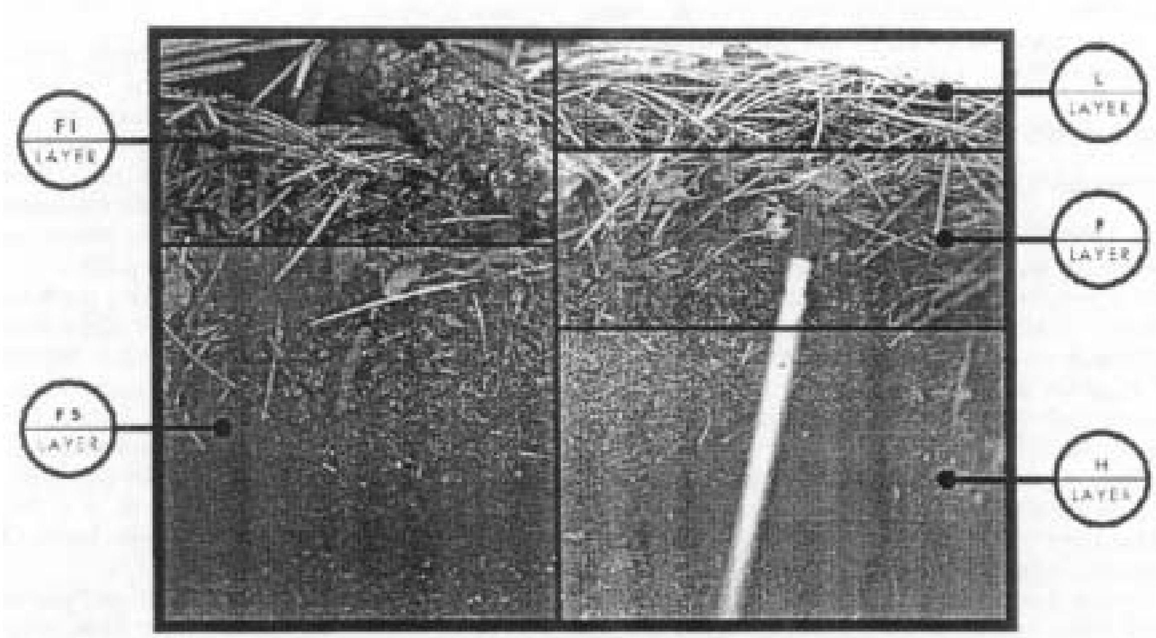

Figure 6-3--Section of ponderosa pine forest floor showing the fire intensity (FI) layer of feul and fire severity (FS) layer of fuel in relation to the L, F, and H layers of the forest floor.

Table 6-2-Average ponderosa pine surface fuel loadings (ton/acre) in the Southwestern United States by location (Sackett 1997).

Location	Number of sites	Forest floor and 0 to 1 inch diameter wood	Woody fuel >l-inch diameter	Total fuel
Kaibab NF	4	15.5	8.6	24.1
Grand Canyon NP	4	17.5	5.6	23.1
Coconino NF	4	14.7	19.8	34.5
Tonto NF	2	6.5	2.7	9.2
Apache-Sitgreave NF	14	11.3	11.2	22.5
San Carlos Apache IR	3	14.4	8.4	22.8
Fort Apache IR	2	15.1	20.5	35.6
Gila NF	10	11.2	7.3	18.5
Navajo IR	1	9.4	4.9	14.3
Cibola NF	3	8.8	8.8	17.6
Santa Fe NF	3	13.2	14.6	27.8
Carson NF	4	13.3	4.3	17.6
Bandalier NM	1	11.6	3.0	14.6
Lincoln NF	2	13.9	7.1	21.0
San Juan NF	5	11.9	4.8	16.7

without crowns overhead. Sapling thickets produce as much as 1.1 tons/acre per year of litter and woody fuels, pole stands 1.5 tons/acre per year, and mature, old-growth groves as much as 2.1 tons/acre per year. A substantial amount of forest floor material remains after an area is initially burned (Sackett and Haase 1996). The amount remaining varies due to the original fuel's configuration and the fire intensity and behavior, which are affected by the overstory condition. This amount persists even with repeat applications of fire. The charred condition of the remaining forest floor material resists re-ignition from the newly cast needles that are consumed quickly.

Postfire Plant Communities

Southwestern Ponderosa Pine

Pre-1900 Succession--Chronicles from 19th century explorers, scientists, and soldiers described a forest type quite different than what is seen today. The open presettlement stands, characterized by well-spaced older trees and sparse pockets of younger trees, had vigorous and abundant herbaceous vegetation (Biswell and others 1973; Brown and Davis 1973; Cooper 1960). Naturally ignited fires burning on a frequent, regular basis in light surface fuels of grass and pine needles maintained these forest conditions. Light surface fuels builtup sufficiently with the rapid resprouting of grasses and the abundant annual pine needle cast. Large woody fuels in the form of branches or treeboles, which fall infrequently, rarely accumulated over a large area.When they were present, subsequent fires generally consumed them, reducing grass competition and creating mineral soil seedbeds, which favored ponderdsa pine seedling establishment (Cooper 1960). These effects created an uneven-age stand structure composed of small, relatively even-aged groups.

The decline of the natural fire regime in these ecosystems started with extensive livestock grazing in the late 19th century when fine surface grassfuels were reduced (Faulk1970). Subsequently, pine regeneration increased because of reduced understory competition, less fire mortality, and more mineral seedbeds (Cooper 1960).

Pre-1900 Succession--In the early 190s forest practices, and reduced incidence of fire, led in directly to stagnation of naturally regenerated stands and unprecedented fuel accumulation (Biswell and others 1973). Stand-stagnation exists on tens of thousands of acres throughout the Southwest (Cooper 1960; Schubert 1974) and still persists where natural or artificial thinning has not taken place.

For several decades, trees of all sizes have been showing signs of stress with generally poor vigor and reduced growth rates (Cooper1960; Weaver 1951). This condition is likely due to reduced availability of soil moisture caused by intense competition and by moisture retention in the thick forest floor (Clary and Ffolliott 1969). Thick forest floors also indicate that soil nutrients, especially nitrogen, maybe limiting because they are bound in unavailable forms (Covington and Sackett 1984, 1992).

A combination of heavy forest floor fuels and dense sapling thickets acting as ladder fuels, couple dwith the normally dry climate and frequent lightning-and human-caused ignitions, has resulted in a drastic increase in high severity wildfires in recent decades

(Biswell and others 1973; Harrington 1982). Fire report summaries (Sackett and others 1996) show a great increase in the number of acres burned by wildfire since 1970 (fig. 6-4). Of all the years since 1915 with over 100,000 acres burned, almost 70 percent occurred between 1970 and 1990.

Another characteristic of today's Southwestern ponderosa pine stands is the sparseness of the understory vegetation, including pine regeneration. The thick organic layers and dense pine canopies have suppressed shrubby and herbaceous vegetation (Arnold 1950; Biswell 1973; Clary and others 1968). Natural regeneration is also limited to areas where the forest floor material has been removed either by fire or by mechanical means (Sackett 1984; Haase 1981). This condition has reduced the wildlife, range, and timber values of these forests and has generally minimized biodiversity.

Management Considerations—The need to alleviate the stagnated and hazardous forest conditions is a primary consideration in the management of Southwestern ponderosa pine stands. The restoration of forest health to the Southwest also needs to address the following concerns:

- Dwarf mistletoe, once held in check by periodic fires, is now a major cause of mortality in localized areas.
- Barkbeetle outbreaks are evident in overstocked stands that are stressed from the high competition for limited soil moisture, especially during drought years.
- Some amount of fire injury to the overstory is almost assured from the application of fire into an area. This may be in the form of crown scorch to the smaller trees and below ground injury to root sand root collars of the larger trees.
- Fuel conditions that contribute to the elimination of whole stands from wildfire need to be reduced. These conditions include heavy forest floor accumulations and ladder fuel conditions created from dense, stagnated sapling thickets.

Although the extent of these conditions will vary throughout ther egion, the combination of any of these situations on a particular forest creates a major concern and problem for the manager. Forest management objectives in the Southwest need to include the maintenance or improvement of existing old-growth stands and actions that promote the creation of future old growth stands.

Because recurrent fire was a primary element in sustaining presettlement forest health leading to the establishment and maintenance of old-growth stands, its use should be emphasized when restoring favorable conditions for ancient pine development. These conditions include low levels of dead organic material (fuels) to lessen the potential of high fire intensity and severity, and open stand structure to reduce crown fire

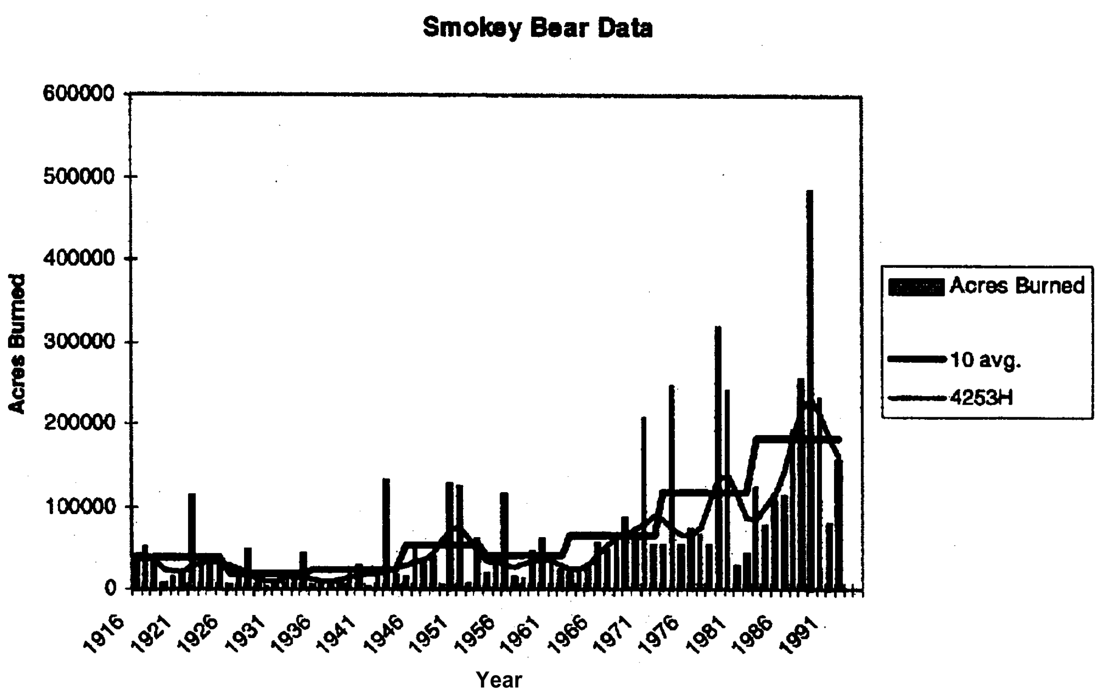

Figure 6-4--The total number of acres burned by wildfires in Arizona and New Mexico from 1916 to 1990, USDA Forest Service Smokey Bear fire summary reports. Heavy line represents 10-year average; light line represents trends using the 4253H mathematical filter, used for smoothing noise in data.

potential and intraspecific competition. Fire can be used to reduce fuel hazard, but its success is temporary. Failures denoted by too little or too much fuel consumption generally result from improper burn prescriptions and by attempting to correct long-term fuel buildup with one treatment. Cooper (1960) questioned whether prescribed fire could be used in the restoration of deteriorated forests. He concluded that planned burning would be too conservative and accomplish little, or would destroy the stand. While this observation has merit, with refined burning techniques as described in Harrington and Sackett (1990), it appears that fire could be applied sequentially to relieve the fuel and stand density condition. However, it is apparent that considerable large tree mortality could result. This seems to be an inescapable cost dictated by years of forest degradation.

Because of these consequences, special attention should be given to the excessive buildup of forest floor fuels in present old-growth sites. Burning of these deep forest floor layers can mortally injure the roots and cambiums of old pines, which previously survived many fires (Sackett and Haase 1996). Options for alleviating this condition are not ideal. Managers could simply accept a 20 to 50 percent loss of old growth in a single fuel-reduction burn as being a cost of decades of fuel buildup. Alternatively, the heavy accumulation of fuels could be manually removed from around the root-collar of the old-growth trees before the fire is applied. Currently, methods are being investigated that will make this mitigation method a feasible option for managers. The use of a burn prescription that removes a portion of the fuel accumulation has not been found for prescribed burning in the Southwest. If glowing combustion is able to begin in the deeper accumulations of material, high moisture content of that material may not prevent total consumption of the forest floor. Nearly complete burnout of duff has been observed in ponderosa pine forests at moisture contents up to 90 percent (Harrington and Sackett 1990) and in mixed conifers up to 218 percent (Haase and Sackett 1998).

In forest regions where old-growth pine groups are absent, designated areas based on site quality and existing stand types should be selected for creating future old growth. The best growing sites should be chosen because old-growth characteristics would be achieved more expeditiously than on poor sites. Moir and Dieterich (1990) suggested that 150- to 200-year-old ponderosa pine (blackjack pine) in open stands with no dwarf mistletoe be selected as the best stands to begin developing old growth. Through sequential silvicultural and fire treatments, the stands should be relieved of wild fire hazards and competition, allowing concentrated growth on a chosen group of trees. A long-term commitment is necessary, because another century may be needed before select old-growth pine is represent (Moir and Dieterich 1990). If younger stands are chosen for old-growth replacement, a greater commitment of time is required for thinning, slash disposal, commercial harvesting, and fire application.

Mixed Fire Regime

Major Vegetation Types

The pinyon-juniper woodlands (fig. 6-5) cover approximately 47 million acres (19 million ha) in the Western United States (Evans 1988) and are characterized by a large number of diverse habitat types that vary in tree and herbaceous species composition,and stand densities. Climatic and physiographic conditions vary greatly within the range of this vegetation type. Pinyon-juniper woodlands in the United States are commonly divided into the Southwestern and the Great Basin woodland ecosystems based on species composition. True pinyon is common in the Southwest and is usually associated with one or several species of junipers, including one-seed, Utah, alligator, and Rocky Mountain junipers. Singleleaf pinyon is identified with the Great Basin and is generally associated with Utah juniper. Other species of pinyon occur in southern California, Arizona, south of the Mogollon Rim, along the United States-Mexico border, and in Texas (Bailey and Hawksworth 1988). Several other species of junipers also are found in the West; one of the more

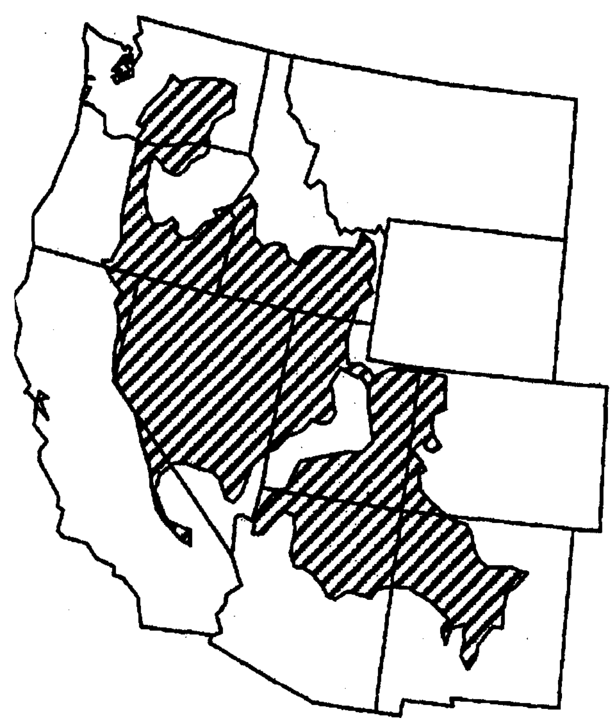

Figure 6-5–Pinyon-juniper woodlands distribution.

common is western juniper, which is found mainly in Oregon and eastern California. Stand densities and composition vary by elevation as it affects available moisture; drier sites tend to be occupied by junipers that are widely spaced and of low stature. Many of these sites are often classified as savannas. Higher elevation sites tend to be dominated by relatively dense stands of pinyon trees of comparatively tall stature and good form.

This report includes western oak species of obvious concern to resource managers but it does not include all oaks found in the Western United States (fig. 6-6, 6-7). Discussion concentrates on the important tree-form deciduous and live oaks of California and of the Southwestern United States (such as Gambel oak and Arizona white oak). These are generally addressed as a group. Little information has been documented for these species (McPherson 1992), but their importance to resource and fire management requires a beginning. Shinnery, predominantly composed of sand shinnery oak, is described as a separate ecosystem (fig. 6-8).

TheTexas savanna (fig. 6-8) as a mapped ecosystem occupies major portions of the Rio Grande Plains of south Texas, the Edwards Plateau of south central Texas and portions of the Rolling Plains, Grand Prairie, North Central Prairies, Blackland Prairies, and Cross Timbers. It corresponds roughly with Sections 315C, D, and E of Bailey's Ecoregions and Subregions of the United States (Bailey and others 1994) and with major

Figure 6-6--Western oak distribution.

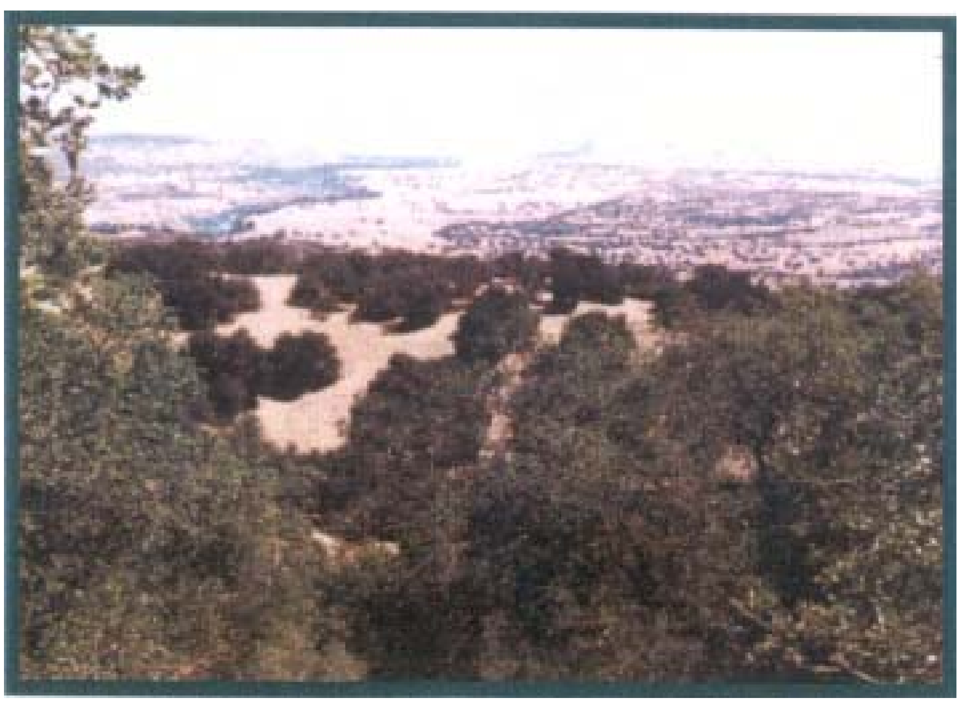

Figure 6-7—Western oak woodlands,Camp Roberts Military Training Reservation, Paso Robles, California.

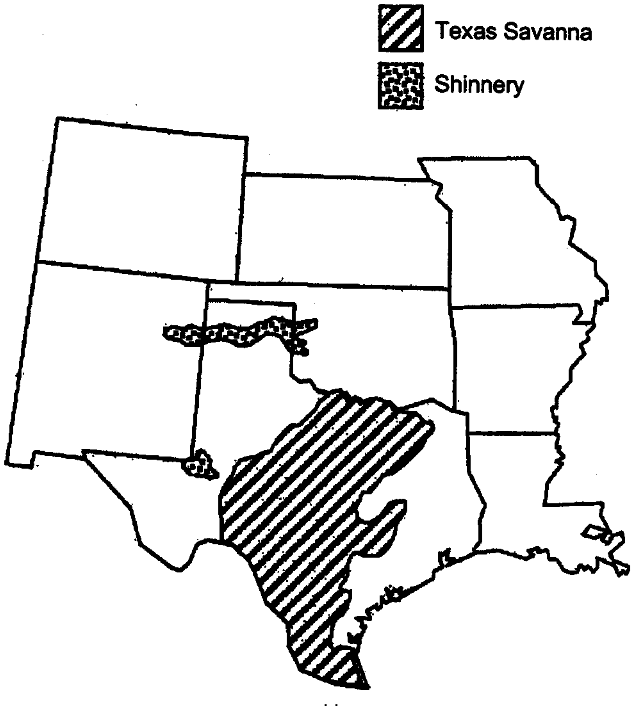

Figure 6-8–Distribution of shinnery and Texas savanna FRES ecosystems.

portions of the Rio Grande Plain and Edwards Plateau vegetation regions found in Box and Gould (1959). As a plant community type, however, it has significant representative elements that extend far north into the southern portion of the Plains Grasslands ecosystem. Infact, in the original Rainbow series volume, "Effects of Fire on Flora" (Lotan and others 1981), the area associated with the Texas savanna was lumped into one huge "Prairie Grasslands" type (which also included the grasslands of the "GreatValley" in California). The vegetation of the "Texas savanna" can be found in the northern portions of Texas in the southern Rolling Plains, the Grand Prairie, the North Central Prairies, the Blackland Prairies, and Cross Timbers, and extends to just south of the Texas Panhandle area (Box and Gould 1959). These other areas will be considered as part of the Texas savanna for the purposes of this publication. These areas receive 20 to 30 inches of precipitation annually—more than half of which falls during the warmest months, and less than a quarter during the period from December through March (Garrison and others 1977).

The vegetation is a savanna with an overstory layer of low trees and shrubs that varies from dense to open. This overstory is of variable composition, having broad-leaved and needle-leaved, deciduous and evergreen species that predominate. These are mesquite, aca-jcias, oaks, unipers, ceniza,and prickly pear species. Honey mesquite is the most widespread woody plant in the Texas savanna type and will receive the most discussion.The grass of the Texas savanna varies from short (< 2 inches) to medium-height (2 to 12 inches), and the herbaceous vegetation in general varies from dense to open. These understory plants are mainly bluestems, indian grass, and switchgrass in the northeast, gramas, buffalograss, Texas wintergrass and *Sporobolus* spp. in the south, central, and northwest, and curly-mesquite and tobosagrass in the west and on the Edwards Plateau. The particular mix of vegetation or specific plant community that one might encounter seems to be well correlated with soil orders, which are variable in the Texas savanna system (Garrison and others 1977).

Fire Regime Characterlstics

Long-term fire frequencies for the pinyon-juniper woodlands have not been clearly defined and are the topic of continuing study and discussion. However, there is agreement that fire was the most important natural disturbance before the introduction of livestock, particularly the large herds in the 19th century (Gottfried and others 1995). It is suspected that prior to the introduction of heavy livestock use, large areas of savanna and woodland periodically burned. These fires could have occurred during dry years that followed wet years when substantial herbaceous growth developed (Rogers and Vint 1987; Swetnam and Baisan 1996).

In the Intermountain West, presettlement mean fire intervals of less than 15 years were documented in the sagebrush steppe where western juniper now dominates (Miller and Rose 1999). Other knowledge that clearly documents the fire frequency, extent, and seasonality of long-term fire regimes was developed from a few studies at the upper limit of the pinyon-juniper type where it occurs with ponderosa pine. Fire scars are rare in living pinyon pines due to the tree's susceptibility to damage by fire or to rot fungi that enter resulting wounds. Fire scars have been noted on junipers but most members of this genus are difficult to age because of missing and false rings. Nonetheless, some fire frequencies have been determined for the Southwest. A sample of fire-scarred pinyon trees from three locations in the Sacramento Mountains in New Mexico indicated a mean fire interval of 28 years with a range of 10 to 49 years (Wilkinson 1997). Despain and Mosley (1990), working in the pinyon-juniper and ponderosa pine ecotone at Walnut Canyon National Monument in Arizona, reported a surface fire interval of approximately 20 to 30 years. Other studies by C. Allen and by T.Swetnam and his associates (Gottfried and others (1995), on productive sites in New Mexico, indicated that stand widefires, which covered more than 25 acres, occurred at 15 to 20 year intervals.

Dense pinyon-juniper stands (450 tree/acre or greater) can burn in crown fires under extreme weather conditions, generally low relative humidity and high windspeeds. The key conditions are a closed canopy to allow the spread of fire through the crowns and abundant dead material on the ground and as snags (Gottfried and others 1995). It appears that presettlement fire regimes in dense stands were a mixture of surface and crown fires, and that intensities and frequencies varied depending on site productivity. The Walnut Canyon site probably sustained patchy surface fires at intervals of 10 to 50 years and could carry crown fires at intervals of 200 to 300 years or longer.

On less productive sites with discontinuous grass cover, fires were probably infrequent and burns were small and patchy. Fire frequencies were probably greater than 100 years in these areas, but did occur more frequently under extreme conditions (Gottfried and others1995). However, where grass cover was more continuous, fire frequencies were probably more frequent (10-year interval or less) and tended to maintain these sites as savannas or grasslands. Surface fires would kill one seed juniper trees less than 3 to 4 feet (1m) tall (Johnsen 1962) but would have less of an impact on older, larger trees that have thicker bark and high crown base heights that exceed flame lengths. This relationship between height and susceptibility to fire also has been observed in western juniper stands (Dealy1990) and in Ashe juniper stands in Oklahoma (Wink and Wright 1973). Fast moving surface fires in the Southwest often do not burn near the trunks of larger trees because the litter layer does not ignite.

In the Great Basin, fire susceptibility depends on the stage of stand development (Meeuwig and others 1990). In young open stands, shrubs and herbaceous cover may be sufficient to carry fire, but this cover declines with time and eventually becomes too sparse as the trees develop. The trees, however, may still be too widely spaced to carry crown fires, except under severe conditions.

In recent centuries, fire regimes in Western oak forests were characterized by frequent, low intensity fires. This was probably due to use of these types by Native Americans, who probably carried out programs of frequent underburning. Higher intensity fires at long intervals have become more likely in the last half of the 20th century.

Few data are available on fire frequencies within the Texas savanna (Fuhlendorf and others 1996). With understory fuels usually exceeding 2,240lb/acre (2,000 kg/ha) each year under undisturbed conditions, it is quite likely that fire frequencies were less than 10 years, and potentially more frequent in the northeast portion of the Texas savanna. Fires occur most frequently during February and March when most grasses are dormant and lightning strikes occur commonly, and from July to September when grasses are dry. Both winter and summer fires with ample fuel loading in the grass understory can topkill trees resulting in major alteration of the woody physiognomy. However, woody plant mortality and stand-replacements are rare. Winter fires that occur with low understory fuel loadings can result in partial removal of the overstory (Ansley and others 1995, 1996b). Species such as mesquite, redberry juniper, and live oak sprout if topkilled by fire and are rarely removed from the vegetation complex by fire. However, Ashe(or blueberry) juniper, which occurs in south-central Texas, can be killed by fire and replaced by herbaceous vegetation.

Fuels

Pinyon-Juniper--The main fuel consideration is the amount of fine fuels, which varies with habitat type, stand history, and climatic conditions. Fuel loading information for woody material is not readily available; however, Perry (1993) measured an average of 20 tons/acre (45 t/ha) after a pinyon-juniper clearcutting operation in Arizona; this stand produced about six cords/acre of fuelwood. Fuel loadings of more than 11 tons/acre are considered heavy. Slash left in partially harvested woodlands may provide fuel ladders for ground fires to spread into the canopies. Grass understory loadings can range from sparse to abundant (200 to 600lb/acre). Typical crown fuels are 3.6 tons/acre (8.1 t/ha) forfoliage and 1.8 tons/acre (4.0 t/ha) for 0 to 0.25 inch branchwood (Reinhardt and others 1997).

Western Oaks--Fuels are quite variable between stands, depending upon species, site, and stand condition. For example, a closed-canopy canyon live oak forest may have little or no live understory. Surface fuels will be made up of leaf and branch litter and the amount will depend upon the time since last fire in the stand. A more open stand may have an understory of shrubs and non woody species. A closed forest of a deciduous species, for example California black oak, may well have an understory of annual grass; but a more open woodland of the same species may have a mix of grass and shrubs as an understory. In the latter case, the combination of grass and shrubs can provide a fuel ladder complex with associated erratic and potentially dangerous fire behavior.

The aerial fuels in these oak stands are variable too. Little information exists to characterize the deciduous species; however, the live oaks can be thought of as roughly comparable to chaparral in terms of crown fuel character—both being sclerophyllous in nature. The green material in these species will burn if fuel moisture is low enough.

TexasSavanna--The predominanat fuel that contributes to a fire's propagation is the herbaceous understory. However, if the mesquite overstory has dead stem material, it can be ignited and potentially kill the plant. Britton and Wright (1971) found that up to 24 percent of mesquite that had been sprayed with a topkilling herbicide were killed with fire that occurred 4 years after spraying. The standing dead stems burned into liveroot crowns. When the overstory is dense---either from a high density of individuals, or from dense resprouted material—a crown fire can be sustained, given the necessary wind and moisture conditions. Such a high density overstory can be found as a phase in Texas savanna stands. Mesquite crown fires would only occur in summer months because the plant is winter deciduous. However, other species of the savanna complex, such as junipers and liveoak, could carry crown fire any time of the year.

Herbage production, which indicates potential fine fuel loading in the understory, was divided into four major productivity classes (Garrison and others1977):

Class	Productivity (lb/acre)
1	2,250 to 3000+
2	1,500 to 2,250
3	750 to 1500
4	0 to 750

Postfire Plant Communities

Pinyon-Juniper

Pre-1900Succession--The pinyon-juniper woodlands are diverse, and successional pathways differ by habitat type throughout the West. Traditional succession toward a "climax"vegetation considers the continuous replacement of one community by another. The driving force in the successional process is competition among plant species of different genetically controlled capabilities responding to changes in the environment (Evans1988). In the woodlands, succession involves the same species but in different amounts and dominance over the landscape. Several successional seres following stand replacing fires have been proposed for the Southwestern or Great Basin pinyon-juniper woodlands. Most of the successional projections are based on stands that had been grazed in the past. Arnold and others (1964), working in northern Arizona, developed one of the first models. A model for southwestern Colorado (Erdman1970), similar to that of Arnold and others (1964), progresses from skeleton forest and bareground, to annual stage, to perennial grass-forbstage, to shrubstage, to shrub-open tree stage, to climax woodland. This pattern takes approximately 300 years; however, new fires could set back succession before the climax is achieved. Arnold and others (1964) indicated that tree reoccupation progressed from the unburned stand inward toward the center of the burn. Barney and Frischknecht (1974) reported a sere for a Utah juniper stand in west-central Utah where pinyon was a minor component.

This ecosystem has had a long history of heavy grazing since the late 19th century. The postfire progression went from skeleton forest and bareground, to annual stage, to perennial grass-forb stage, to perennial grass-forb-shrub stage, to perennial grass-forb-shrub-young juniper stage to shrub-juniper stage, and to juniper woodland. Junipers were well developed 85 to 90 years after a fire. They indicated that the speed of tree recovery would depend on the stage of tree maturity at the time of the fire; older seed producing stands would recover more rapidly than younger, immature stands.They noted the importance of animal transport and storage of juniper seeds in the speed of tree recovery. As new juniper could start producing seed within about 33 years of establishment, hastening tree recovery.

Post 1900 Succession-- Data on successional trends apparent in the 1900s show that on similar sites succession may follow several pathways (Everett 1987a; Everett and Ward 1984). Shrubs, rather than annuals, have been the initial vegetation on some burned sites (Everett and Ward 1984), while the shrub stage may be reduced or absent on some New Mexico sites (Pieper andWittie1990). Predicting the course of succession is difficult since it depends on a number of factors (Everett 1987a). Specific successional pathways depend on fire severity and related damage to the original vegetation, area burned, available seed sources either in the soil or from adjacent areas, species fire resistance and ability to reproduce vegetatively, site conditions, and climatic parameters throughout the successional process. Everett and Ward (1984) indicated that the "initial floristic model" is appropriate after a burn; initial species composition and density may be as or more important than the progressive succession. Most preburn species returned within 5 years of a prescribed burn in Nevada (Everett and Ward 1984) and in southern Idaho (Bunting 1984).

The major human influence on the pinyon-juniper woodlands and fire's role in these ecosystems has been ranching. Most of the Western rangelands were overgrazed, especially in the period following the 1880s. Some areas around the Spanish controlled areas of New Mexico have been heavily grazed since the16th century. Overgrazing has had an important effect on the role of fire in the woodlands. The reduction of cover ofherbaceous species resulted in insufficient fuels for fires to spread and to control tree establishment. Fires ignited by lightning or humans tend to be restricted in space. Fire suppression activities by land management agencies also reduced the occurrence of fires.

Woodland and savanna stand densities have increased throughout most of the West. Some people believe that the woodlands have invaded true grasslands because of the lack of fire, but this is open to debate (Gottfried and Severson 1993; Gottfried and others 1995; Johnsen 1962; Wright and others 1979). Climatic fluctuations, such as the drought in the Southwest in the early 1950s, and global climate change also have affected the distribution of woodlands in the West. In the Intermountain West, Miller and Rose (1999) quantitatively established that the co-occurrence of wet climatic conditions, introduction of livestock, and reduced role of fire contributed to the postsettlement expansion of western juniper. Prior to 1880, fire was probably the major limitation to juniper encroachment. Other human influences related to the harvesting of wood products by early American Indians (Gottfried and others 1995) and the harvesting of large quantities of fuelwood to make charcoal for the mines and domestic wood for supporting populations in Nevada (Evans 1988) and near Tombstone in Arizona.

Management Considerations-- During the 1950s and 1960s, large operations were conducted to eliminate the pinyon-juniper cover in the hope of increasing forage production for livestock (Gottfried and Severson 1993; Gottfried and others 1995). Other objectives were to improve watershed condition and wildlife habitat. Mechanical methods, such as chaining and cabling, were used and resulting slash was piled and burned. Burning these large fuel concentrations generated high heat levels that damaged soil and site productivity (Tiedemann 1987). Many of these piled areas were sterilized and remain free of vegetation after over 20 years. Individual tree burning was used on some woodland areas. Most of the control operations failed to meet their objectives. Many areas failed to develop sufficient herbaceous cover to support renewed periodic surface fires.

A relatively undisturbed site with a rich variety of understory species may recover differently than an abused site with little understory development. Similarly, an older stand of junipers with a less diverse population of perennial species will recover differently than a younger stand (Bunting 1984). Burning in stands with few desirable understory species may worsen the ground cover situation, and depending on the characteristics of the tree component, destroy a valuable wood resource (Everett 1987b). A potential problem exists if the preburn or adjacent vegetation contains undesirable species, such as red brome. Very hot fires can seriously slow initial succession of desirable species (Bunting 1984). Everett and Ward (1984) indicated that relay floristics, the migration of species into the site, is more important for the later stages of development. Wink and Wright (1973) found that soil moisture was important in determining rate of understory recovery; it is more rapid when soil moistures are high. Dry conditions may increase drought stress of surviving herbaceous plants (Wink and Wright 1973) and retard seed germination. Aspect and elevation can be used to predict some general successional trends (Everett 1987a).

Currently, prescribed fire isused to reduce accumulations of slash from fuelwood harvesting or to reduce or eliminate the tree cover in an attempt to increase range productivity and biodiversity. In Arizona, slash is usually left unpiled. Small piles are constructed occasionally and are burned as conditions and crew availability allows. There is increasing interest in managing the pinyon-juniper woodlands for sustained multi-resource benefits including, but not limited to, tree products, forage, wildlife habitat, and watershed protection (Gottfried and Severson 1993). This is particularly true for high site lands that have the ability to produce wood products on a sustainable basis. Prescribed burning to dispose of slash is less desirable in partially harvested stands, where the selection or shelterwood methods have been used to sustain tree product production. Burning tends to damage residual trees, especially where slash has accumulated at the base, and advance regeneration. Established, smaller trees are particularly important for the next rotation because of the difficulty of achieving adequate regeneration of these relatively slow growing species. It may be desirable to move slash away from areas of satisfactory regeneration prior to burning or to avoid burning in them.

Several different slash disposal options may be applicable to any one management area (Gottfried and Severson 1993). Burning of large piles is unacceptable because of soil site degradation (Tiedemann1987) and no longer recommended in the Southwest (USDA Forest Service 1993). However, small piles of slash may be burned in low intensity fires to encourage floristic richness or to promote temporary increases of nutrient content in herbaceous vegetation. Piled or unpiled slash can also be left unburned to provide habitat for small mammals or to break up sight distances for wild ungulates. It also can be scattered to provide protection for establishment of young trees and herbaceous species, and to retard over land runoff and sediment movement.

Mechanical methods of clearing pinyon-juniper are increasingly expensive, but prescribed fire is an economical alternative.The method used in Arizona is to ignite the crowns from prepared fuel ladders of cut lower limbs that are piled around the base of the tree. Ladders are ignited one season after the limbs are cut. In denser stands, fire spreads into the crown layer and through the stand from fuel ladders that are created below strategically placed trees. A method used in central Oregon on sites converted to juniper from sagebrush/grass is to conduct prescribed fires several years after harvesting trees. The increased production of herbaceous vegetation following cutting provides fuels to carry the fire, which reduces residual slash and kills juniper seedlings.

Research in the Great Basin suggests that fire works best on sites with scattered trees (9 to 23 percent cover) where the trees begin to dominate the understory and in dense stands (24 to 35 percent cover) (Bruner and Klebenow1979). Wright and others (1979) indicated that prescribed spring burning was successful in sagebrush/pinyon-juniper communities. Bruner and Klebenow (1979) recommended an index to determine if a fire will be successful or if conditions are too dangerous. This index is based on the addition of maximum wind speed (mi/hr), shrub and tree cover (percent), and air temperature (°F). Burning can be successful if scores are between 110 and 130. Dense stands where pinyon is more common than juniper are easier to burn than pure juniper stands (Wright and others 1979). Bunting (1984) indicated that burning of western juniper stands in southwestern Idaho was only successful during the mid-August to mid-September period; burning in the fall did not achieve desired results because of low temperatures, low wind speeds, and lack of fine fuels. Prescribed fire can be used in previously treated areas to control new tree regeneration. This technique works best if the area is ungrazed for one or two seasons prior to burning.Wink and Wright (973) reported that a minimum of 890 lb/acre (1,000) kg/ha) of fine fuels is needed to burn and kill Ashe juniper seedlings and to burn piled slash. Success where alligator juniper dominates has been limited because of the trees' ability to sprout, so prescribed fire is not recommended (USDA Forest Service 1993).

Ecosystem Management-- Reintroducing low intensity fire into the pinyon-juniper woodlands could help meet ecosystem management goals. For example, prescribed fire could be used after harvesting to limit tree regeneration and to maintain overstory stand densities that would promote vigorous understory vegetation for livestock and wildlife. Fire could be used during the earlier part of the rotation period, when crown cover is less, and modified later to protect adequate tree regeneration. The prescription would vary by the amount and condition of woody debris in the stand so that stand replacing crown fires are prevented. Pockets of regeneration could be protected.

Fire could also be used to maintain herbaceous cover dominance in natural savannas and ecotonal grasslands. However, as indicated above, all surface fire options would require that the land be rested from grazing prior to treatment so that suficient fuels can develop to carry the fire. It usually requires 600 to 700) lb/acre (672 to 784 kg/ha) of fine fuel to carry a fre in the Great Basin (Wright and others 1979).

Fire has also been used to create mosaics of woodland and openings within some Southwestern landscapes. Mosaics are beneficial to wildlife and livestock (Gottfried and Severson 1993) and can create an aesthetically pleasing landscape. Aerial and ground firing techniques have resulted in mosaics on some juniper/mesquite grasslands in southern Arizona.

Western Oaks

Pre-1900 Succession--There is little doubt that western oak trees evolved over a time when climatic change was occurring and when disturbance including fire was common. The deciduous or evergreen habit probably is related to environmental moisture--evergreen oaks belonging to more arid systems (Caprio and Zwolinski 1992; Rundel 1987). Postfire succession during pre-Euro-American settlement was probably much like the dynamics that we see today, but there were probably more oaks than we find today. Some species were easily top-killed; many species sprouted in response to fire.

Post-1900 Succession-- The current reduction in the occurrence of the oaks in many areas may be due to a number of factors, including increased fire severity, grazing,over removal to provide more pasture land, and urban encroachment. Fire is probably not the primary factor, but it can kill a stand of oaks outright. Some oaks are more easily top-killed than others, which is generally a function of bark thickness. See the categorization of oak sensitivity to fire by Plumband Gomez (1983). Almost all of the oak species sprout after fire, if root crown or underground portions are still alive (Plumb 1980).

Management Considerations--In some parts of the West, oaks have become subjects of intense resource management interest. The ranges of some species have become severely reduced; some species do not seem to be reproducing at a desired rate (Bartolome and others 1992). Competition to seedlings from understory vegetation may be hampering seedling survival (Adams and others 1992); grazing may play a part as well. Effective management of these species has yet to be established. The use of prescribed fire as a means of reducing competition and opening up closed canopy stands is being attempted (Clary and Tiedemann 1992). Although results are not definitive yet, it shows promise. For now, the use of prescribed fire in western oaks should be approached with caution and patience. Some species are sensitive to fire (table 2-1) but may survive under certain conditions (Paysen and Narog 1993). Many oaks seem to be prone to disease, such as heart rot. Injury from fire or other treatment may not kill a tree, but might conceivably inflict damage that could provide a port of entry for disease. Much research remains to be done on these species. For now, management treatments should be carried out carefully.

Texas Savanna

Pre-1900 Succession—Historical accounts differ as to original density and distribution of mesquite in Texas. Bartlett (1845) described much of Texas rangeland as open grasslands with scattered large mesquite (a mesquite savanna). Marcy (1866) described some upland areas of centralTexas as "covered with groves of mesquite trees," and an area in the lower Texas Panhandle as "one continuous mesquite flat, dotted here and there with small patches of open prairie." These observations suggest that honey mesquite was a natural part of the northern Texas vegetation complex prior to Euro-American settlement and, apparently in some instances, occurred as dense stands. There is no indication as to the growth form of mesquite trees prior to Euro-American settlement. Fire was apart of the environment when these explorers traveled through Texas (Wright and Bailey 1980), but the specific role it played in shaping the scenes they observed is difficult to know. However, biological agents and fire are credited with having limited mesquite densities on rangelands before Euro-American settlement in the Southwest (Jacoby and Ansley 1991).

Post-1900 Succession—Honey mesquite density increased in the Southwest during the 20th century. It is likely that most of the multistemmed thickets that occur in Texas today have greater stem and foliage density because of increased anthropogenic disturbance of the canopy (including use of fire to topkill shrubs, which induces sprouting) than would have occurred naturally. Individual shrub densities have increased since the late19th century as well. This has also been attributed to human influence—either through suppression of natural fires, or dissemination of mesquite seed by the herding and migration of domestic livestock (Brown and Archer 1989).

Much of thev egetation in the Southwest is in a state of flux and may have been changing for centuries in many areas. This seems to be particularly true for the Texas savanna type. Its dynamics, however, may have been accelerated by the influence of recent human activities.

Intensive animal grazing coupled with extremes of climate may be instrumental in causing active fluctuation of vegetation composition and physiognomy. Domestic livestock have played a major role in dissemination of mesquite seed into mesquite-free areas (Archer 1995; Brown and Archer 1989). Observations of recently seeded Conservation Reserve Program (CRP) stands on cropland near mesquite stands indicates that in the absence of cattle grazing, mesquite seeds were probably deposited by wildlife (coyotes, hogs, birds). However, this appears to be restricted to the margins of already existing mesquite stands. Early settlers accelerated dissemination into mesquite-free areas first via the cattle drives that occurred about 1900, and second with continuous grazing within fenced areas.

Current landscape patterns may reflect a trend that has been ongoing for centuries, or phases in a pulse equilibrium that may exist in much of the Southwest. The current pattern may depend upon recent combinations of weather and human activity. Mesquite encroachment, or encroachment of other woody species would probably occur in the absence of domestic livestock grazing, but such grazing has probably accelerated this process.

Management Considerations-- Historically, the Texas savanna has provided a home to an abundance of wildlife. But, in recent times, landclearing for agricultural purposes has reduced the habitat for some of these species (Garrison and others 1977). Livestock grazing has been a predominant factor in managing this vegetation type. The woody overstory plants of the savanna, especially mesquite, have been viewed as pests by most landowners. Mesquite's thorny branches, increasing density on rangeland, and perceived competition with forage grasses have made it the target of eradication efforts over recent years. Chemical and mechanical controls have been the primary agents used in this effort (Fisher 1977). More recently, fire has gained increased acceptance as a management tool (Wright and Bailey 1982).

Mesquite now has an emerging image as a resource that should be managed rather than eradicated (Ansley and others 1996a; Fulbright 1996; Jacoby and Ansley 1991). Unfortunately, decades of control attempts have destroyed many mature stands of mesquite that contained single to few-stemmed trees. These trees were desirable in that they occupied far less surface area than multistemmed growth forms that resulted from destruction of aerial tissue and subsequent resprouting. Complete elimination of mesquite has been a goal that few landowners have achieved, and the concept of complete removal is questionable, both economically and environmentally (Fisher 1977).

Mesquite has many potential benefits to the ecosystem when maintained at controlled densities such as in a savanna. Such benefits include nitrogen fixation, livestock shade, habitat for nesting birds, and the potential as firewood or wood products. Mesquite has the potential to produce commercial hardwood in some regions with higher rainfall (Felker and others 1990). In lower rainfall areas, shrubby growth forms of mesquite can have other benefits, such as wildlife habitat. A mesquite savanna offers a pleasant landscape and may improve the value of a property over either an unmanaged woodland or a treeless grassland.

Recent research suggests that mesquite savannas can be sustained as long as the herbaceous understory is maintained at sufficient densities to outcompete mesquite seedlings (Archer 1989; Brown and Archer 1989; Bush and VanAuken 1990). A savanna of this nature can be created and maintained in large part by using prescribed fire—one of the more environmentally acceptable and most economically sustainable options for managing woody plants (Ansley and others 1996a). In the initial stages of stand treatment, herbicides may be a useful supplement to the use of prescribed fire. However development and maintenance of the desired savanna growth form can often rely on the use of low-intensity fire, which can be achieved by burning under certain fuel loadings, humidities, and air temperatures. Creating a savanna from thickets using low-intensity fires will take time and should be part of a long-term management plan.

Response of honey mesquite to fire is highly variable and is a function of fine understory fuel loading and condition and of season of the year (Ansley and others 1995; Lotan and others 1981; Wright and others 1976; Abundant fine fuels tend to produce hotter fires and result in more topkill of the woody plants than lighter loadings (Wright and Bailey 1982). Summer fires will produce more topkill than winter fires (Ansley and others 1998). Fine herbaceous fuel loading and season of the year can work in various combinations to produce partially defoliated mesquite, or completely topkilled mesquite that quite often produces abundant sprouts from the root crown. Mesquite age also affects survival of individual plants after fire. Individual trees 1.5 years of age or less are easily killed by a fire when the soil surface temperatures are above 500 °F (260 °C) (Wright and others 1976). At 2.5 years of age, they can be severely harmed, and if older than 3.5 years, they are seemingly fire resistant at these soil temperatures.

Stand-Replacement Fire Regimes

Major Vegetation Types

The major vegetation types within this fire regime type are varied. Broadly, they include grassland and shrubland vegetation types (fig. 1-2).

Grasslands

The grassland types (fig. 6-9) include:

- The **plains grasslands**, which range from Canada south to northern Texas in a broad swath that covers much of the Mid-Western United States.
- The **mountain grasslands**, which consist of open, untimbered mountainous areas from Canada south through the Northern and Central Rocky Mountains and the Coastal Range.
- The **desert grasslands**, which occur in the Southwestern States and in the Great Basin.

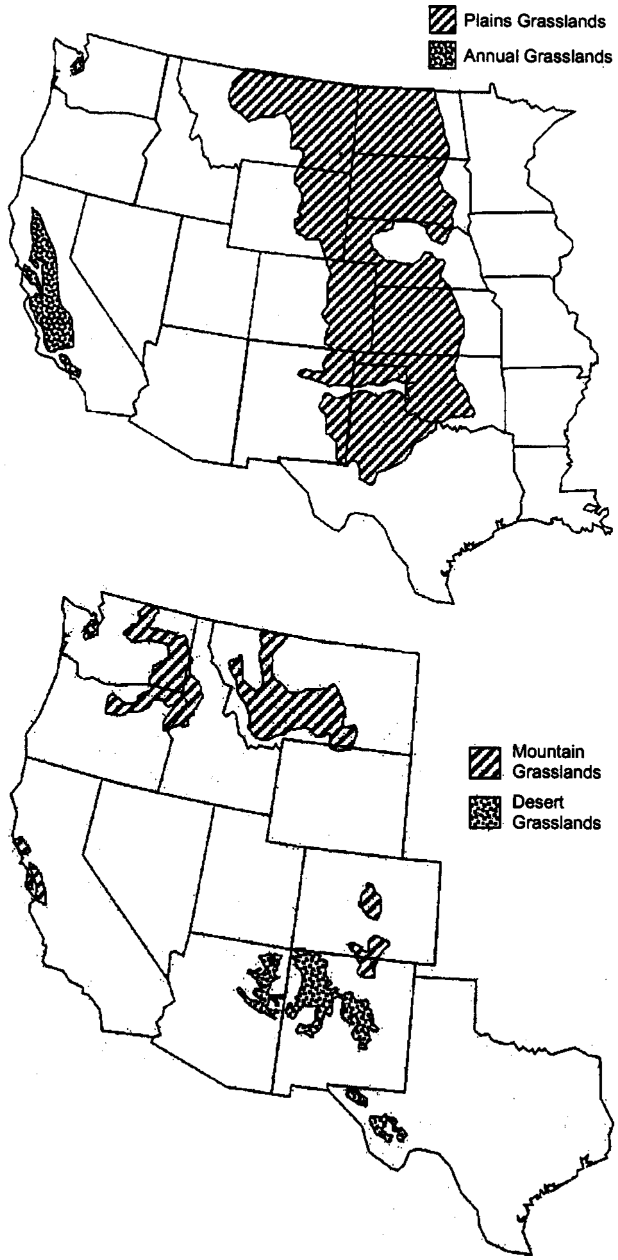

Figure 6-9--Distribution of plains, mountain, desert, and annual grassland FRES ecosystems.

- The **annual grasslands**, which are concentrated for the most part in the valleys and foothills of California and along the Pacific coast.
- **Cheatgrass** (fig. 6-10), which has invaded and gained dominance in many plant communities in the Intermountain and Columbia Basin regions (Monsen 1994).

Figure 6-1--Cheatgrass

Shrublands

Shrublands are described here as desert shrubland types and the chaparral-mountain shrubtype. Desert shrublands transcend North America's four major deserts-Mojave, Sonoran, Chihuahuan, and Great Basin (fig. 6-11, table 6-3). These deserts encompass about 600,000 square miles (1,717,000 km^2) within the physiographic Basin and Range Province, surrounded by the Rocky Mountains and Sierra Nevada in the United States, and the Sierra Madre Occidental and Sierra Madre Oriental in Mexico (MacMahon 1988; MacMahon and Wagner 1986). They are characterized by low but highly variable rainfall, 10 inches/ year 25 cm/year), and high evapotranspiration. Each desert differs in precipitation patterns, temperature variables, and vegetation structure (Burk 1977; Crosswhite and Crosswhite 1984; MacMahon 1988; MacMahonand Wagner 1986; Turner and Brown 1982; Turner and others 1995).

Bailey's (1978) Desert Division includes Mojave, Sonoran, and Chihuanhuan Deserts, considered warm deserts because their precipitation is mostly rain. The Mojave receives winter rainfall, the Chihuahuan summer rainfall, and the Sonoran both. Winter rainfall tends to be of long duration, low intensity, and covers large areas, where as summer rainfall is of short duration, high intensity, and covers limited areas (MacMahon 1988). The Mojave Desert has greater elevation and temperature variations than the Sonoran

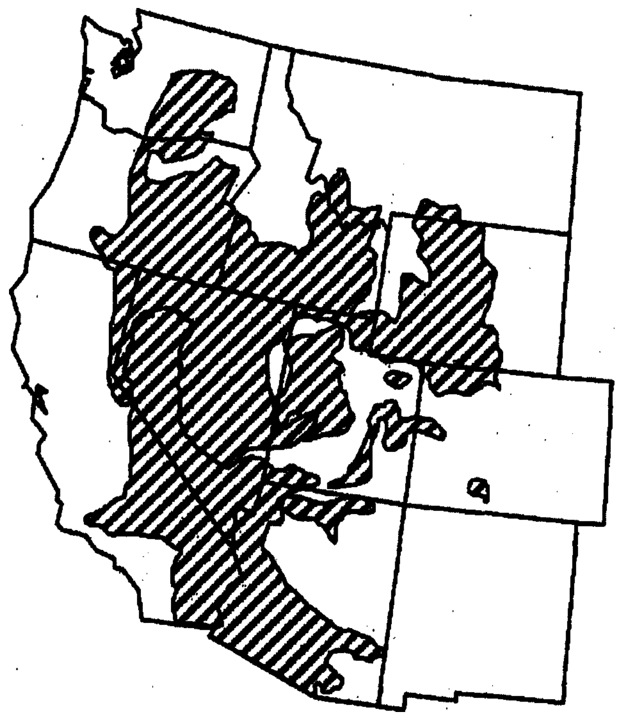

Figure 6-11--Distribution of desert shrub FRES ecosystems.

Table 6-3 –Physiognomic fuel types for desert shrublands associated with the four North American deserts.

Desert shrublands	North American deserts			
	Chihuahuan	Sonoran	Mojave	Great Basin
Sagebrush F-29				
Great Basin sagebrush K-38			X	X
Desert shrub F-30				
Blackbrush K-39			X	X
Saltbush/greasewood K-40	X	X	X	X
Creosotebush K-41	X	X	X	
Creosotebush/bursage K-42		X	X	
Mesquite bosques K-27	X	X	X	
Paloverde/cactus shrub K-43	X	X		
Southwestern shrubsteppe F-33				
Grama/tobosa shrubsteppe K-58	X	X		
Trans-Pecos shrub savanna K-59	X	X		

[a]FRES(F) shrubland ecosystems and the Kuchler Potential Vegetation System (K) equivalents (Garrison and others 1977).

Desert, which is lower, flatter, and warmer. Although the Chihuahuan Desert lies south of the Sonoran, it varies more in elevation and has colder winters.The Mojave Desert is considered transitional between the Sonoran and Great Basin Deserts, respectively, sharing components of each at its extreme southern and northern ends. The Great Basin desert is considered a cold desert because its precipitation is primarily snow (MacMahon 1988).

Vegetation in these regions varies from predominantly short grass prairie, consisting of sparsely distributed bunchgrasses, to predominantly shrubs, sometimes with scattered small trees, and often with exposed areas of soil (fig. 6-12). Desert and desert shrubland vegetation has been classified in numerous ways (Shreve and Wiggins1964; Turner 1982; Turner and Brown 1982; Vasek and Barbour1977). We focused on desert shrublands within the United States (table 6-1).

Figure 6-12–Bare soil, evident between shrubs and small trees, is a common characteristic of North American deserts as seen in the Mojave Desert, California.

Although these shrublands are distributed as a continuum of natural ecosystems, the use of vegetation classification systems gives us a convenient functional format for making fire management decisions. For our purposes, desert vegetation will be subdivided according to the FRES ecosystems as organized in table 6-3. We included the FRES sagebrush and Southwestern shrubsteppe types in our description of desert shrublands based on their similar fuels types, geographical proximity, and species integration.

Great Basin Sagebrush--This type characterized by sagebrush species (fig. 6-13) covers plateaus and vast plains at elevations ranging between1,600 and 11,000 feet (490 and 3,500 m) with varied soils derived from lava flows, ancient lakebeds, and alluvium (Garrison and others (1977). The Great Basin sagebrush, the largest range ecosystem in the Western United States, covers about 247 million acres (100 million ha) of arid lands (Blaisdell and others 1982). Sagebrush and associates are valuable for soil stabilization, wildlife habitat, animal feed, and ecosystem stability.There are about 22 species and subspecies; some have been studied extensively (Harniss and others 1981; Koehler 1975; Monsen and Kitchen 1994; Roundy and others 1995; Tisdale and Hironaka 1981). Sagebrush, composed of dwarf and tall sagebrush species, range between 1 and 7 feet (0.3 and 2m) tall and grow in dense clumps or scattered plants. Shadscale, spiny hopsage, Mormon tea, and milkvetch are important codominants in this vegetation type. Understory grasses such as wheatgrass, brome, fescue, and bluegrass, and variable forbs form discontinuous patches with bare soil.

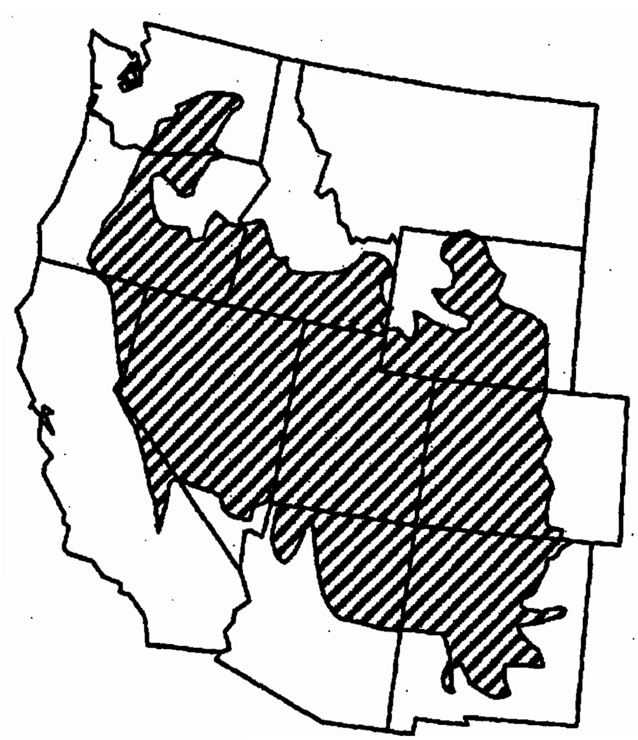

Figure6-13-Distribution of Great Basin sagebrush FRES ecosystems.

Blackbrush-- This type is composed of dense to scattered low stature shrubs and dense to open grass at elevations below 6,550 feet (2,000 m) (fig. 6-14). Blackbrush is one of the least studied landscape dominant shrubs in the United States. It prefers level topography and is not common on slopes or in drainages (Lei and Walker 1995). It maintains the highest cover of any desert shrub community. This transitional community between the Great Basin and the Mojave Desert occurs where annual precipitation is about 7 inches (18 cm) (MacMahon 1992). Moisture may limit its range. Blackbrush usually occurs in almost purestands, although it intergrades with creosotebush and bursage at lower ecotones and sagebrush/juniper ecotones at higher elevations (Lei and Walker 1995).

Saltbush Greasewood-- This shrubland is characterized by halophytes and succulent subshrubs. Vegetation dominants includes had scale, black greasewood, and saltbush with saltgrass, winterfat, and sagebrush also present. This shrubland is common to all four deserts (table 6-3) and occurs on approximately 42 million acres (17 million ha) on heavy depauperate soil, often with underlying hardpan or alkaline flats. It is found below the sagebrush zone, generally below elevations of 6,900 feet (2,100 m). Saltbush and black greasewood are dominant and co-dominant species throughout much of their range from Canada to northern Mexico, eastern California to Colorado and northeast Montana.

Creosotebush--This vegetation consists of low to medium-tall, typically open shrubs (fig. 6-15) that grow on bajadas, valley floors, gentleslopes, sand dunes, and in arroyos below 5,000 feet (1,500 m) in the Mojave, Sonoran, and Chihuahuan Deserts. Creosotebush is a widespread dominant or co-dominant that also forms transitional vegetation between the three warm deserts. Creosotebush occurs in mixed to pure stands of open, low but variable diversity plant communities on about 46 million acres (18.4 million ha) (Cable 1973).

Joshua Tree--In parts of the Mojave Desert, creosotebush is associated with the Joshua tree woodland (fig. 6-16). Joshua trees can resprout after fire, develop fire-resistant bark on trunks, have protected apical meristems usually high above surrounding fuels, and reseed from offsite sources. Resource manager at the Joshua Tree National Monument in California are testing prescribed burning as a tool to create fuel

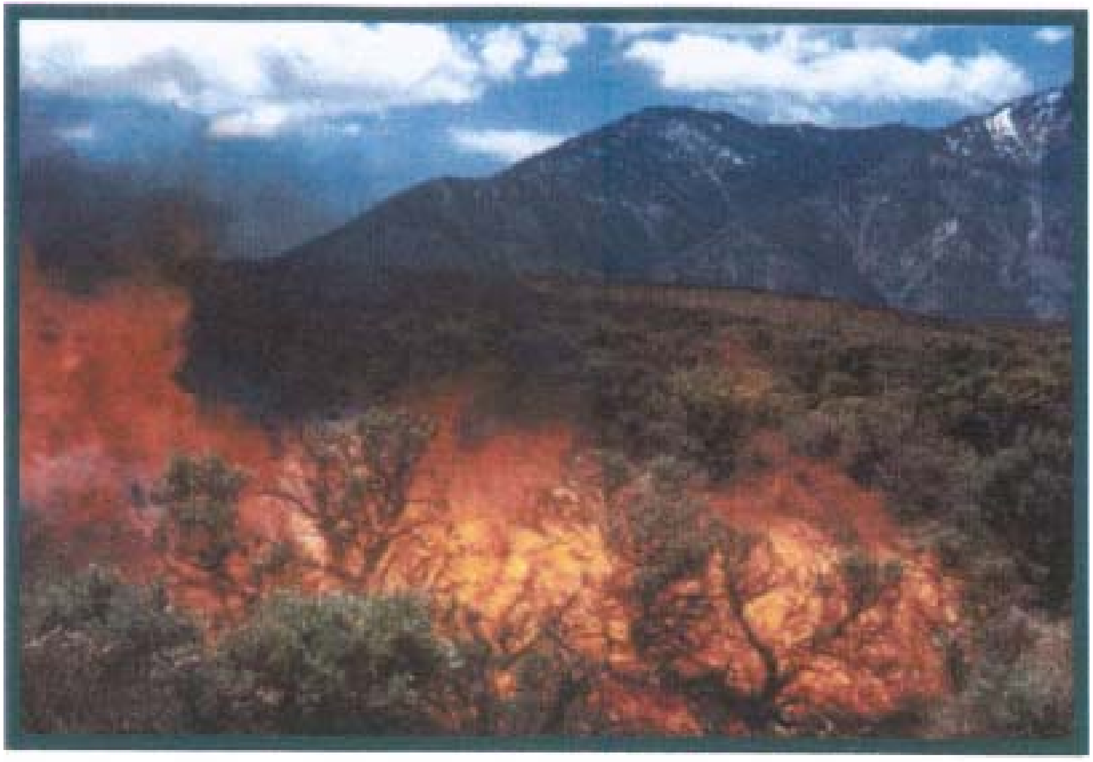

Figure 6-14—Prescribed burning to reduce blackbrush fuels at the urban wildland interface, Carson City, Nevada.

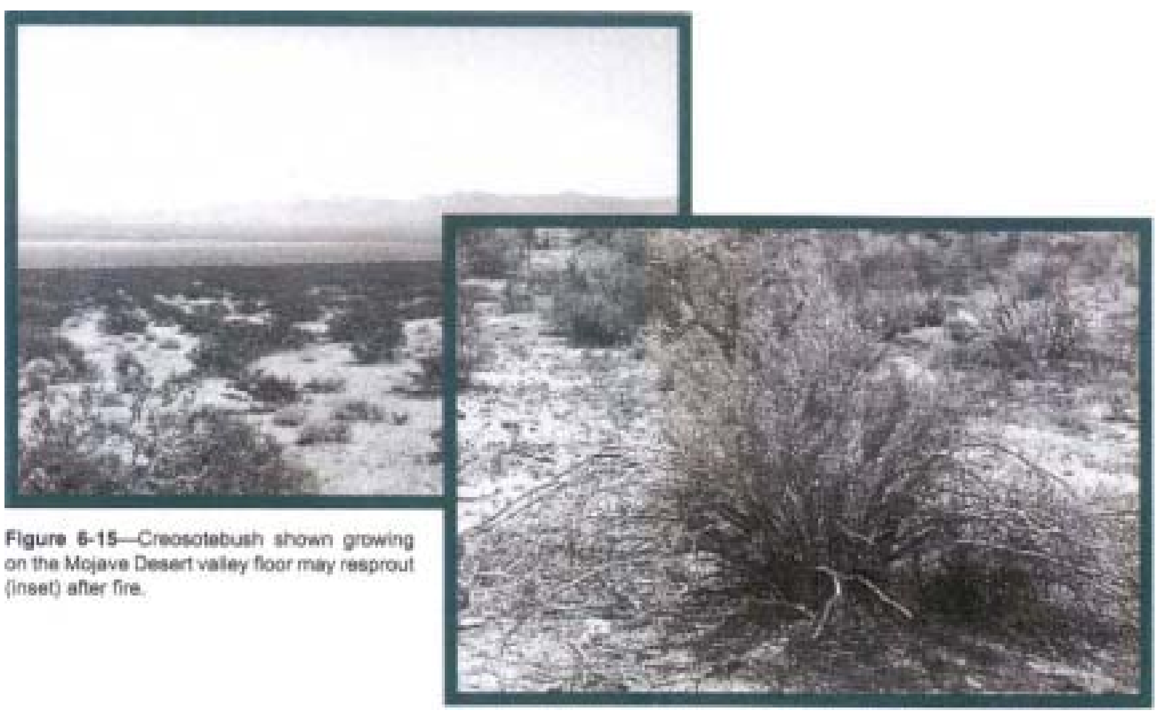

Figure 6-15—Creosotebush shown growing on the Mojave Desert valley floor may resprout (inset) after fire.

Figure 6-16-Joshua tree clones provide clusters of fuel in otherwise sparse desert shrublands, Mojave Desert, California.

breaks to reduce large-scale destruction of this unique resource by wildfires (fig. 6-17).

Creosotebush-Bursage-- This is a transitional plant association found below 5,250 feet (1,610 m) elevation. It merges with the paloverde-cactus shrub association found in the Sonoran Desert. In this region creosotebush-bursage has higher species diversity including a larger tree component (table 6-4).

Paloverde-Cactus Shrub --This type is characterized by open-to-dense stands of low-to-medium tall shrubs, small trees, cacti, and succulents (fig. 6-18). Paloverde, prickly pear, cholla, saguaro, and bursage are dominant species in this vegetation type. These communities area diverse mosaic of mixed vegetation that occur in the Sonoran Desert at elevations generally below 4,000 feet (1,200 m) (table 6-4).

Southwestern Shrubsteppe— This shrub type or the semi desert grass-shrub type (called desert grasslands in the FRES system) is composed of gently sloping desert plains found below the Rocky Mountains and between the low mountain ranges of the Sonoran Desert, Mexican Highland, and Sacramento section in Arizona, New Mexico, and Texas (fig. 6-19). Annual precipitation in this ecosystem varies from 10 inches (25 cm) in western areas to 18 inches (46 cm) to the east. Despite the fact that half of the rainfall occurs during warm months (frost free periods occur 180 days or more of the year), evapotranspiration is between 80 and 90 inches (203 to 229 cm) per year and may exceed the precipitation by a factor of 10.

Vegetation is composed of short grasses and shrubs of variable composition. Grasses inhabit the more developed Aridisol and Mollisol soils. Shrubs inhabit the shallow soils. Junipers occur exclusively on Entisols, which are predominantly found in the South.Yucca, mesquite, creosotebush, and tarbush are the dominant woody plants, while black grama, tobosa, and threeawn are the dominant herbaceous plants. Curlymesquite and other grama species also contribute significantly to the biomass of these shrubsteppe communities, which are used mainly as rangeland.

Two shrubsteppe typesarerecognized.The **Grama-otobosa shrubsteppe** occupies areas at elevations between 1,610 and 7,045 feet (488 to 2,135 m) and includes the more shrub dominated communities of the shrubsteppe. Black grama, sideoats, and tobosa are climax indicators occupying arid grassland communities throughout the Southwest. Black grama prefers more gravelly upland sites; sideoats is less selective, while tobosa prefers heavier clay lowland soils. The **Trans-Pecosshrub savanna** is found on

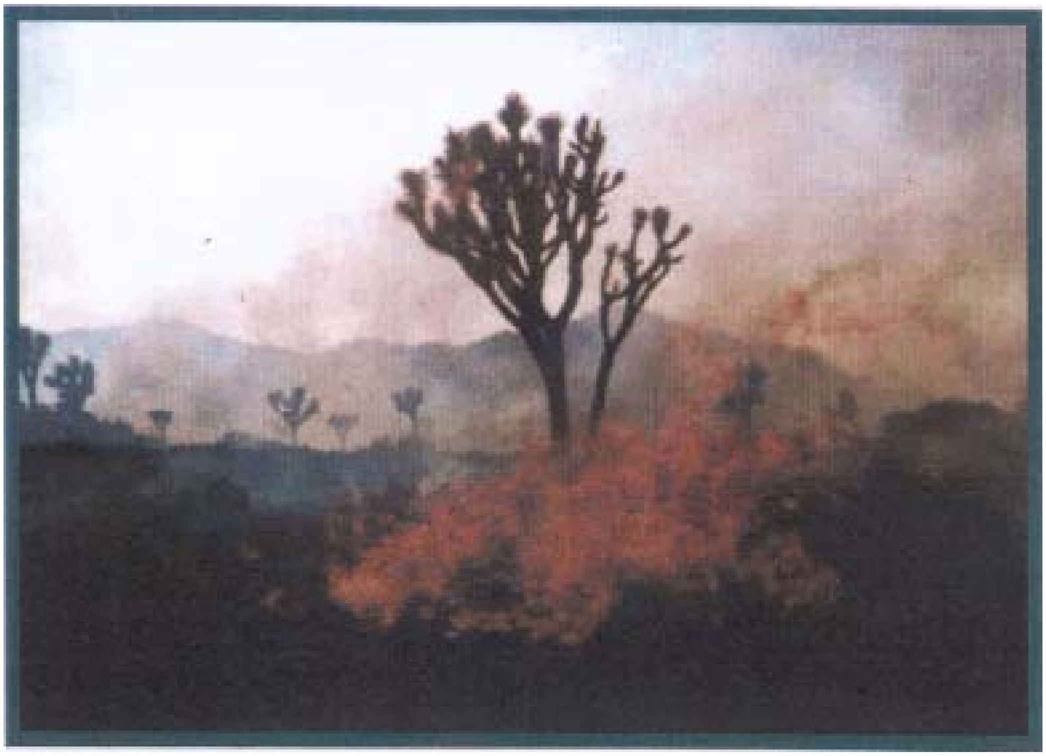

Figure 6-17--Prescribed burning in a Joshua tree forest to reduce fuel loading at the urban/wildland interface, Covington Flats, Joshua Tree National Park, California.

the Stockton Plateau and southwestern portion of Edwards Plateau. It has a higher average elevation (4,000 to 6,000 feet; 1,220 to 1829 m) and greater rainfall than the grama-tobosa shrubsteppe. This is a shrub dominated type characterized by grasses and the common occurrence of junipers (fig. 6-20). Junipers occupy more than 6 million acres (2.4 million ha) of rangeland in dense to open communities with oaks, Texas persimmon, and mesquite.

Chaparral--Mountain Shrub--This ecosystem type (fig. 6-19, 6-21) occupies lower and middle elevation mountain areas in the Pacific States, the Southwestern States, and the Rocky Mountains. The vegetation consists of dense to open shrubs or low trees with deciduous, semideciduous, and evergreen species represented. Some of the types are so dense that understory vegetation is practically eliminated, while other types support a highly productive understory.

Fire Reglme Characteristics

Fire frequency was variable in the stand-replacement fire regime types and depended upon ignition sources and plant community development. In the grassland types, fires could occur in any given year, provided the grass was cured and dry enough to burn. Although fire frequencies could not be measured precisely, mean fire intervals probably ranged from about 4 to 20 years depending on climate and ignition sources (Gruell and others 1985a). In the plains and grasslands, Native Americans ignited fires for a wide variety of cultural reasons. This was the predominant source of ignition in heavy use areas particularly at lower and middle elevations. But, a never-present ignition source was lightning, which was probably more important in valleys surrounded by forests than in plains grasslands due to differences in efficiency of lightning (Gruell and others1985b). Grasslands, occupying flat to gently rolling terrain, would burnover large areas until a break in terrain or a change in weather stopped the fires. Fires swept over extensive areas sometimes covering several hundred square miles.

Desert shrublands have been influenced over the last 12,000 years by climatic shifts, varying soils, and fire. Prior to Euro-American settlement, fires in these desert shrublands were set by lightning and Native Americans (Humphrey 1974; Komerek 1969). Wyoming big sagebrush experienced fire intervals ranging from 10 to 70 years (Vincent 1992; Young and Evans 1991). Arid land fire history studies report fire intervals between 5 and 100 years (Wright 1986). Griffiths (1910) and Leopold (1924) reported that before 1880

Table 6.4--Physiognomic and taxonomic descriptions of vegetation types modified from Kuchler (1964) showing habitat type[a] fuel, and forage associated with each. Note: Although numerous grass species are not listed for each vegetation type, they have become cosmopolitan throughout each type as a result of anthropogenic disturbance.Their impact on the fire dynamics of these desert ecosystems should be considered in making fire management decisions.

Vegetation. **-Fuels (Fu)** **-Forage (Fo)**	**Dominant species** ***Associated genera**	**Tree[b]**	**Shrub**	**Herb**	**Cactus**
Great Basin sagebrush[c]	*Artemisia tridentata*		S		
Dense to open low to medium	• *Artemisia, Atriplex, Chrysothamnus, Coleogyne*		S		
shrubs	• *Ephedra, Eriogonum, Tetradymia*		s		
Fu-0 to 2,000 lb/acre	• *Ashgalus, Lupinus, Phacelia*			H	
Fo-0 to 700 lb/acre	• *Agropyron*			G	
Blackbrush		S			
Dense to open broadleaf	• *Artemisia, Gutierezia, Haplopappus*		S		
evergreen shrubs	• *Ephedra*		s		
± herbaceous understory	• *Hilaria*			G	
Fo-250-500 lb/acre					
Saltbush/black greasewood	*Atriplex confertifolia/Sarcobatus vemiculatus*		S/s		
Open small shrubs	• *Lycium, Artemisia, Atriplex, Grayia, Krascheninnikovia*[d]		S		
Fu-250 to 750 lb/acre	• *Allenrolfea, Menodora, Suaeda*		ss		
Fo-50 to 200 lb/acre	• *Kochia*			H	
	• *Distichlis*			G	
Creosotebush	*Larrea divaricata*		S		
Open dwarf to medium shrubs	• *Yucca brevfolia*[e]	T			
Fu-40 to 100 lb/acre	• *Lycium, Baccharis*		S		
Fo-12 to 40 lb/acre	• *Encelia, Franseria, Sphaeralcea*		s		
Creosotebush/Bursage	*Larrea divaricata/Ambrosia dumosa*		S/s		
Open dwarf to medium shrubs	• *Cerci'dium, Dalea, Prosopis, Olneya*	T			
Fu-40 to 100 lb/acre	• *Lycium, Acacia, Fouquieria*		S		
Fo-12 to 40 lb/acre	• *Encelia, Franseria*		s		
	• *Hilaria*			G	
	• *Opuntia, Ferocactus*				c
Mesquite Bosques	*Prosopis giandulosa; P. velutina*	T			
Open to dense forest low	• *Cercidium, Olneya, Prosopis, Populus, Dalea, Salk*	T			
broadleaf deciduous trees	• *Acacia, Baccharis, Lycium*		S		
Fu-250 to 1000 lb/acre					
Fo-0 to 500 lb/acre					
Paloverde/Cactus Shrub	*Cercidium microphyllum/Opuntia* spp.	T			c
Open to dense low trees,	• *Cercidium, Olneya, Prosopis*	T			
shrubs, and succulents	• *Jatropha, Lama, Lycium, Simmondsia,*		S		
Fu-100 to 250 lb/acre	*Acacia, Condalia, Fouquiera, Celtis*				
Fo-30 to 100 lb/acre	• *Calliandra, Ephedra, Franseria, Janusia*		s		
	• *Camegiea*				c
	• *Ferocactus, Echinocereus, Opuntia*				c
Grama-tobosa shrubsteppe	*Hilaria* spp.. *Bouteloua* spp.			G	
short grass with shrubs	*Larrea*		S		
Fo-0-600 lb/acre	*Yucca* spp.		ss		
- - - - - - -					
Trans-pecos shrub savanna	*Juniperus* spp.	T			
shrubs with short grass	*Hilaria* spp., *Bouteloua* spp., *Muhlenbergia* spp.			G	
Fo-0-600 lb/acre					
- - - - - - - -					

[a]Based on Kuchler's classification system
[b]T=tree; S=shrub; s=subshrub; ss =succulent shrub; H=herbaceous; G=grass; c=cactus
[c]Great Basin sage is broken into four productivity classes (Garrison and others 1977)
[d]*Eurotia lanata (Pursh) Moq. =Krascheninnikovia lanata (Pursh) A. D. J. Meeuse & Smit*, (Jepson 1993)
[e]Yucca *brevifolia* (Joshua trees) become a significant tree component in parts of the Mojave Desert and grama-tobosa shrubsteppe

Figure 6-18--Mixed vegetation of the paloverde/cactus shrub in the Sonoran desert near Four Peaks, Maricopa County,Arizona.

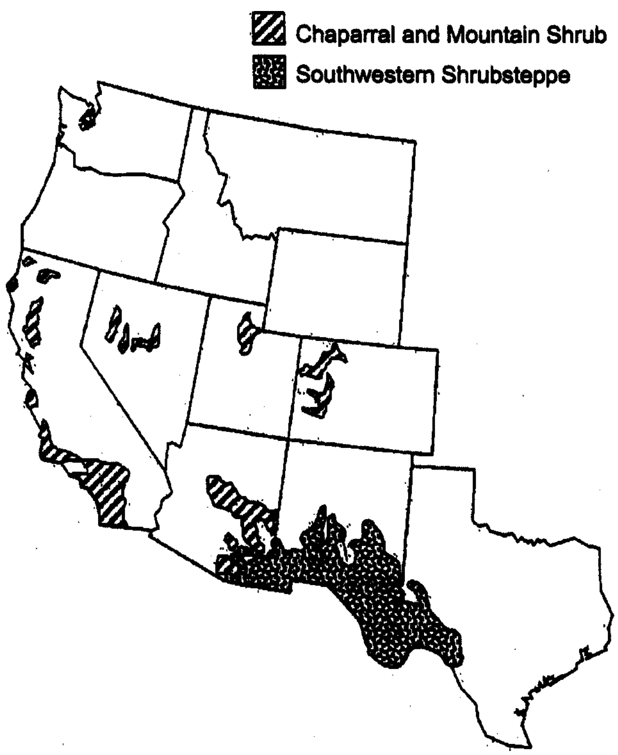

Figure 6-19--Distribution of Southwestern shrubsteppe and chaparral-mountain shrub FRES ecosystems.

Figure 6-20--Mixed fuels found in junier shrub savanna (New York Mountains,California).

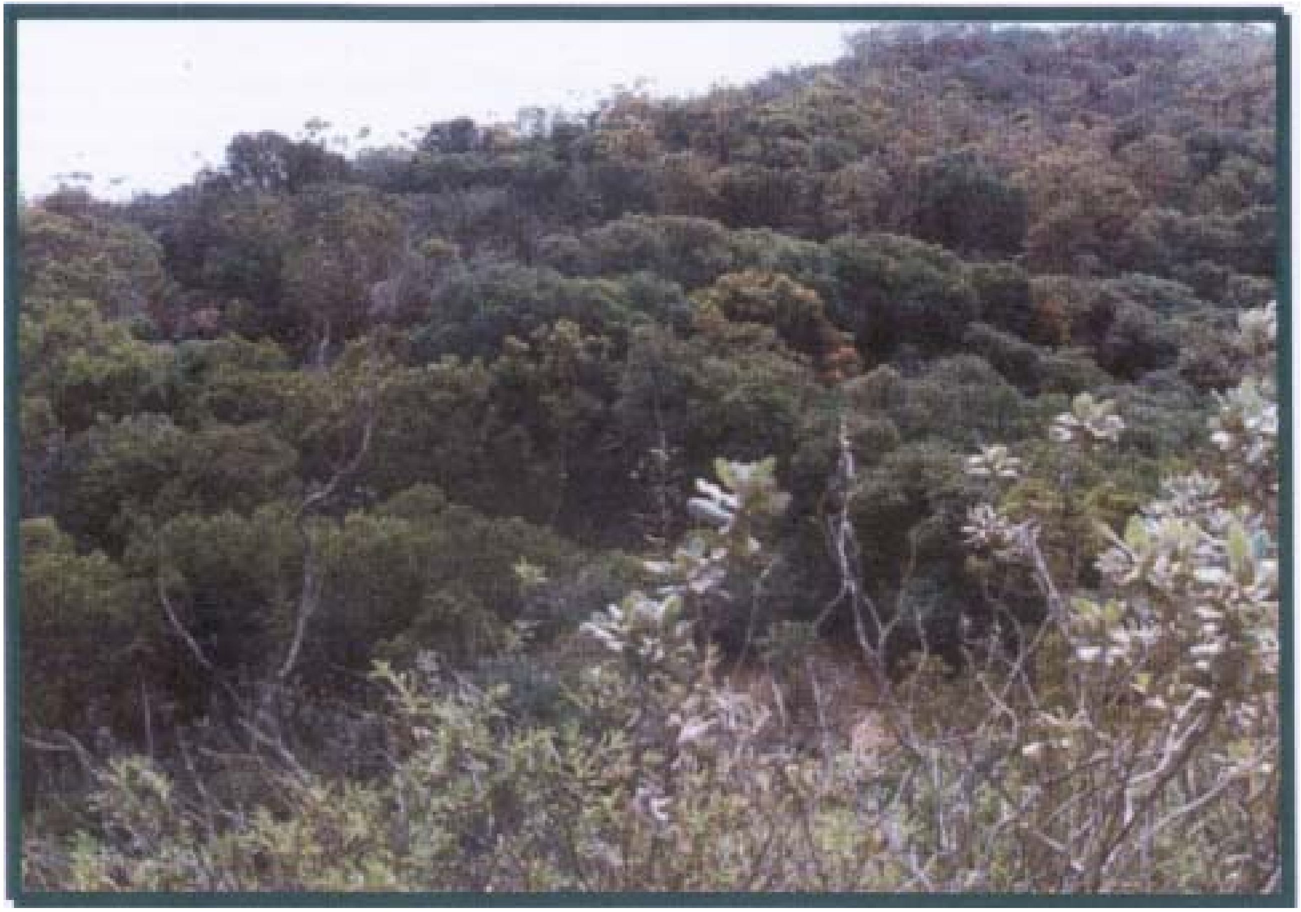

Figure 6-21-Typical chaparral vegetation (Arctostaphylos, Ceanothus), Mill Creek, San Benardino National Forest, California.

desert grasslands produced more grass and fires recurred at approximately 10-year intervals. Before settlement deserts were characterized by sparse vegetation, broken by barren soil, and were not expected to burn except under unusual circumstances. But when fire occurs in warm desert shrub habitat, along recovery is expected. This recovery depends on geographical location, species composition, and climatology after the burn. Recovery is more rapid in areas receiving higher precipitation. The various desert shrublands vary in wildfire risk ranging from nonexistent risk of the sparsely vegetated salt flats to high risk associated with heavy fuel loadings often found in the mesquite type. Postfire survival by desert plants may depend on genetic variation (Munda and Smith 1995), resprouting capability, resistant seeds, and delayed mortality.

In California chaparral, fire intervals for large fires (more than 5,000 acres) typically ranged from 20 to 40 years (Wright and Bailey 1982). But at higher elevations and north aspects fire return intervals were longer, perhaps as in frequent as 50 to 100 years. Young stands of chaparral whose canopy has not closed and stands that have not restocked well after disturbance often have a grass component that can burn on any given year, as is the case with the grasslands. These fires may or may not be stand-replacement fires, depending upon the amount of heat transferred from the grass component to the sparse shrub overstory. Fully developed chaparral stands can be difficult to ignite unless there is some component of dead material and good fuel continuity. However, given an ignition and some wind, they will propagate a moving fire even when virtually no dead material exists in them. Because these are crown fires, they are almost always stand-replacement fires. With both the grasslands and chaparral, all or most of the above ground portion of the plants are killed. Most of the perennial grasses have a perennating bud at or near ground level, often protected by bunched stems that act as insulators; often, tufts of these stems remain after fire. Chaparral shrubs are often killed down to the root collar; sometimes the entire individual is killed outright.

Fuels

Grassland Fuels -- -When cured and dry, grassland fuels are ideally suited for burning. For the most part, they fall into the fine fuel category; however, the compact arrangement of stems in the "tufts" of bunchgrasses makes these portions of the plant difficult to ignite regardless of their dryness. Once ignited, however, they can smolder forlong periods if enough old stem material has accumulated.

Plant density is also a critical factor in a grassland's ability to propagate fire. Heat output is relatively low from grassfuels, so fairly continuous fuels are necessary for fire spread to occur. Light winds can sometimes compensate for moderately sparse fuels by providing required flame bathing. The amount of fuel can vary with site condition, precipitation, and disturbance history. Typical annual productivity in desert grasslands can vary from next to nothing upwards to 1,000 lb/acre (1,120 kg/ha); in plains and mountain grasslands, productivity can be as high as 2,000 lb/acre (2,240 kg/ha) (table 6-5).

The character of a grassland fire is also affected by the overall geometry of the stand, which changes throughout the life cycle of the plants in the stand. The most dramatic example of this can be seen in annual grasslands where the plants germinate, seed, and die ina single season. A stand of recently cured annual grass can be quite dense and tall (up to 6 or 7 feet); its bulk density can be optimum for propagating a fast moving fire. In a relatively short period, a process of stand collapse begins and the bulk density of the stand becomes steadily modified. By the end of the season, the biomass is in a dense thatch on the ground and will begin decomposing--in some localities, fairly completely. Fire can still propagate during these later stages, as long as not too much moisture has accumulated in the thatch, but spread reates will not be as great.

Cheatgrass is a highly flammable fuel because of its finely divided plant structure, long period in a cured condition, rapid response to drying, and a tendency to accumulate litter (Bradley 1986a). Cheatgrass dries 4 to 6 weeks earlier than perennials and can be susceptiblet to fire 1 to 2 months longer in the fall. It produces large quantities of seed that usually develop into dense stands providing ideal fuel continuity for fast spreading fires. It grows well in areas of low precipitation that frequentlyundergo severe fire seasons.

Desert Shrublands-- Fuel include cacti and other succulents, grasses, shrubs, small trees, and mixtures of these. Fuels occur in discontinuous patches to areas wheretrees, shrubs, and grasses are contiguous. Fuel loadings may reach 2,000 lb/acre (2,240 kg/ha) (fig. 6-22, 6-23). Seetable 6-4 for fuel loading and forage production for each associated shrub community.

Table 6-5--- Fuel loadings (lb/acre) from FOFEM fuel models (Reinhardt and others 1997) for FRES grassland ecosystem types based on annual productivities.

Fuel class	Desert	Plains	Mountain
Sparse	300	600	900
Typical	600	1,250	1,900
Abundant	900	1,900	2,800

Fuel loading in **sagebrush** varies depending on the site and species. Based on shrub height and percent cover, big sagebrush varies from 0.26 to 4.6 tons/acre (0.55 to 10.2 t/ha). For a stand 2.5 feel in height and 20 percent cover, conditions typically found, sagebrush foliage and stem wood averages 1.5 tons/acre (Brown 1982). Herbage production for this vegetation type can vary from about 200 lb/acre (224 kg/ha) under poor growing conditions (Brown 1982) to 1 ton/acre (2.2 t/ha) under favorable conditions (Garrison and others 1977). Forage production generally is one-fifth of the annual herbage production. Humphrey (1974) noted that sagebrush was more subject to burning than any other desert type.

Dwarf sagebrush (14 habitat types) is usually relegated to shallow soil sand is not considered a fire management problem because fuel continuity is poor and it generally cannot carry fire.Tall sagebrush (29 habitat types) occurs on deeper soils, often has a substantial grass component, and burns readily (Blaisdell and others 1982). The presence of a herbaceous understory increases the potential for big sagebrush to carry a fire. Threetip, basin, Wyoming, and mountain big sagebrush occupy about 60 percent of the total sagebrush area. This sagebrush association is practical to burn (Blaisdell and others 1982). Techniques for managing sagebrush/grass ecosystems with fire and other means are discussed by Blaisdell and others (1982), Bushey and Kilgore (1984), McGee (1976, 1977), and Onsager (1987) (fig. 6-24). Fuel and fire behavior models were developed by Brown (1982), Frandsen (1981), Reinhardt and others (1997), and Tausch (1989) for burning in Great Basin sagebrush. Fire behavior studies in big sagebrush show that fire intensity and rate-of-spread can be two to three times greater when sagebrush foliage is cured, yet the proportion dead has little effect on predicted fire behavior (Brown 1982).

In **blackbrush** fuel production ranges from 0 to 500 lb/acre (0 to 560 kg/ha), and forage production ranges from 0 to 150 lb/acre (0 to 168 kg/ha). Blackbrush is negatively associated with fine fuels of litter and grasses. In **saltbush-greasewood** fuels production varies from year to year, depending on the amount of

Figure 6-22—During wet years, a herbaceous layer develops in the bare spaces between the dense thorn-shrub of the Sonoran desert, Maicopa County, Arizona, increasing the potential for major fires.

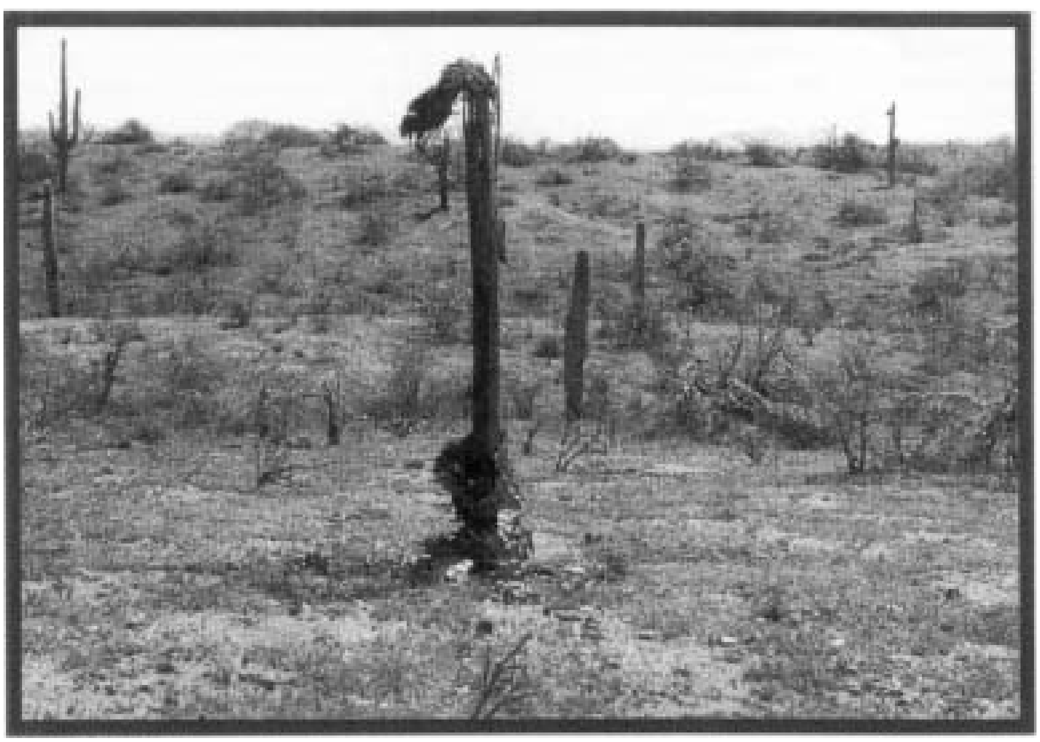

Figure 6-23—A wildfire burned 10,000 acres of this Sonoran desert thorn-shrub in Four Peaks,Tonto National Forest, Arizona.

Figure 6-24-- Fire is used as a range management tool for sagebrush found on the Great Basin plains.

precipitation. Production is also related to soil salinity and texture (West 1994). Herbage production is generally 0 to 500 lb/acre (0 to 560 kg/ha).

Creosotebush has low leaf to stem biomass,yet its standing dry biomass may reach about 3.8 ton/acre (8.5 t/ha) and produce about 892 lb/acre (1,000 kg/ha) perannum of new fuels (Chew and Chew 1965). The resinous foliage is flammable, but fire generally will not carry well in this community because the plants are usually surrounded by bare soil. Herbage production ranges from 40 to 100 lb/acre (44 to 112 kg/ha), about one-third of which is considered forage. High species diversity within the **creosotebush/bursage** shrub type produces diverse fuels. In some areas dense stands with herbaceous understory supply contiguous fuels for fire.

Mesquite bosques (fig. 6-25), characterized by low deciduous mesquite trees, are typically found in high moisture areas, and may produce up to 2,000 lb/acre kg/ha) (2,240 of herbage, particularly in areas that flood periodically and where the mesquite has been artificially reduced. Fuels are highly concentrated in mesquite bosques. Herbage production is commonly between 756 and 1,000 lb/acre (840 and 1,120 kg/ha) with forage production from 0 to 500 lb/acre (0 to 560 kg/ha) (Garrison and others 1977). Higher fuel loading on a site will increase the fire mortality of mesquite. Areas with 2.25 ton/acre (5.06 t/ha) of fine fuel sustain up to 25 percent mortality, but only 8 percent mortality for 1.1 ton/acre (2.47 t/ha) (Wright 1980). Dunes may form in association with mesquite thickets.

In **paloverde-cactus shrub** fuels production ranges from 100 to 250 lb/acre (112 to 280 kg/ha); about 35 percent of this vegetation has forage value. Fuels in the **Southwestern shrubsteppe** are mixed grass-shrublands. The Grama-tobosa region has a higher grass component while the Trans-Pecos shrub savanna has a higher shrub component. The variable fuels in the Trans-Pecos shrub savanna produce up to 450 lb/acre (505 kg/ha) of forage. Creosotebush and yucca are present, but grama and tobosa primarily contribute to the maximum 1,500 lb/acre (1,680 kg/ha) herbage production in this type.

Chaparral--Generally, fuels are not as easily ignited as grass fuels, but once ignited will burn readily if conditions are right. Plant density can vary with site, and sometimes with species. This is but one factor that affects fuel continuity in a stand. Another factor is the basic within-plant geometry that varies by species. Geometry and arrangement of the woody fuel portion and the leaves of chaparral plants are key to understanding the ability of chaparral stands to propagate fire. The woody fuel inside a given shrub varies in size class ranging from fine fuel (< 0.12 inch diameter)

Figure 6-25--Mesquite thickets form higly concentrated fuels in desert washes, Mojave Desert, California

to heavy fuel (3 or 4 inches and larger in the case of some manzanita species). The arrangement and dis tribution of these size classes within a shrub varies by species. Two extremes illustrate this: the arrangement of the woody portions of chamise and manzanita species (fig. 6-26). The smaller woody size classes are quite dominant in chamise and tend to be in proximity throughout the crown; the opposite is true for manzanita.The leaves of chamise are small and needlelike and are often relatively dense on a given twig.

Manzanita is a broad leaf shrub. The leaves of some species are relatively sparse---being held distant from each other by the woody structure of the shrub. Other species of manzanita have dense clusters of leaves--so dense that their thick sclerophyllous structures act like an insulator. Other chaparral species, some members of the *Ceanothus* genus for example, have only a moderate amount of fine woody material and have small broad leaves that are sparsely distributed throughout the shrub crown. In general, the geometry of chaparral shrubs is not well suited to the spread of fire. Chamise is an exception, especially in dense stands with overlapping crowns.The maintenance of crown fires in chaparral almost always requires dry, windy conditions, which commonly occur in this vegetation type.

With few exceptions, fully developed stands of chaparral have no understory layer of vegetation, and therefore no potential for the "ladder effect" to propagate fire. However, when a litter layer exists, which occurs under gentle slope conditions, it can significantly aid fire spread under marginal burning conditions. In this situation, fuel moisture content becomes an important factor.

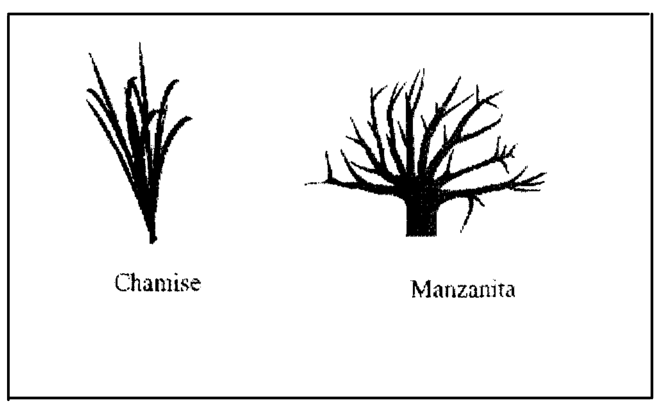

Figure 6-26--Arrangement, distribution, and size of woody fuels canvary by species.

The dynamics of dead fuel production in chaparral remain a mystery. Some suggest that dead fuel production increases with stand age (Rothermel and Philpot 1973). While this is undoubtedly true, age is not the only factor involved (Paysen and Cohen 1990). Complexities of onsite growing conditions and periodic events seem to be important. For example, the authors suspect that an unusual drought can produce fine dead material in chaparral stands that may be only present on site for a year or so — making the assessment of dead fuel dynamics unclear. Considerable down and dead material can be found in old chaparral stands. The concept of "old," unfortunately, has to remain a relative one for now. The age at which significant amounts of dead material are produced in a given stand of a given species composition cannot be predicted yet.

Postfire Plant Communities

Plains Grasslands

Pre-1900 Succession--The literature on plains grasslands communities is rife with contradictory interpretations of grassland dynamics. A few facts seem to be agreed upon. First of all, pollen records and rat middens indicate that most of the Central Plains was covered with boreal forest dominated by spruce, while much of the Northern Plains was glaciated during the Pleistocene. There are indications that the Southern Plains and the arid grasslands of the Southwest were also dominated by various conifer and broad leaf trees. The climate change that brought about the end of the glacial period ushered in the retreat of the boreal forest and its replacement by grasslands--a kind of vegetation able to cope with the drier climate and soil conditions that predominated.

Fire was not a predominant force in delimiting the extent of the plains grasslands. But given their existence and their flammability characteristics, the presence of fire had to have an impact on the character of the grasslands, their species composition, and the distribution of dominance. Modifications of climate and soil development led to invasion of some grassland areas by woody species. Under these circumstances, fire probably had a distinct role to play in the maintenance, or loss, of these grassland areas. Working in concert with grazing animals, fire could check the advance of more fire-sensitive, woody species, providing enough grass fuel was available. It could also encourage the advance of woody species that were adapted to disturbance and harsh climate conditions. Where invasion by woody species was not an issue, fires could maintain a highly productive mode in some grasslands, and in others cause shifts in grassland species composition; under conditions of drought,it could result in severe site damage.

Clearly, fire was a common element in presettlement times, and there is some conjecture that its frequency might have increased with the arrival of Euro-American settlers (Jackson 1965). For years, attempts to suppress fires in the plains were either nonexistent or not effective. As late as the 1890s, from the Dakotas to the Texas Panhandle, fires would run unchecked for days. During this period, fire, drought, and grazing played a role in maintaining, and at times debilitating, the grassland character. When fire, or any other phenomenon that reduced the vegetative cover, occurred during periods of serious drought, wind erosion often retarded the processes of succession.

Post-1900 Succession--The general set of natural forces affecting succession just prior to 1900 has not really changed in principle. Land use has alternately intensified, and disappeared, and returned again in some cases. Some of the plains grasslands have been converted to agricultural use — producing corn, wheat, barley, and various legumes; some have been put to intensive grazing use — successfully in some instances, and in others with disastrous results. In the Southern Plains, the conjunction of inappropriate farming practices and a devastating drought in the 1930s brought about a perceived ecological disaster and social phenomenon, called the "dustbowl," that shook the fabric of Southwestern culture. In retrospect, no surprises should have existed.

The semiarid climate of the plains grassland area, the ever-present potential for drought, yearly temperature extremes, and the potential for high winds exist today, as they have existed for centuries. They were operative in forming the plains grasslands and continue to drive the processes of succession. The factors relevant today are the firmly entrenched agricultural practices and the use of the grasslands as pasturage for grazing animals. Land use patterns such as these, once terminated, will drive the processes of succession in various directions-dominated by the presence of the existing natural factors. Deviations from successional patterns of past centuries are difficult to predict other than on a case-by-case basis.

Management Considerations-- Management of plains grasslands should be undertaken with a view toward maintaining stability under local climate and soil conditions. In the Northern Plains, a temperature range of more than 130° F between yearly maximum and minimum temperatures can occur (a range of 174 °F has been recorded in one place).The average growing season can range from 116 days in the northern most portion to 160 days in the southern part (Rogler and Hurt 1948). Native grasses tend to be hardy and drought resistant — such species as blue grama, buffalograss, western wheatgrass, and needlegrass. If the native grasses are to be used as livestock forage, then over utilization should be guarded against. Native range utilization by livestock should be supplemented by locally produced forage and seed crops whenever needed to protect native species.

The Southern Plains are also characterized by temperature extremes and a highly variable climate. Precipitation is comparatively light and infrequent; a major proportion of it falls during the active growing season, from April through September (Savage and Costello 1948). Humidity is low, winds are high, and evaporation is rapid. Hot temperatures and high winds often reduce the effectiveness of precipitation that does occur. Over utilization of rangelands during drought always has to be guarded against.

Fire can be either a disaster or a useful element in the plains grasslands, depending on its timing and severity. Arange fire that denudes a large area preceding a drought can set the stage for severe soil movement in many areas of the Great Plains—the high winds and frequently aolean soils indicating the process. When good recovery is favored by adequate precipitation, fire can improve productivity for a while. The effectiveness of fire, both good and bad, can be mitigated by current levels of productivity and by intensity of utilization. Recently grazed grassland, or a year of low productivity, can reduce the impact of fire by minimizing fuel consumption, fireline intensity, and general extent of burning.

The use of fire as a management tool can improve productivity if it is applied in a manner consistentwith the grassland's productivity, given climate and soil character. Kucera (1981) contrasted the application of prescribed fire between the more moist, highly productive grasslands and those of lower moisture availability and less productivity (fig. 6-27). Timing of the application centers on the development of thatch. In the higher productivity grasslands, the buildup of thatch tends to suppress productivity after a few years. In the lower productivity grasslands, the development of thatch provides a means of storing moisture and thus increases productivity--at least over a period of a few years. Thus, relatively frequent application of prescribed fire in the high productivity grasslands can be beneficial by removing thatch that has accumulated beyond desirable levels.

Mountain Grasslands

Pre-1900 Succession--Although bunchgrass species vary in their individual susceptibility to fire damage, repeated fires at intervals of about 5 to 40 years (Gruell and others 1986) maintained the bunchgrass community. The abundance of individual species no doubt varied not only by site conditions but by the actual frequency and seasonal timing of fire. A successional process of major importance was the continual

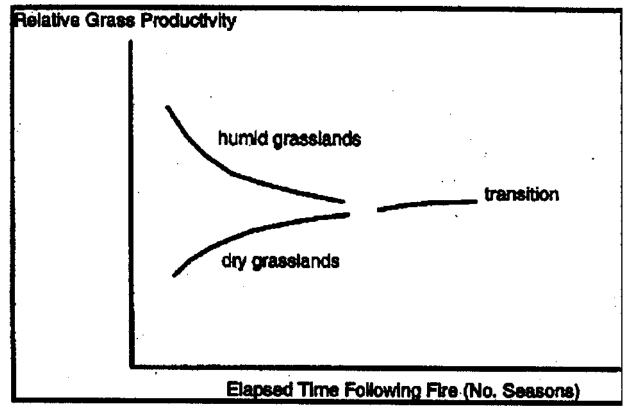

Figure 6-27--The productivity of humid grasslands versus dry grasslands after fire.

checking and reduction of woody plant encroachment. Mountain grasslands were intertwined with forests and shrublands ranging from rose and aspen in Alberta toconifersand sagebrush of Rocky Mountain foothills. Encroachment into grasslands by woody species was an ongoing process kept in check by repeated fires.

Post-1900 Succession-- Grazing by livestock, elimination of Native American ignitions, and fire control efforts greatly reduced the amount of fire in these grasslands. As a result tree species such as ponderosa pine, Douglas-fir, and lodgepole pine, and sagebrush have increased substantially along ecotonal boundaries. In some areas dense Douglas-fir forests now dominate sites to such an extent that evidence of former grasslands is lost except by soil analysis (Bakeman and Nimlos 1985). Elimination of periodic burning has apparently reduced diversity of herbaceous species in some areas (Wright and Bailey 1982).

In a study of fire regimes in the Interior Columbia River Basin involving grasslands and other vegetation types, Morgan and others (1994) suggested that human influences have had a variable effect on the nature of fire regimes. Fires tended to be less frequent but not always more severe. For example, where exotic annuals have invaded sagebrush steppe vegetation, fires have been so frequent that sagebrush does not have time to reestablish, and the annuals return quickly. Changes in fire regimes can move in one direction as a result of active fire suppression that results in a buildup of fuel, or in another direction as a result of livestock grazing and other activities that break up fuel continuity. No single successional formula can be offered for grasslands in general.

Management Considerations--Prescribed fire can be effectively used to hold back woody plant encroachment and maintain high levels of productivity in mountain grasslands. The complexity of mountain grasslands, however, requires careful consideration of species composition and site dryness to design prescriptions for successful prescribed fire (Wright and Bailey 1982). For this, knowledge of species response can be helpful.

Idaho fescue is sensitive to fire partly because it is susceptible to smoldering in the clump that can kill plants or reduce basal area. It tends to recover slowly from fire; however, on some sites it can withstand burning (Bradley 1986b). Burning when soils are moist, such as in the spring, helps to minimize damage. Needlegrasses can also be severely damaged depending on severity of fire. Damage from wildfires can be minimized by the grazing of livestock to reduce fuels. Needle-and-threadgrass reproduces by seed and can increase markedly in 2 to 4 years after fire (Gruell and others 1986). Bluebunch wheatgrass and Sandberg bluegrass recover quickly from fire (Bradley 1986c; Howard 1997). Rough fescue generally responds favorably to fire even after an initial reduction in basal area. Preburn coverages can be attained in 2 to 3 years (McMurray 1987).

Cheatgrass

Succession--Cheatgrass was accidentally introduced into the United States sometime around the turn of the 20th Century, supposedly through contaminated grain (Pyke and Novak 1994). Cheatgrass did not emerge as a noteworthy element in the Great Basin environment until the period between1907 to 1930 (Morrow and Stahiman 1984). By 1930, it had achieved its current distribution (Pyke and Novak 1994). In the early 1900s it had been noted in isolated places—notably embankments, railroads, and highways. During the next 3 decade, it spread rapidly into overgrazed sagebrush rangeland (Billings 1994).

Following disturbance by fire in areas where cheatgrass is present, it reestablishes from abundant seed.Even if fire destroys 90 percent or more of its seed, it can reestablish and compete significantly with native perennials (Bradley 1986; Monsen 1992). Over a period of years, cheatgrass gains dominance over perennials and increases the flammability of the site (Peters and Bunting 1994). Repeated fire will diminish the perennial seedbank and allow cheatgrass to increase its dominance. Once cheatgrass becomes abundant enough to increase the likelihood of fire, repeated fires may occur frequently enough to eliminate shrubs such as sagebrush and native perennials. As wildfires become more common cheatgrass can essentially dominate a site (Monsen 1994).

Management Considerations--Natives pecies can occupy sites that were dominated by cheatgrass, but this is not a common occurrence. Use of mechanical tillage, herbicides, and properly timed fire can be effective in reducing cheatgrass cover if other species that germinate under cool conditions can be introduced. Prompt rehabilitation of burned areas by seeding accompanied by livestock restrictions is important. Fire usually gives cheatgrass a competitive advantage. However, prescribed fire can be used to reduce cheatgrass and to allow seeded species a chance to establish. The narrow prescription window during which substantial seed can be destroyed is from the time cheatgrass becomes flammable, when it leaves the purple stage, until seed falls a short time later.

During the 1990s a green stripping program gained favor. The objective was to reduce wildfire frequency and size by establishing strips of fire-resistant vegetation, such as forage kochia, at strategic locations on the landscape to slow or stop wildfires (Pellant 1994). Greenstripping is aimed at effectively disrupting fuel continuity, reducing fuel accumulations and volatility on areas with a high density shrub cover such as sagebrush, and increasing the density of plants that retain higher moisture contents.

Annual Grasslands

Pre-1900 Succession--In California where this type prevails, the Spanish settlers kept poor records, so knowledge of native vegetation types is poor. Many believe that the prehistoric vegetation was perennial (Garrison and others 1977), but meager evidence is available to support this belief. However, evidence from the early 1800s indicates dominance by annual grasses.

Post-1900 Succession--Intensive agricultural development has taken over much of the original annual grasslands. At the lower elevations of the ecosystem, cultivated lands make up one of the richest agricultural areas in the world (Garrison and others 1977). Remnants lie at upper elevations in the Sierra foothills, and many are components of a hardwood savanna or shrub savanna that are quite common in these foothills. The annual grasslands are quite responsive to rainfall, and productivity and species dominance both vary accordingly. Fire is very much apart of the ecosystem and does not seem to have detrimental effects. In fact, it is being used by ranchers to eliminate woody overstory species and enhance productivity of the grasses.

Management Considerations--The most productive portions of the ecosystem are not producing annual grasslands, but rather agricultural crops. Clearly, as long as this activity can be sustained, it will remain the primary management activity in the "bottom land" portions of this system. In the upland portions, grazing and fire can be achieved to attain various management goals. However, they are both system disturbances and must be used judiciously. Annual rainfall is probably the most important consideration in applying management treatments in a manner consistent with ecosystem viability. Drought years should probably not be accompanied by intensive disturbance activities.

Desert Shrublands

During the era of Euro-American settlement, fire frequencies initially increased. Newspaper records between 1859 and 1890 report that settlers engaged in active fire suppression, including deliberate overgrazing of rangeland to reduce fuels. Woody species were favored by the reduction of grass and forb competition caused by overgrazing (Wright 1986). Grazing altered the role of fire in those desert areas once dominated by grasses. The consequent reduction of major fires was followed by shrub invasion into desert grasslands (Bahre 1985). Early 1900s wildland management policies continued to promote historical fire suppression and rangeland use in desertland scapes. A new management strategy was initiated when desert managers recognized that continued shrub encroachment was associated with overgrazing and fire reduction (Komerek 1969; Leopold 1924). Shifts in land management resulted in reduced grazing, increased fuels and, thus, changed the fire dynamics. Currently, burning of thousands of acres is becoming more common, and fire has become a serious management issue in some shrubland areas (Blaisdell and others 1982; Bunting and others 1987; Narog and others 1995; Schmid and Rogers 1988; Wilson and others 1995a).

Desert shrubland management traditionally focused on shrub eradication in favor of grasses. The objective was to improve forage for livestock and increase efficient management of range by increasing livestock and wildlife visibility. Fire, disking, herbicides, and heavy grazing were all commonly used. Often, the end result of this heavy range management was to decrease the amount of annual biomass and actually reduce the productivity of these ranges.

The use of fire in desert shrublands is controversial. Experts do not agree on historical fire cycles or what the land-use goal must be. Presently, desert range managementpractices rely on generalized studies made on limited areas. Anthropogenic influence has changed the vegetation and its dynamics in these dry sensitive areas. High fuel loading, from multiple branching shrubs, and contiguous herbaceous fuels are now common in many of these deserts. Fire can be used to achieve desired objectives in many of these desert shrubland communities (Bunting and others 1987; Lotan and others 1981; McGee 1977; Wright 1990). Fire also may contribute to the loss of desirable fire intolerant species that are sometimes replaced by less desirable fire tolerant species. The present resource management challenge is to determine which species to maintain and what management priorities are suitable for each specific area.

Sagebrush

Pre-1900 Succession---Historical accounts of sagebrush habitat are sketchy, but fires in big sagebrush were set both by lightning and humans. The many species and subspecies of sagebrush are quite susceptible to fire. Typical succession after fire would begin with a grass/forb dominance, and eventually lead to sagebrush recovery in 30 or more years.

Until the mid-1800s, the American bison was the primary herbivore impacting the fuels of sagebrush/grasslands (Young and others 1979). In the late 1800s, overstocked free ranging cattle led to a depletion of perennial grasses and other palatable forage. The subsequent introduction and spread of cheatgrass in the early 1900s corresponds with increased fire frequency and the reduction of big sagebrush. This, in turn, increased erosion and further damaged perennial native grassand forb components (MacMahon 1992).

Post-1900 Succession-- Since 1900 the cultivation and abandonment of marginal land, abusive grazing, and widespread recurrent prescribed burning of sagebrush resulted in an imbalance between the numbers and sizes of shrubs, and associated native grasses and forbs (Blaisdell and others 1982). Thus, much of the resource potential of the sagebrush range was depleted. By 1936, 85 percent of sagebrush lands were considered depleted (Tisdale and others 1969). Prescribed fire was used to remove shrubs and replace them with native perennial grassforage (Cornelius and Tablot 1955; Pechanec and others 1954; Pechanec and Stewart 1944; Reynolds and others1968). This ecosystem readily burns, particularly where there is a contiguous understory of grasses. Habitat changes coincident with increased fire have included plant community composition changes (Blaisdell 1949; Hassan and West 1986), altered soil seed banks (Blank and others 1995), and increased soil repellency (Salih and others 1973). The absence of sagebrush is often an indicator of past burns (Humphrey 1974). Secondary consequences of wildfires in sagebrush can include range deterioration, flooding, erosion, lowered grazing capacity, and reductions in the amount and quality of wildlife habitat. Extensive research has focused on rangeland degradation (Young and others 1979) and loss of productivity (Beetle 1960; Harniss and others 1981).

Management Considerations-- Sagebrush land managers are now confronted with recovering its productivity. Sagebrush production loss continues even with recent improvements in management. Currently, the value of the sagebrush rangelands is being reevaluated. Multiple factors need to be incorporated into resource management plans. Big sagebrush can gain dominance over the herbaceous layer in 5 to 30 years after a burn. Season of burn modifies species dominance (White and Currie 1983) and affects postfire sagebrush productivity (Mueggler and Blaisdell 1958). For example, silver sagebrush mortality is higher and regrow this less after a dry fall burn (White and Currie 1983). After fires, sagebrush mortality is proportional to fuel reduction. Although many sagebrush species are readily killed by fire, at least three species (threetip sagebrush, silver sagebrush, and California sagebrush) are known to resprout (Malanson and O'Leary 1985; Tisdale and Hironaka 1981). Most sagebrush species reseed after fire, but may require fire intervals of up to 50 years to regain their dominance (Bunting and others 1987). Frequent fires can cause type conversion from sagebrush species to rabbitbrush, horsebrush, and snakeweed. Where wheatgrass occurs, the burn season is extended and wildfires are reported to consume more area per burn.

Introduced cheatgrass can outcompete indigenous herbaceous species. This brome is undependable forage because of its large fluctuations in yield from year to year. After two to three reburns, sagebrush sites can be converted to stable cheatgrass; fire return intervals of 5.5 years maintain cheatgrass dominance. Cheatgrass is often accompanied by other invasive, noxious, and undesirable species. Together these pose a serious fire hazard, particularly following wet springs.

Planning prescribed fires in sagebrush should include specific objectives and consider many factors such as species and subspecies of sagebrush, soils (Salih and others 1973; Simanton and others 1990), fuelloading, fuel moisture content, and wind speed (Britton and Ralphs 1979; Brown 1982). Early spring or late summer burns can be used to promote native perennial grasses. There is little postfire recruitment for 3 to 5 years following a fire in perennial grasses, yet surviving grasses and accompanying forbs increase biomass production. Often forbs will dominatean area for several years postburn. Harniss and Murray (1973) found increases in herbage production for 20 years after a burn.

Attempts at restoring sagebrush rangeland to achieve higher biomass yields are being investigated (Downs and others 1995). In general, shrublands that have been converted to grasses by large wildfires are difficult to restore. Fire negatively impacts soil seedbeds important forsage brush regeneration (Blankand others 1995). Sagebrush seed can be viable up to 4 years. Sagebrush can be restored through reseeding. Cheatgrass seedbanks present on sagebrush sites may negatively influence reestablishment of native bunchgrasses and shrubs (Hassan and West 1986). If sagebrush is in good "natural" condition an initial postfire influx of cheatgrass will occur. Given adequate precipitation, perennial native grasses and shrubs can out compete cheatgrass by the second year (West and Hassan 1985). Postfire rehabilitation efforts can be unsuccessful if other measures such as grazing are not incorporated (Evans and Young 1978). Species and associations of the sagebrush-grass type are influenced by edaphics and microclimate (Meyer 1994). Restoration efforts are complicated by the level of site disturbance and ecosystem variability and specificity (Blaisdell and others 1982; Blank and others 1995). Wildfire in cheatgreass dominated sites may afford managers an opportunity to reseed with perennial grasses and reduce the cheatgrass to lengthen the fire return interval. Presence of woody fuels may provide a hotter fire that can kill more cheatgrass seeds. Herbicide applications may facilitate native shrub and grass reestablishment (Downs and others 1995).

Wildlife such as pronghorn, deer, elk, coyotes, rabbits, rodents, and an endangered prairie dog reside in sagebrush rangelands. Abundant avifauna (over 50 species) that nest and feed in sagebrush include eagles, hawks, owls, doves, chukar, and sage grouse. Wild ungulates and domestic sheep may benefit from the maintenance of high quality sagebrush browse (Rodriguez and Welch 1989). Wildfires have removed large areas of sagebrush and may have destroyed a significant amount of sage grouse habitat (Downs and others 1985). Short-and long-term effects of fire on wildlife in this habitat need further evaluation (Gates and Eng 1984).

Blackbrush

Succession--Historical documentation of blackbrush fire cycles is limited. As late as 1981 (Lotan and others 1981; Martin 1975), land managers did not perceive desert fires as a serious landmanagement problem because of small fire size and minimal damage to resources. Current data refute this perspective (Narog and others 1995; Wilson and others 1995a, 1995b). Cyclic desert precipitation above 10 to 14 inches (25 to 36 cm) may increase biomass and fuel continuity enough to increase fire behavior potential.

Since 1900, it appears that neither fire nor exotic annuals have altered soil microflora apparently required for blackbrush survival or reestablishment. However, burning has promoted succession to grassland by destroying the cryptogamic crust that stabilized the soil. Frequent large fires have eliminated blackbrush from some areas. Some sites show no recovery after almost 4 decades (Wright and Bailey 1982). Currently, burning is not a recommended practice for range enhancement purposes in this shrub type (Ballison and others 1985) because blackbrush is often replaced by species of similar forage potential.

Management Consideration--Fire has been used for range improvement by reducing the shrub to grass ratio in areas where shrubs are gaining dominance. Land managers must also focus on protecting cacti and succulents, which will complicate fire management because of their various responses to fire (Thomas 1991). Fire may continue to be a necessary tool to modify fuel buildup. Currently, increases in desert shrubland fires and fire size have become a serious concern particularly with the recent increase in urban encroachment and resource degradation issues on these lands.

Research is needed to develop management and restoration recommendations for blackbrush (Pendleton and others 1995). Fire destroys the short-lived blackbrush seedbanks (produced by masting) necessary for it to reestablish. High temperatures, wind, and low humidity are usually required to propagate fire in blackbrush. If blackbrush becomes decadent or in some way presents a wild fire hazard, removal by burning may be appropriate. In some cases mature shrubs may survive low intensity fires; however, fire generally kills both seeds and mature shrubs. Although blackbrush is somewhat effective for erosion control, it may take more than 60 years to reestablish after a disturbance such as fire (Bowns and West 1976).

Wildlife such as deer, elk, desert bighorn, pronghorn, squirrels, rabbits, and game and nongame birds use blackbrush for cover, browse, and seeds. Livestock are more limited: sheep and goats browse blackbrush, but its low palatability and nutritional value make it unsuitable for cattle and horses.

Saltbush-Greasewood

Succession--Little is written regarding historic fire patterns in the saltbush-greasewood type. In some areas little change has occurred since 1900 in black greasewood dominated vegetation, while in others both saltbush and black greasewood have expanded into areas previously dominated by sagebrush (Sparks and others 1990). Rangeland seeding and invasion of grasses forming a highly flammable understory have increased the fire frequency in the saltbush-greasewood type. Postfire recovery is often rapid due to postfire resprouting and vigorous reseeding strategies used by the various shrub species in this vegetation type.

Management Considerations--In the past fire management was not a concern in saltbush-greasewood vegetation because sparse understory, bare soil frequently found in intershrub spaces, and the low volatilization of many saltbush species made this vegetation type resistant to fire (Tirmenstein 1986). These communities may burn only during high fire hazard conditions. In wet years brought by El Nino, such as 1983 to 1985, fine fuels may become contiguous across otherwise gravelly soils. Recently these fine fuels have become a fire hazard problem (West 1994). Grazing and other disturbance can encourage increases in biomass production, especially in the spring (Sanderson and Stutz 1994). Introduced cheatgrass has increased the fire risk, particularly when the area is ungrazed (West 1994). Disturbance may also allow this vegetation type to increase its range. Many species in this type resprout (West 1994). Black greasewood vigorously resprouts after fire or other disturbance. Season of burn, fire intensity, and fuel loading may be important factors to consider when using fire to regenerate or increase the productivity of this veg etation type (Harper and others 1990). Intense fall fires may increase plant mortality in spite of a species' resprouting potential.Some A*triplex* species resprout and others produce abundant seeds. Thus postfire reestablishment from onsite and offsite seed sources is possible.

Saltbush-greasewood vegetation provides valuable forage for livestock and wildlife, particularly during spring and summer before the hardening of spiny twigs. It supplies browse, seeds, and cover for birds, small mammals, rabbits, deer, and pronghorn. Saltbush and black greasewood can be used to revegetate mine spoils and stabilize soils. Saltbush concentrates salts in leaf tssue and may be used to reduce soil salts can reclaim degraded land for agriculture. Outplanting methods are being developed for saltbush restoration projects (Watson and others 1995).

Creosotebush

Succession--Historically, creosotebush was restricted to well-drained knolls and foothills. However, by 1858 it had begun to invade the grama grasslands and by the early 1900s creosotebush had encroached into areas dominated by grasslands (Valentine and Gerard 1968). Overgrazing and drought contributed to the expansion of creosotebush range (Buffington and Herbel1 965). Fire suppression may be contributing to this expansion.

Management Considerations-- Creosotebush invades desert grasslands. Although creosotebush may suffer up to 80 percent dieback during drought, it still resprouts (Humphrey 1974). On the other nad, it is sensitive to fire, especially in spring (Brown and Minnich 1986; McLaughlin and Bowers1982). Fire and herbicides have been used to control creosotebush. High fuel loading and spring and summer burning will lead to higher creosotebush mortality from fire (Martin 1966). This indicates that wildfires could have kept it from invading grasslands before

Euro-American settlement (Wright and Bailey 1982). Selective thinning of creosotebush by fire suggests that this ecosystem is not resilient to burning and creosotebush may be replaced by other species, particularly with recurrent fires (Cable 1973). For example, bush muhly growing under creosotebush canopies may out-compete smaller shrubs and become the dominant after fire. Following heavy precipitation, herbaceous fuel increases and may increase fire potential in the creosotebush vegetation type (Brown and Minnich 1986). Creosotebush can withstand some fire exposure (O'Leary and Minnich 1981). Brown and Minnich (1986) reports low recovery for creosotebush after low-severity fire, and limited sprouting and germination were observed after fire in most of the species in the creosotebush associations.

Sheep will use creosotebush for cover, but creosotebush is unpalatable browse for livestock and most wildlife. However, pronghorn, bighorn sheep, mountaing oats, game and nongame birds, fox, small mammals, and many reptiles and amphibians are some of the wildlife that use creosotebush for cover and its seed for food. Interestingly, the protected desert tortoise *(Gopherus ugussizi)* typically burrows in soil stabilized by this plant (Baxter 1988). Creosotebush can be outplanted to facilitate rehabilitation of disturbed desert areas where it improves microsites for other plants and for fauna.

Creosotebush-Bursage

Fire use prior to 1900 may have limited the range of creosotebush-bursage and kept it from invading desert grasslands (Humphrey 1974). Since the early 1900s white bursage has become dominant to creosotebush on disturbed sites. McAuliffe (1988) reports that creosotebush may use white bursage as a nurse plant. Bursage species are easily topkilled but can resprout. Following a fire, cover of creosotebush and bursage is reduced but then increases over time (Marshall 1994). Because fuel loading can vary seasonally and annually, fire management considerations in the creosote-bursage type requires a site-specific analysis of plant cover, fuel loading, and fuel continuity.

Mesquite

Succession--Mesquite density and distribution increased prior to 1900 with fire suppression and seed disperal by livestock. After 1900 mesquite continued to increase even though numerous eradication practices such as biological control, herbicides, mechanical removal, and prescribed burning were used to limit its density and spread—with mixed results (Glendening 1952; Jacoby and Ansley 1991; Wright 199; Wright and Bailey 1982).

Management Considerations--Fire as a management tool for controlling mesquite has its limitations. Mesquite may become more prevalent 5 years following a burn than it was before fire (Martin 1983). Mesquite can root sprout; top-killed individuals may resprout from dormant buds found in upper branches or from the base of the trunk below the ground surface. Mesquite seedlings can survive fire (Cable 1961), but on a burned site mesquite is sometimes reduced (Wright 1980). Fire may kill a good proportion of mature mesquite, particularly the smaller trees (<2 inch diameter) (Cable 1949, 1973). It is most susceptible to fire during the hottest and driest part of the year (Cable 1973). Drought years may increase mortality of mesquite if eradication is attempted. If managers wish to open dense mesquite stands, then roots must be killed, not just above ground biomass. Fire can be used to reduce the density of young mesquite populations, particularly during dry seasons that follow 1 to 2 years of above normal summer precipitation (Wright 1980). Adequate precipitation, no grazing, and using fire about every 10 years allow grasses to successfully compete with mesquite (Wright 1980). Rehabilitation of mesquite-invaded grasslands requires removal of livestock before burning, otherwise the shrubs outcompete the grasses (Cox and others 1990). Shrub reinvasion depends on grazing management combined with continued use of fire at the desired frequency (Wright 1986).

In managing for mesquite savanna(Ansley and others 1995, 1996b, in Press; James and others 1991), shaded rangeland may be a preferred condition rather than attempting to completely eradicate mesquite (see the **Texas Savanna** section). Low-intensity fire may allow mesquite to retain apical dominance on upper branches while reducing overall foliage. Season, air temperature, relative humidity, and duration and temperature of fire were factors reported to affect mesquite response to fire (Ansley and others in press). Mesquite topkill is related to heat in the canopy, not at the stem bases. Single and repeated summer burns kill mesquite aboveground, but do not kill roots (Ansley and others 1995). Prescribed burning may be used to kill mesquite seedlings while leaving tree sized and shaped older individuals (James and others 1991).

Paloverde-Cactus Shrub

Succession--Prior to 1900, fires in paloverde-cactus shrub were not considered to be important and occurred mainly in the restricted desert grasslands (Humphrey 1963). Conversion of desert shrubland to grassland to enhance forage for livestock and wildlife had been the primary land-use goal during the 1800s (Martin and Turner 1977; Phillips 1962). The high shrub component in this desert is attributed to historic overgrazing and overburning.

Since 1900, increases in ignitions and fire size are evidence of changing land management practices in the paloverde-cactus shrub. Exotic grass invasion now supplies a contiguous fuel source in many areas so that the historical small and infrequent fires were replaced by more frequent and larger fires (Narog and others 1995). Rogers (1986) speculated that finer fuels and highter rates of spread may allow desert fires to become larger than nondesert fires before being controlled. Although many of the species in this vegetation type can resprout (Wilson and others 1995b), postfire communities generally experience changes in species composition, particularly with an increase in the grass component, at the expense of cacti and succulents (Cave and Patten 1984; McLaughlin and Bowers 1982; Rogers and Steele 1980).

Management Considerations--Current management policy for some of the paloverde-cactus shrub vegetation now includes multiple interests with an increasing emphasis on recreation and tourism. This new policy involving reduced grazing, an increasing number of ignitions, and a greater herbaceous component is altering the fire regime (Robinett 1995). Fire dynamics information is required to effectively manage these changing needs. The increase in fire frequency and size may have serious consequences particularly for plant and wildlife species of special interest such as the giant saguaro (Thomas 1991; Wilson and others in press) and the desert tortoise; both may be fire intolerant. Little information exists on maintaining desert speciesin the presence of fire. Restoration in the paloverde-cactus shrub type needs to be addressed if the thousands of acres recently burned are to be rehabilitated.

Southwestern Shrubsteppe

Succession-- Historically, fire suppression and seed dispersal by herbivores have allowed grama-tobosa range to become dominated by creosotebush, tarbush, and mesquite. Tobosa is an early postfire seral component. Since the 1900s fire has been used to regenerate decadent stands of tobosa. Fire may stimulate or damage grama depending on climatic conditions, season, and fire severity. Reestablishment after fire is generally through stolons. Grama species can regenerate by seed, stolons, rhizomes, ortillering; tobosa mainly regenerates by rhizomes.

Management Considerations-- Tobosa can be managed with prescribed fire, which causes low mortality, improves palatability, and increases biomass production. Tobosa is one of the few native grasses that have competed well with nonnative grasses. Spring burns produce the best results when precipitation is adequate. Litter of up to 3.0 tons/acre (6.7 t/ha) easily carries fire and is completely consumed. Broomweeds, snakeweeds, and fire whirls are prescribed burning hazards in tobosa. For optimum forage production prescribed burns should be conducted every 5 to 8 years on tobosa stands. Nonbunch grass species of grama may take 2 to 3 years to recover following fire.

Grama and tobosa supply abundant forage for livestock and wildlife. Grama is palatable all year, but tobosa is poor forage in winter months. Black grama is drought adapted and can be used for restoration to prevent soil erosion.

Trans-Pecos Shrubsteppe

Succession--Historically, junipers were relegated to rock out crops and upland limestone sites, preferring shallow limestone soils. Fire suppression and overgrazing have allowed the woody species to expand from their historically more limited range onto the mixed prairie, sometimes in dense stands (Sparks and others1990). Dense juniper stands are highly competitive and reduce understory grassy forage. Junipers dominate over oaks on drier sites, are shade intolerant, and may be succeeded by pinyon pine. Junipers are facultative seral trees with extensive lateral roots that effectively compete for surface moisture in xeric environments.They may or may not root sprout depending upon species. Chemical control, mechanical control, and prescribed burning have been used to reduce juniper density to improve rangeland forage productivity.

Management Considerations--Management techniques to reduce juniper and shrub density to improve rangeland for livestock are employed in many areas. Prescribed burning in junipers is recommended to open dense stands; however,ground fuels are not always adequate to carry fire. Ahlstrand (1982) found that plant response to fire in this community is predominantly by vegetative means. He suggests that prescribed burning can be used to improve the grass component at the expense of the shrubs. Pretreatment with chemicals or mechanical methods is also recommended. A minimum of 1,000 lb/acre (1,120 kg/ha) of continuous fine fuels is needed for prescribed burns (Rasmussen and others 1986). Fire history studies suggest that fire-free intervals of less than 50 years restrict the expansion of junipers, and that nested fire cycles have actually driven the juniper's range (Bunting 1994). Fire rotations of 10 to 40 years are recommended to control junipers. Reburn intervals between 20 to 40 years or when junipers reach 4 feet tall are recommended to maintain converted grasslands (Wright and others 1979). Variable fire effects in this type can be obtained (Tausch and others 1995). For specific fire prescriptions refer to Wright and others (1979).

Mechanical treatment followed 5 years later by burning to kill saplings is recommended to maintain a landscape mosaic of open stands and grassland. Mature junipers in moderate to dense stands are resistant to fire, yet may suffer some mortality. Small stemmed individuals are easily killed by fire. Rapidly burning grass fires occuring at intervals of 10 years or more are adequate to allow juniper saplings to reach sufficient heights 3 to 6 feet (1 to 2 m) to withstand fire injury. Burned areas may be invaded through seed dispersal, and establishment can occur within 10 to 40 years (Rasmussen and others 1986). Dead junipers are volatile fuels, and spot fires from fire brands can be a problem.

Fauna in the Trans-Pecos shrubsteppe ecosystem are similar to the species found in desert grasslands. Pronghorn and deer are widely distributed across the shrubsteppe range as are dove, quail, rabbits, and small rodents. Javelina are common in the south. Common carnivores include coyote, bobcat, eagle, owl, and hawk. Juniper berries and acorns are a favored food by many species.

Chaparral-Mountain Shrub

Pre-1900 Succession--The species that we refer to collectively as chaparral evolved as a component of the understory of Laurentian forest types.They were adapted to harsh conditions and could withstand disturbance. Chaparral development had no particular relation to fire (Axelrod 1989).With warming and drying trends, chaparral species became more opportunistic and were able to fill niches once occupied by species less able to compete under these conditions. A disturbance that chaparral was able to cope with was fir — an element whose presence was probably important in providing opportunity for chaparral to attain status as a recognizable vegetation type. By the end of the 19th century, newspaper accounts of fires burning through this type for days and weeks in southern California became common. By accounts of historic fires, maps of vegetation from the first third of the 20th century, local lore, and by remnants of previous vegetation, a picture of chaparral's ecological amplitude begins to emerge with fire as an important environmental component. This logic continues into the 21st century.

Post-1900 Succession--The benefit of more or less real-time observation allows us the opportunity to fine tune our view of chaparral's role in succession. The dynamics of chaparral's environment have made it difficult to definitively document chaparral's role in the successional process. Several salient points can be made, however, with little fear of argument:

- Chaparral succeeds many forest types after a major disturbance — whether from fire or logging. It is often seral-especially at elevations where we currently consider chaparral as a montane understory type. Given a reasonable number of disturbance-free years, the forest type will regain dominance.
- Chaparral often succeeds chaparral after fire, especially at elevations where we consider chaparral as the dominant vegetation type. Species composition can shift drastically, probably depending on whether the fire occurred before or after seed set for a given species. The concept of an infinite store of chaparral seed in the soil is becoming more and more questionable due to seed predation by rodents, ants, and birds (Quinn 1980).
- The concept of chaparral being a fire climax refers to a delicate balance between characteristics of the chaparral species on a site and the fire regime. Fire frequency and timing can tip the balance so that chaparral can be overtaken by herbaceous vegetation types, such as annual grasses, and in southern California by anallied **"soft chaparral"** type--a highly volatile semiwoody group of shrubs. But the present fire regime appears to be about the same as during the presettlement period. Conard andWeise (1998) presented evidence that fire suppression has offset increased human ignitions during the past century, thus preventing fire frequency from increasing to the point of degrading the ecosystem. Area burned per year, size of large fires, and seasonality of fires in chaparral changed litle during the past century.

Management Considerations--Management of chaparral has been directed primarily at concerns about fuel hazard, wildlife habitat, and as a cover type that plays an active role in maintaining slope stability and watershed capability. Some would prefer to manage chaparral as a problem because it occupies potential rangeland that could be used for livestock grazing. All can, in their place, be perfectly good reasons for managing chaparral, but you have to pick one-or maybe two.

Prescribed fire can be used to remove dead fuel for hazard reduction, increase structural diversity for wildlife habitat purposes, and increase the proportion of young biomass in a stand — for both hazard reduction and wildlife habitat improvement. In some areas, but not all, prescribed fire can be used to maintain stands of chaparral in their current state (that is, to maintain a fire climax). For prescribed fire to be successful, species that reproduce only from seed, the presence of seed must be assured. Some chaparral seeds need scarification, which fire often provides. Besides heat-shock scarification, smoke-induced germination is important to many chaparral species (Keeley and Fotheringham 1998). Seeds of chamise can be destroyed if directly exposed to fire. Many chaparral species sprout after fire; reproduction from seed is not as much of an issue for these species as long as individual plants are not killed. However, next to nothing is known about the effects of physiological age on sprouting ability of chaparral species.

Individual shrubs can be killed outright by fire. Shrubs lacking in vigor will probably not respond to fire in their normal fashion. Thus, stresses such as protracted drought might cause an unexpected effect if fire were to be introduced. An extremely severe fire can result in little reproduction from either sprouting or seed germination. A series of fires wiht short return intervals may result in reduced chaparral shrub density if shrubs burn before they reach seed-bearing age, or young shrubs developing from sprouts are physiologically unable to respond.

Extremely old chaparral stands can be found with little or no dead material in them, and others can be found with a significant down and dead component (Conard and Weise 1998). The difference can be dictated by species composition, site conditions, and history of the site. Management of stands with a lot of dead material in them has to be taken on a case by case basis. From a fuels standpoint, these stands do have an elevated hazard level. Whether or not they present a serious threat should be evaluated in light of their juxtaposition to other resources and the condition of the other resources. Old chaparral stands should not automatically be considered as "decadent."

Conversion of chaparral to rangeland has to be undertaken with caution. Soil and slope conditions should be evaluated to avoid loss of soil. Forth is reason, steep slopes and easily eroded soils should be avoided in conversion projects. In all cases, chaparral management should be undertaken with a clear view of species present, site conditions, stand history, fuel situation, and successional potential.