Timothy E. Paysen
R. James Ansley
James K. Brown
Gerald J. Gottfried
Sally M. Haase
Michael G. Harrington
Marcia G. Narog
Stephen S. Sackett
Ruth C. Wilson

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 the 100th meridian. Shrublands include sagebrush, desert shrub, southwestern shrub steppe, Texas savanna, and chaparral-mountain shrub ecosystem types. Woodlands include southwestern 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 southwestern New Mexico and southeastern 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 at 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 K019		M	1a:2-10	m	1			
	Pine-cypress forest K009	Arizona cypress 240			M	1,2	m	1	
Pinyon-juniper 35	Juniper-pinyon K023	Rocky Mountain juniper 220			M	1			
	Juniper-steppe K024	Western juniper 238			M	1			
		Pinyon-juniper 239			M	1			
		Arizona cypress 240			M	1			
Southwestern oaks [d]	California oakwoods K030	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 K031	Interior live oak 241			M	1			
Shinnery 31	Shinnery K071	Mohrs oak 67			M	1			
Texas savanna 32	Ceniza shrub K045				M	1			
	Mesquite savanna K060	Mesquite 68, 242			M	1			
	Mesquite-acacia savanna K061				M	1			
	Mesquite-live oak savanna K062	Western live oak 241			M	1			
	Juniper-oak savanna K086	Ashe juniper 66			M	1			
	Mesquite-oak savanna K087				M	1			
Sagebrush 29	Sagebrush steppe K055						M	2a:20-70	
	Juniper steppe K024	Rocky Mountain juniper 220					M	2a	
	Great basin sagebrush K038	Western juniper 238					M	2a:20-70	
	Wheatgrass-needlegrass shrubsteppe K056						M	2a	
Desert shrub 30	Mesquite bosques K027	Mesquite 68, 242					M	1,2a	
	Blackbrush K039						M	1,2a	
	Saltbrush-greasewood K040						M	1,2a	
	Creosotebush K041						M	1,2a	
	Creosotebush-bursage K042						M	1,2a	
	Paloverde-cactus shrub K043						M	1,2a	
	Cresotebush-tarbush K044						M	1,2a	
SW shrubsteppe 33	Grama-tobosa K058						M	1,2a	
	Trans-pecos shrub savanna K059						M	1,2a	
Chaparral-Mountain shrub 34	Oak-juniper woodland K031						M	1,2a	

(con.)

Table 6-1—Con.

			Fire regime types						
			Understory		Mixed		Stand-replacement		
FRES	**Kuchler**	**SAF**	**Occur**[a]	**Freq**[b]	**Occur**	**Freq**	**Occur**	**Freq**	**Nonfire**
	Mountain mahogany-oak scrub K037						M	1,2a	
	Transition of K031 & K037						M	1,2a	
	Chaparral K033						M	1,2a	
	Montane chaparral K034						M	1,2a	
	Coastal sagebrush K035						M	1,2a	
Plains grasslands 38	Grama-needlegrass-wheatgrass K064						M	1	
	Grama-buffalograss K065						M	1	
	Wheatgrass-needlegrass K066						M	1	
	Wheatgrass-bluestem-needlegrass K067						M	1	
	Wheatgrass-grama-buffalograss K068						M	1	
	Bluestem-grama prairie K069						M	1	
	Mesquite-buffalograss K085	Mesquite 68, 242					M	1	
Desert grasslands 40	Grama-galleta steppe K053						M	1,2a	
	Grama-tobosa prairie K054						M	1,2a	
	Galleta-three-awn shrubsteppe K057						M	1,2a	
Annual grasslands 42	California steppe K048						M	1,2a	
Mountain grasslands 36	Fescue-oatgrass K047						M	1	
	Fescue-wheatgrass K050						M	1	
	Wheatgrass-bluegrass K051						M	1	
	Foothills prairie K063						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.

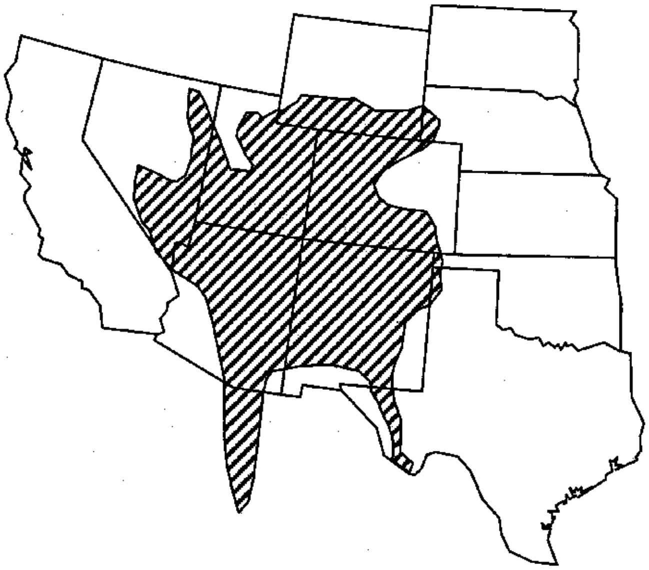

Figure 6-1—Southwestern ponderosa pine distribution.

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 slow decomposition 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 for sapling thickets, pole stands, 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; Wade 1969; 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 others 1968, 1976, 1977), age (Aldon 1968), height and diameter (Sackett and Haase 1991), 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 F layer 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 F layer 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 t/ha) 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, pole stands, mature old growth (yellow pine) groves, and open areas in the groves

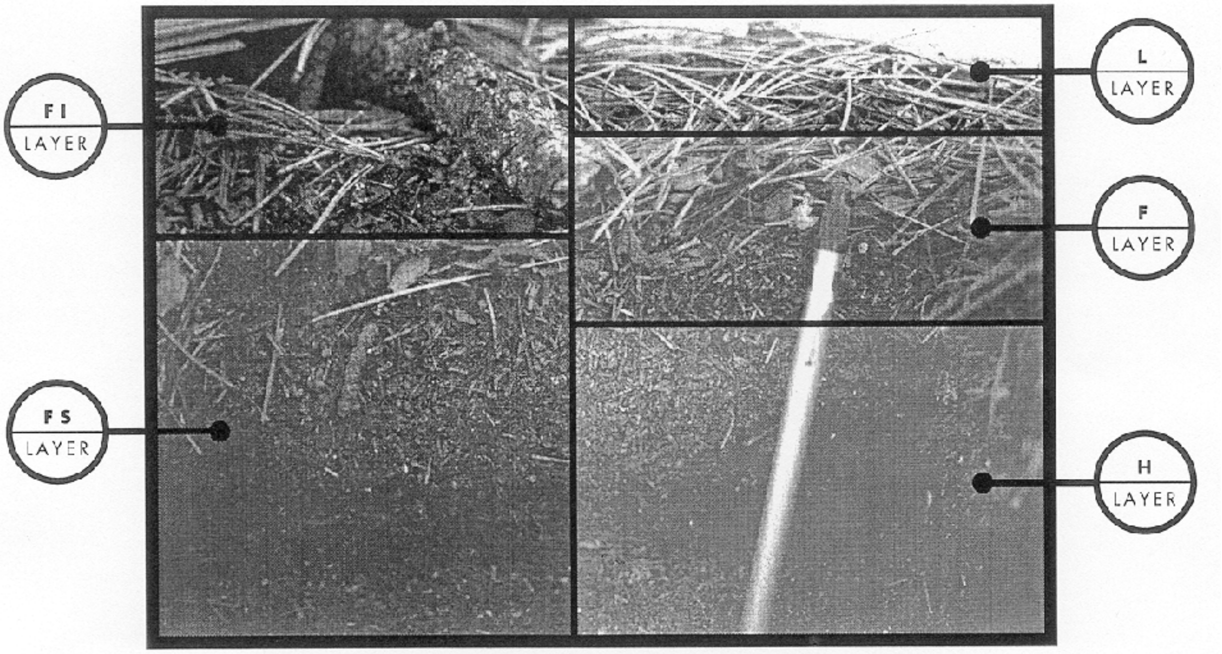

Figure 6-3—Section of ponderosa pine forest floor showing the fire intensity (FI) layer of fuel and fire severity (FS) layer of fuel in relation to the L, F, and H layers of the forest floor.

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 >1-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 built up sufficiently with the rapid resprouting of grasses and the abundant annual pine needle cast. Large woody fuels in the form of branches or tree boles, 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 ponderosa 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 grass fuels were reduced (Faulk 1970). Subsequently, pine regeneration increased because of reduced understory competition, less fire mortality, and more mineral seedbeds (Cooper 1960).

Post-1900 Succession—In the early 1900s forest practices, and reduced incidence of fire, led indirectly 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 (Cooper 1960; 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, may be 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, coupled with 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.
- Bark beetle 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 belowground injury to roots and 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 the region, 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

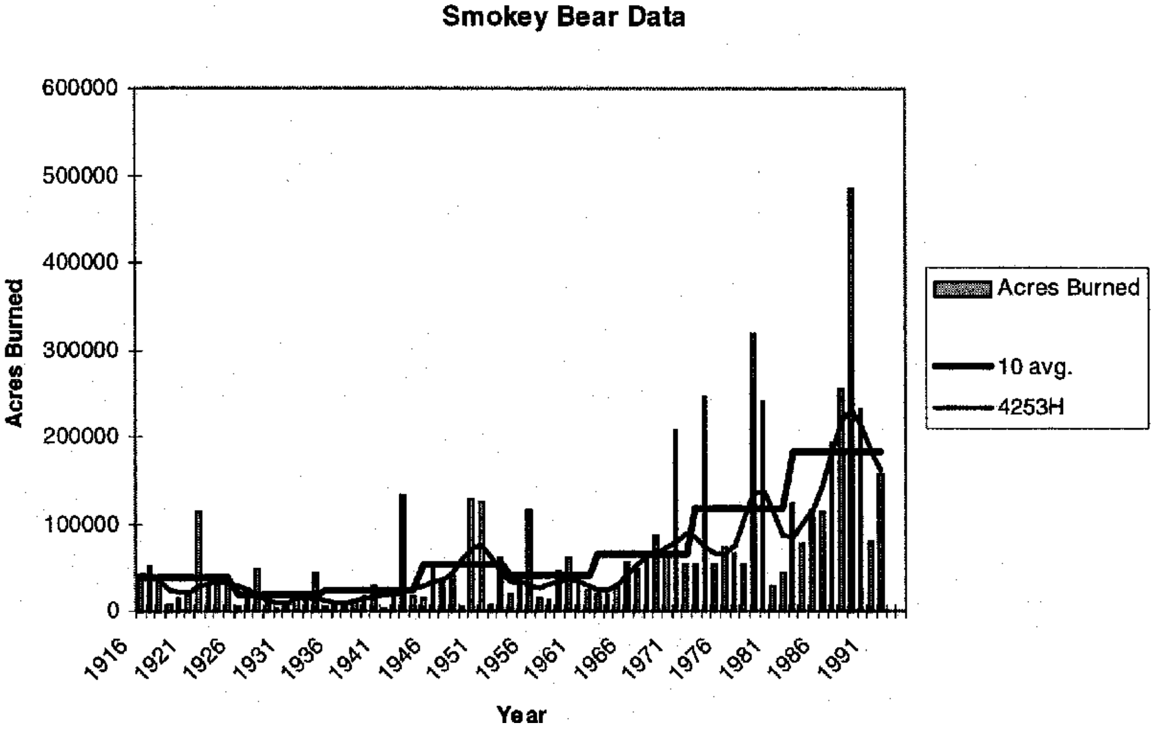

Figure 6-4—The total number of acres burned by wildfires in Arizona and New Mexico from 1916 to 1990, USDA Forest Service Smokey Bear fire summary reports. Heavy line represents 10-year average; light line represents trends using the 4253H mathematical filter, used for smoothing noise in data.

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 wildfire 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 represented (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 Regimes

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

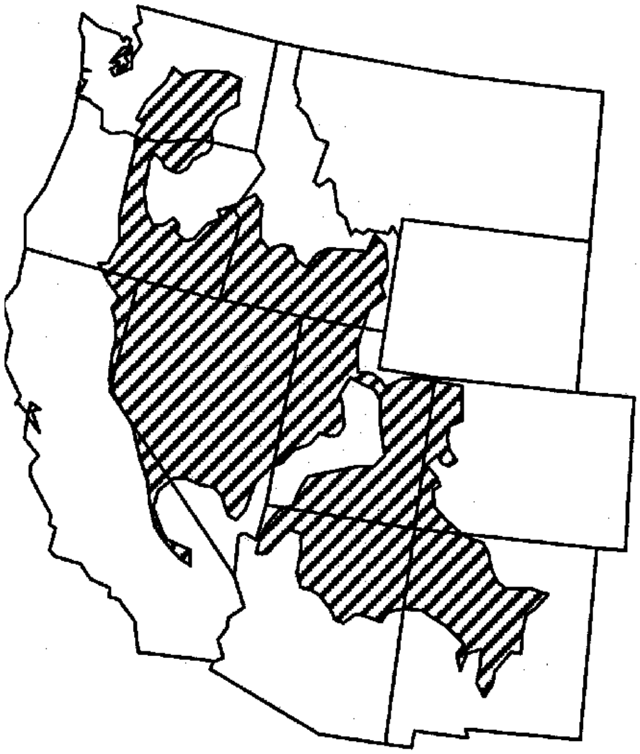

Figure 6-5—Pinyon-juniper woodlands distribution.

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).

The Texas 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.

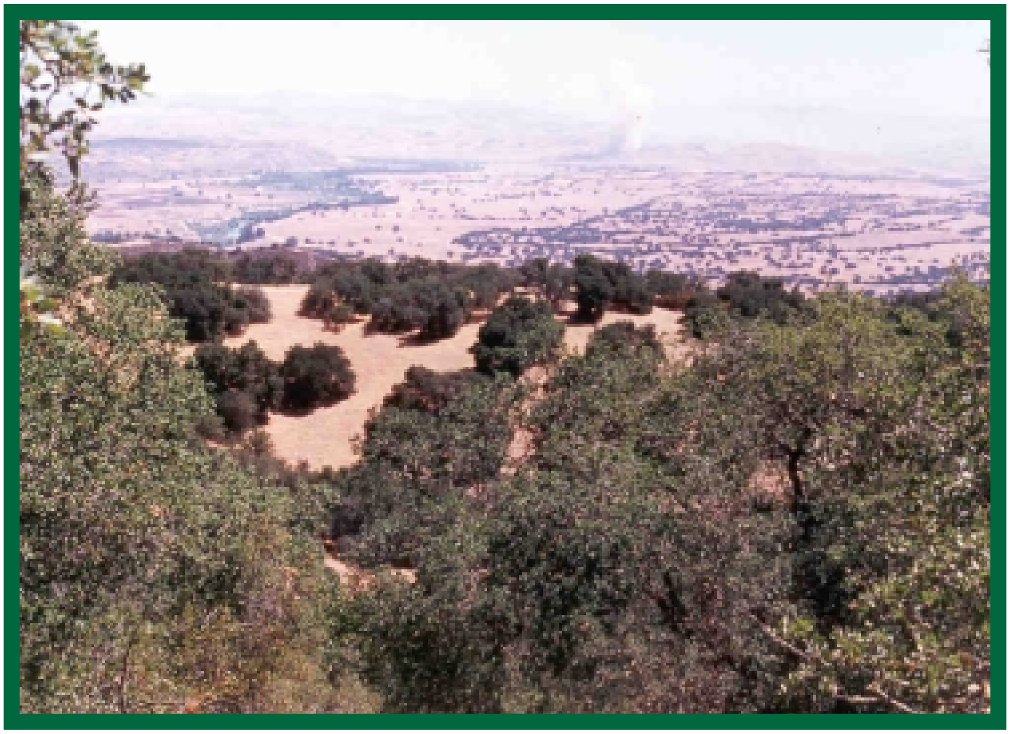

Figure 6-7—Western oak woodlands, Camp Roberts Military Training Reservation, Paso Robles, California.

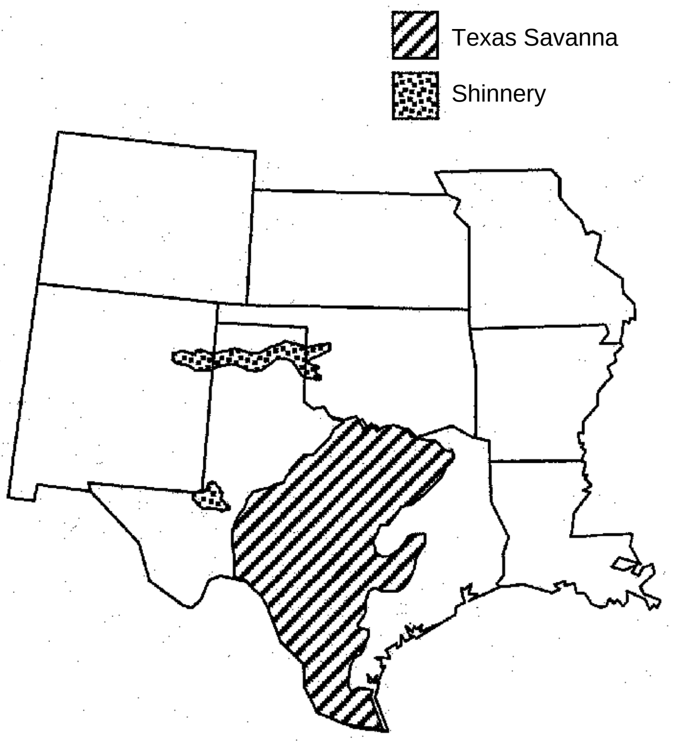

Figure 6-8—Distribution of shinnery and Texas savanna FRES ecosystems.

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. In fact, 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 "Great Valley" 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, acacias, oaks, junipers, 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, indiangrass, 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 Characteristics

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 standwide fires, 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 wind speeds. 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 others 1995). 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 oneseed juniper trees less than 3 to 4 feet (1 m) 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 (Dealy 1990) 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 2,240 lb/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 600 lb/acre). Typical crown fuels are 3.6 tons/acre (8.1 t/ha) for foliage 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 nonwoody 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.

Texas Savanna—The predominant 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 live root 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 live oak, 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 others 1977):

Class	Productivity (lb/acre)
1	2,250 to 3,000+
2	1,500 to 2,250
3	750 to 1500
4	0 to 750

Postfire Plant Communities

Pinyon-Juniper

Pre-1900 Succession—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 (Evans 1988). 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 (Erdman 1970), similar to that of Arnold and others (1964), progresses from skeleton forest and bare ground, to annual stage, to perennial grass-forb stage, to shrub stage, 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 bare ground, 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. A 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 and Wittie 1990). 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 the 16th century. Overgrazing has had an important effect on the role of fire in the woodlands. The reduction of cover of herbaceous 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 is used 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 (Tiedemann 1987) 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 overland 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 Klebenow 1979). 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 (1973) 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 sufficient 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 fire 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, overt 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 Plumb and 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 (1854) described much of Texas rangeland as open grasslands with scattered large mesquite (a mesquite savanna). Marcy (1866) described some upland areas of central Texas 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 a part 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 late 19th 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 the vegetation 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, land clearing 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 out compete mesquite seedlings (Archer 1989; Brown and Archer 1989; Bush and Van Auken 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.

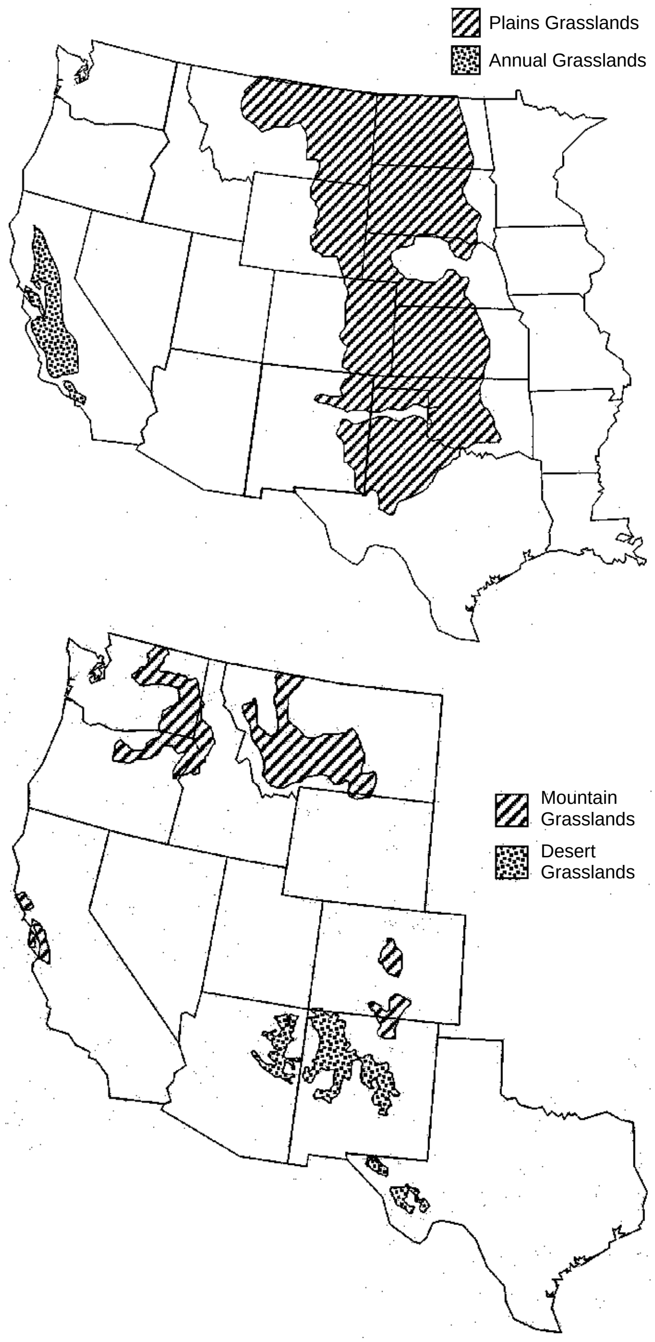

Figure 6-9—Distribution of plains, mountain, desert, and annual grassland FRES ecosystems.

- 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 6-10—Cheatgrass.

Shrublands

Shrublands are described here as desert shrubland types and the chaparral-mountain shrub type. 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 500,000 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 1985). 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; MacMahon and Wagner 1985; 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, whereas 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

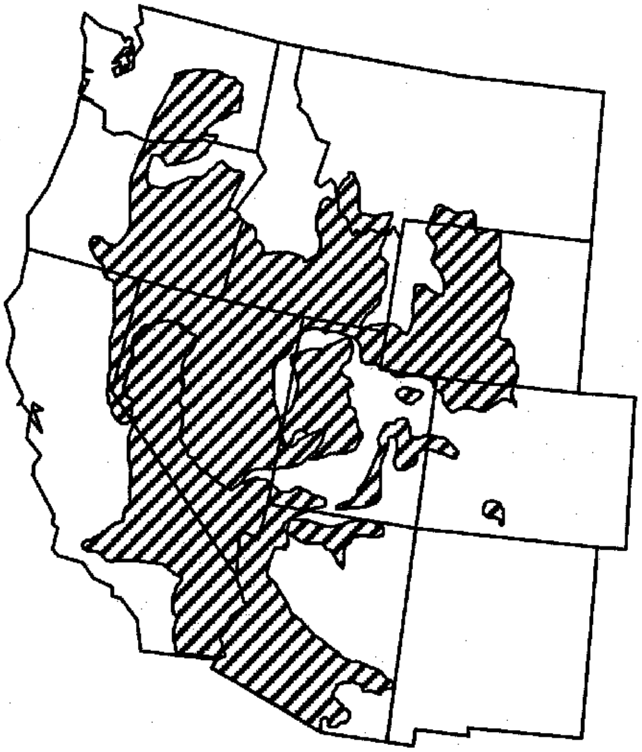

Figure 6-11—Distribution of desert shrub FRES ecosystems.

Table 6-3—Physiognomic fuel types for desert shrublands[a] associated with the four North American deserts.

	North American deserts			
Desert shrublands	**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 shortgrass prairie, consisting of sparsely distributed bunch grasses, 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 Wiggins 1964; Turner 1982; Turner and Brown 1982; Vasek and Barbour 1977). We focused on desert shrublands within the United States (table 6-1).

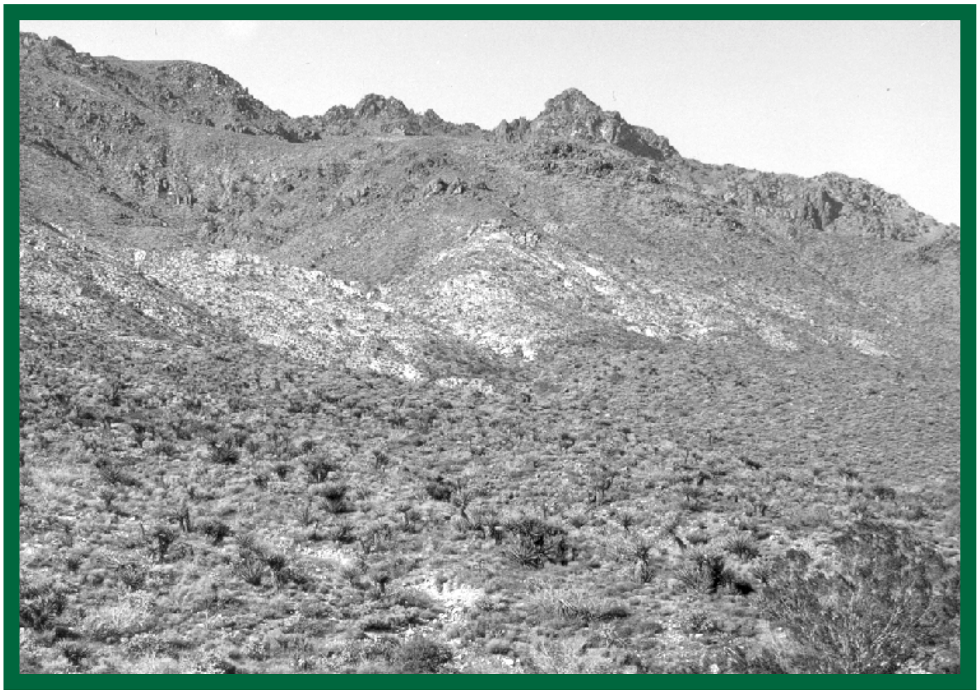

Figure 6-12—Bare soil, evident between shrubs and small trees, is a common characteristic of North American deserts as seen in the Mojave Desert, California.

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 between 1,600 and 11,000 feet (490 and 3,500 m) with varied soils derived from lava flows, ancient lake beds, 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 2 m) tall and grow in dense clumps or scattered plants. Shadscale, spiny hopsage, Mormon tea, and milkvetch are important co-dominants in this vegetation type. Understory grasses such as wheatgrass, brome, fescue, and bluegrass, and variable forbs form discontinuous patches with bare soil.

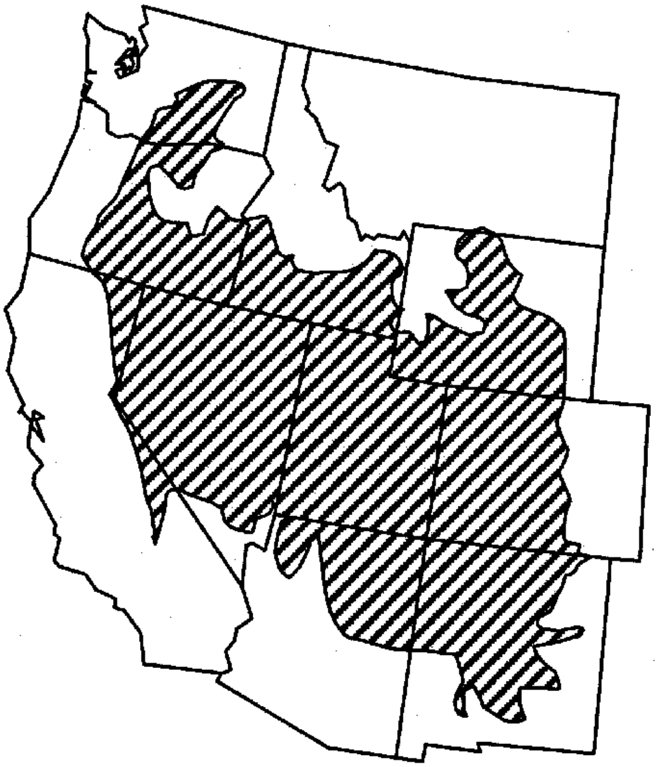

Figure 6-13—Distributiuon of Great Basin sagebrush FRES ecosystems.

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 pure stands, 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 include shadscale, 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, gentle slopes, 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 managers 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, pricklypear, cholla, saguaro, and bursage are dominant species in this vegetation type. These communities are a 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 semidesert 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 types are recognized. The **Grama-tobosa 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-Pecos shrub 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 Regime 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, an ever-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 others 1985b). Grasslands, occupying flat to gently rolling terrain, would burn over 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[a] **-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	• *Astragalus, Lupinus, Phacelia*			H	
Fo-0 to 700 lb/acre	• *Agropyron*			G	
Blackbrush		S			
Dense to open broadleaf	• *Artemisia, Gutierrezia, Haplopappus*		S		
evergreen shrubs	• *Ephedra*		s		
± herbaceous understory	• *Hilaria*			G	
Fo-250-500 lb/acre					
Saltbush/black greasewood	*Atriplex confertifolia/Sarcobatus vermiculatus*		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 brevifolia*[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	• *Cercidium, 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 glandulosa; P. velutina*	T			
Open to dense forest low	• *Cercidium, Olneya, Prosopis, Populus, Dalea, Salix*	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, Larrea, Lycium, Simmondsia, Acacia, Condalia, Fouquiera, Celtis*		S		
Fu-100 to 250 lb/acre					
Fo-30 to 100 lb/acre	• *Calliandra, Ephedra, Franseria, Janusia*		s		
	• *Carnegiea*				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					
- - - - - - - -					

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[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.

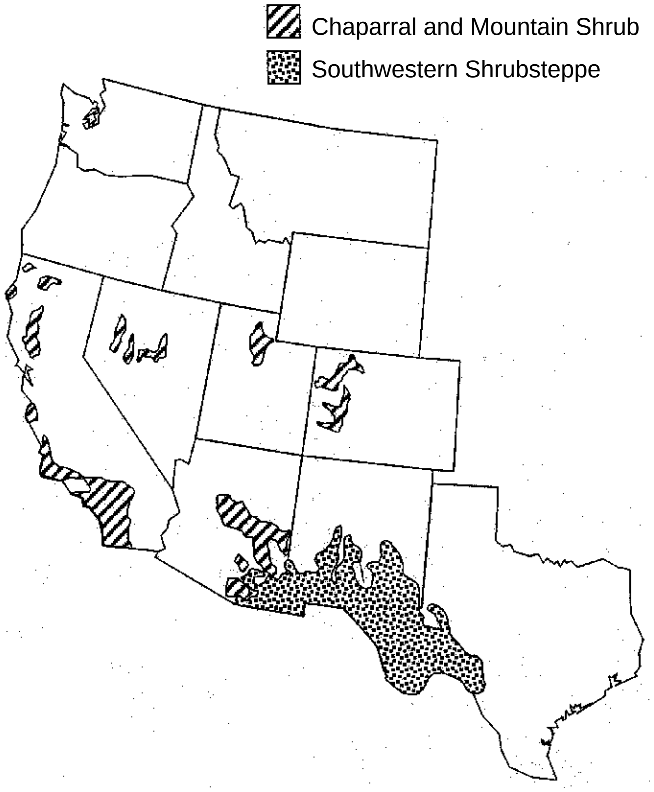

Figure 6-19—Distribution of Southwestern shrubsteppe and chaparral-mountain shrub FRES ecosystems.

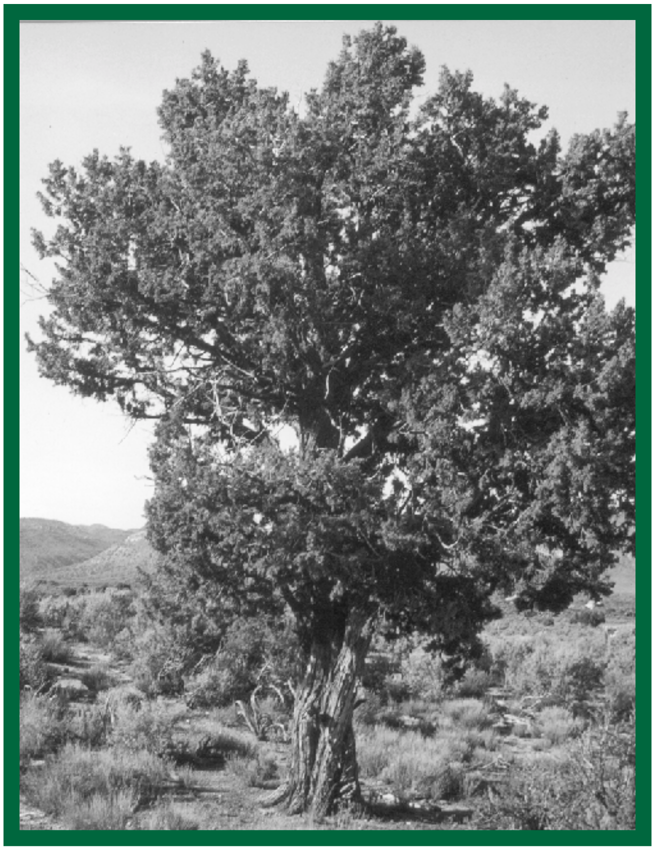

Figure 6-20—Mixed fuels found in juniper shrub savanna (New York Mountains, California).

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, a long 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 saltflats 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 infrequent 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 aboveground 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 for long 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 grass fuels, 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 in a 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 rates 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 susceptible 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 frequently undergo severe fire seasons.

Desert Shrublands—Fuels include cacti and other succulents, grasses, shrubs, small trees, and mixtures of these. Fuels occur in discontinuous patches to areas where trees, shrubs, and grasses are contiguous. Fuel loadings may reach 2,000 lb/acre (2,240 kg/ha) (fig. 6-22, 6-23). See table 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 feet in height and 20 percent cover, conditions typically found, sagebrush foliage and stemwood 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 soils and 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, Maricopa County, Arizona, increasing the potential for major fires.

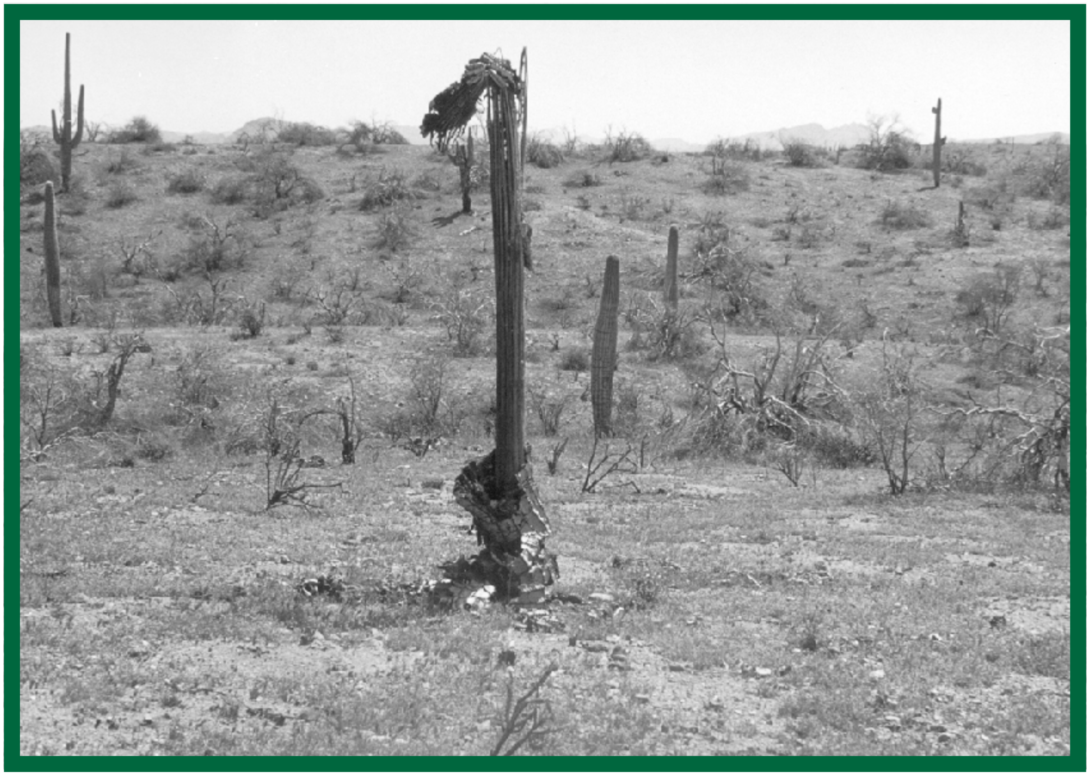

Figure 6-23—A wildfire burned 10,000 acres of this Sonoran desert thorn-shrub in Four Peaks, Tonto National Forest, Arizona.

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) per annum 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 (2,240 kg/ha) 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 750 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 highly 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 distribution 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 broadleaf 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.

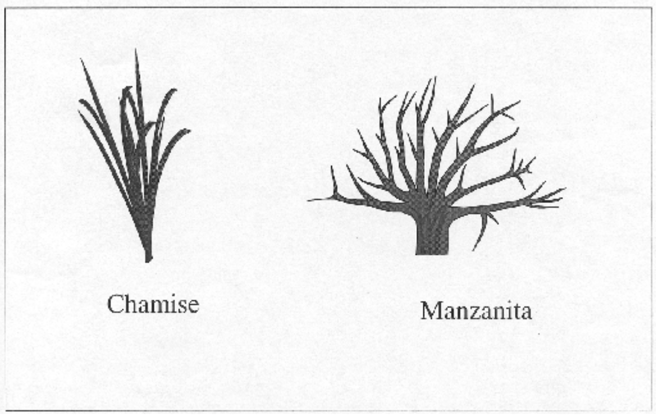

Figure 6-26—Arrangement, distribution, and size of woody fuels can vary by species.

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 onsite 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 broadleaf 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 "dust bowl," 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 northernmost 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 overutilization 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. Overutilization 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. A range 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 consistent with 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 checking and reduction of woody plant encroachment. Mountain grasslands were intertwined with forests and shrublands ranging from rose and aspen in Alberta to conifers and sagebrush of Rocky Mountain foothills. Encroachment into grasslands by woody species was an ongoing process kept in check by repeated fires.

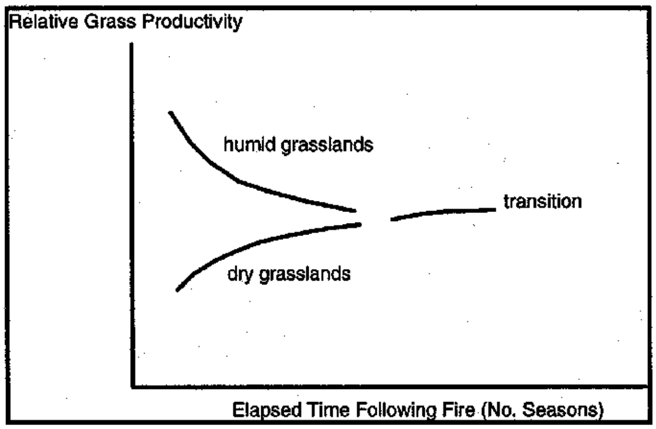

Figure 6-27—The productivity of humid grasslands versus dry grasslands after fire.

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 become 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-thread grass 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 between 1907 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 decades, 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 1986a; 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 seed bank 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—Native species 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 greenstripping 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 a part 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 “bottomland” 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 desert landscapes. 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 management practices 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 grass and 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 grass forage (Cornelius and Talbot 1955; Pechanec and others 1954; Pechanec and Stewart 1944; Reynolds and others 1968). 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 regrowth is 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), fuel loading, fuel moisture content, and windspeed (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 dominate an 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 for sagebrush regeneration (Blank and others 1995). Sagebrush seed can be viable up to 4 years. Sagebrush can be restored through reseeding. Cheatgrass seed banks present on sagebrush sites may negatively influence reestablishment of native bunch grasses and shrubs (Hassan and West 1986). If sagebrush is in good "natural" condition an initial postfire influx of cheatgrasss will occur. Given adequate precipitation, perennial native grasses and shrubs can outcompete 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 cheatgrass 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 land management 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 (Callison and others 1985) because blackbrush is often replaced by species of similar forage potential.

Management Considerations—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 wildfire 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 vegetation type (Harper and others 1990). Intense fall fires may increase plant mortality in spite of a species' resprouting potential. Some *Atriplex* 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 tissue and may be used to reduce soil salts and 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 Herbel 1965). 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 hand, it is sensitive to fire, especially in spring (Brown and Minnich 1986; McLaughlin and Bowers 1982). 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) report slow 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, mountain goats, 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 agassizi*) 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 dispersal 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 1990; 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 aboveground 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 higher 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 species in 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, or tillering; 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 firewhirls are prescribed burning hazards in tobosa. For optimum forage production prescribed burns should be conducted every 5 to 8 years on tobosa stands. Nonbunchgrass 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 outcrops 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 others 1990). 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 occurring 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 firebrands 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 fire—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 an allied "**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 and Weise (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 little 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 with 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. For this 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.

Notes

Ronald L. Myers

Chapter 7: Fire in Tropical and Subtropical Ecosystems

Apart from savannas and grasslands, wildland fire in tropical environments has received scientific scrutiny only within the past few decades. It is now widely recognized that the vast majority of wildland fires occur in the tropics and subtropics (Goldammer 1993). Because a global treatment of the effects of fire in tropical and subtropical ecosystems is beyond the scope of this volume, the reader is referred to J. G. Goldammer (1990), wherein Mueller-Dombois and Goldammer (1990) outlined generalized tropical and subtropical fire regimes.

In this chapter, we focus only on fire effects in subtropical Florida, Puerto Rico, the United States Virgin Islands, and Hawaii by drawing on appropriate literature from southern Florida, the Caribbean, Mexico, and the Pacific Islands.

Understory Fire Regimes

Major Vegetation Types

In the regions covered here, major vegetation types having woody dominants exposed to surface fires that are generally nonlethal to the overstory occur only in subtropical Florida. They include (1) pinelands and savannas dominated by slash pine (var. *densa*), (2) wetland savannas and woodlands dominated by pondcypress, (3) cabbage palmetto forests and savannas, and (4) southern live oak hammocks (a local term for groves or forests dominated by hardwoods). Close analogs of these vegetation types are the Caribbean pine forests, woodlands, and savannas of the Bahamas, Cuba, and coastal Central America from Belize to Nicaragua; some of the other pine types in Cuba; the mountain pinelands of Hispaniola; and some of the seasonally inundated palm swamps and savannas in the tropical and subtropical Americas. There does not appear to be an obvious analog of the pondcypress type: a subtropical, fire-tolerant, wetland conifer forest or savanna, which occurs in seasonally flooded depressions throughout the Southeastern Coastal Plain of the United States, but reaches its greatest coverage in subtropical Florida. For a synopsis of the role and effects of fire in southern Florida ecosystems see Wade and others (1980) and Myers and Ewel (1990).

All of southern Florida's vegetation types characterized by understory fire regimes are found on low, flat, poorly drained substrates. The landscape consists of a vegetation mosaic where hydrology exercises considerable influence over the availability of fuels and

Table 7-1—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**
Longleaf-slash pine 12	Subtropical pine forest K116	S. Florida slash pine 111	M	1a: 1-5	m	1b			
		Cabbage palmetto-slash pine 86	M	1a	m	1b			
		Caribbean pine[c]	M	1a	m	1b			
	Palmetto prairie K079						M	1a	
Oak-gum-cypress 16	S. floodplain forest K113	Bald cypress 101			m	2a	M	2b,3	
	Cypress savanna K091	Pondcypress 100	M	1					
	Mangrove K105						M	2	
		Live oak 89	M	1b	m	2a			
Wet grasslands 41	Everglades K092	Marshes [c]					M	1a	
Tropical hardwoods [d]							M	2,3	
Melaleuca [d]			M	1	M	1,2			

[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 vegetation type fits as a subdivision of FRES or Kuchler types, but is not an SAF type.

[d] Special type for chapter 7, not a FRES, Kuchler, or SAF type.

interacts with fire in determining the dominance of species. Along a hydroperiod (period the water table is above the soil surface) gradient, pondcypress occurs on the wettest sites. Relatively higher sites support pine, palm, or oak. The juxtaposition of the pine, palm, and live oak-dominated vegetation is controlled to a large degree by landscape features such as ponds, lakes, and drainages that create fire shadows where the intensity and incidence of fire is reduced. Twentieth century anthropogenic alterations of both fire and hydrologic regimes, coupled with fire exclusion in some areas and the introduction of non-native species, has markedly altered some of southern Florida's historic vegetation patterns and fire regimes.

Fire Regime Characteristics

Fire is essential for existence of the southern Florida woody vegetation types having understory fire regimes. These types are subject to nonlethal, relatively frequent, low-intensity surface fires ignited by either lightning or humans, although lethal long-return-interval ground fires may be important in the development of certain types of pondcypress communities. The shortest fire return intervals of 1 to 5 years are found in the pinelands and savannas. The longest fire intervals measured in centuries are found in bald cypress forests. If fire is removed as a process, or if the normal range of variability of the fire regime is altered, the character of the ecosystem changes.

Lightning is frequently cited as the primary ignition source responsible for southern Florida's fire-maintained vegetation. Florida has the reputation of being the "lightning capitol" of the United States, if not the world, with frequent, intense electrical storms generated during the summer by a double sea breeze system operating from both the east and west coasts of the Florida peninsula. Lightning fires occur during every month of the year, but with peaks in ignitions and area burned concentrated during the summer months (Snyder and others 1990). This period coincides with the season of locally generated convectional storms (fig. 7-1). The greatest area burned from lightning-ignited fires usually occurs at the onset of this season when fuels are dry and wetlands exhibit their lowest water levels.

We have little historical documentation of the burning activities of pre-Columbian Native Americans, so it is difficult to assess their influences on southern Florida fire regimes. The Calusa Indians disappeared shortly after contact due to European diseases and Spanish slave raiding parties from Cuba and the Greater Antilles. Because there were no Spanish settlements in, nor expeditions through, southern Florida, there are no known written records of Calusa burning practices. However, the conventional wisdom that lightning ignitions were the sole determinant of southern Florida's fire regimes has been challenged (Myers and Peroni 1983; Robbins and Myers 1992; J. R. Snyder 1991). Extensive, ecologically significant Native American burning in southern Florida can be inferred from archeological surveys, especially the number and distribution of occupation sites; from documented burning practices by the post-Columbian Seminoles, who were forced into southern Florida in the 1700s; by extrapolating from general principles of aboriginal burning elsewhere; and most significantly, from the aseasonal flammability of southern Florida's fuels. The latter means that the landscape could burn at any time of the year, and extensive areas possessed available fuels outside of the summer lightning season.

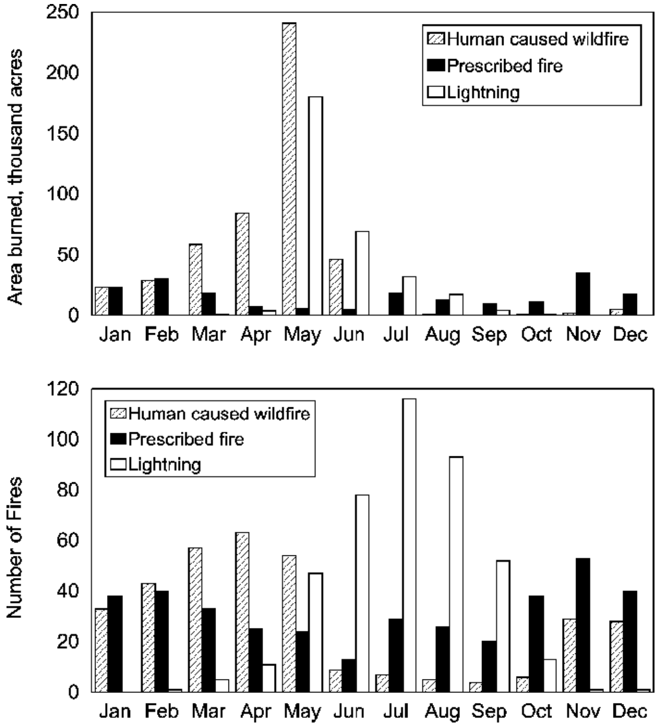

Figure 7-1—Monthly distribution of total number of fires and area burned from 1948 through 1997 in Everglades National Park, Florida, by ignition source. Courtesy of John Segar (Everglades National Park) and James Snyder (Big Cypress National Preserve).

The most compelling evidence suggesting an important aboriginal influence on fire regimes in subtropical Florida is provided in a vegetation and fire history of Andros Island in the Bahamas (Kjellmark 1995, 1996). Alternating periods of human occupation and depopulation over the past several thousand years are well documented on Andros Island. Pollen analyses indicated that during periods of human occupation, pinelands of Caribbean pine expanded at the expense of tropical hardwoods. During known periods of depopulation, the pinelands contracted considerably, persisting only on the western side of the island where lightning-caused fire may have been more prevalent. The Bahamian pinelands are nearly identical in structure, function, and species composition to some of southern Florida's pinelands.

Florida's present-day human-caused fires are concentrated in January through May with the greatest area burned in March and April. This predictable dry period, occurring before the onset of the lightning season, coupled with the year-round availability of fuels in some vegetation types, suggest that Native American burning likely shifted the seasonal component of fire regimes, increased the incidence of ignitions, and ignited fires at times when wetland vegetation types were likely to burn extensively. This suggests that reliance solely on lightning ignitions in the Everglades might lead to a landscape that may never have previously existed.

Fuels

The fuels controlling Florida's understory fire regimes consist of ground cover of perennial grasses, low woody shrubs, saw palmetto, and surface litter. Due to the presence of volatile oils and waxes, live surface fuels can readily carry fire, contributing to Florida's year-round incidence of fire. Postburn accumulation of fuel is rapid as most grasses, shrubs, and palmetto resprout within a week of the burn regardless of the season. In denser pine stands, needle drop from crown scorched trees can form a continuous litter fuel bed within weeks of a burn. This rapid accumulation of fuel allows for low-intensity reburns on some sites within a year. Sites without pine, or with open stands, may require up to 3 years of fuel accumulation before an effective burn can recur. In circumstances where fuels accumulate for a decade or more, the probability of lethal effects increase, either due to crown scorch and consumption, or through smoldering ground fires in the accumulated duff at the base of trees.

Although many of southern Florida's fuels are available throughout the year, a predictable dry period begins in the winter months and continues until the onset of the summer convectional storms. At times, this dry period is protracted leading to severe drought in March, April, and May. Because water levels have receded and fuel moistures are low, more fuels are exposed, increasing loadings on local areas and on a total area basis. These drought conditions may be exacerbated further if the winter includes widespread frosts or freezes, which top-kill many plant species of tropical origin.

Pinelands—There are two types of subtropical pinelands—flatwoods on sandy soils and pine rockland on limestone. Ground cover in the pine flatwoods consists of a diverse array of shrubs, grasses, forbs, and saw palmetto. The palm fronds, grasses, and pinestraw compose the fuel that carries the fires. Volatile compounds in the leaves of some of the grasses, particularly wiregrasses, the ericaceous coastalplain staggerbush and the aquifoliaceous gallberry and dahoon holly, wax myrtle, and the fronds of saw palmetto, allow for vigorous burning even at high dead and live fuel moistures. For example, the moisture of extinction of the NFFL Southern Rough Fuel Model (Albini 1976), which is representative of these pineland fuels, is a high 40 percent.

Pine rockland fuels comprise grasses, shrubs, and pine needles, which also are represented by the NFFL Southern Rough Fuel Model. In contrast to flatwoods, the shrub layer is composed primarily of species of West Indian origin including white bully, varnish leaf, cocoplum, and myrsine. Some of these shrubs will reach tree stature in the absence of fire. Of about 100 shrub species found in the pine rocklands, only seven come from the pine flatwoods flora (Snyder and others 1990). The herbaceous layer is a diverse mixture of herbs of both tropical and temperate origin, with a high percentage of endemic species.

Postfire Plant Communities

Pine Flatwoods and Pine Rocklands

Vegetation Dynamics—Fire is not a succession-initiating process because the pinelands are fire-maintained vegetation types. Fire is as vital as rainfall in maintaining the vegetation. Postfire species composition is virtually identical to the prefire vegetation composition. Some mortality of the overstory pine may occur in any or all age classes; the soil surface may be exposed, but only for a few weeks; the fuel biomass is reduced and nutrients are released. Released nutrients coupled with exposed soil interact to stimulate flowering in a number of the ground cover species. Some species, particularly the grasses, will rarely if ever flower without fire. The season when a fire occurs can have a strong inluence on the flowering response of some species (Robbins and Myers 1992).

The pinelands of southern Florida fall into two broad categories based on soil substrate and composition of the ground cover: subtropical pine flatwoods and subtropical pine rocklands. Pine flatwoods occur on flat, poorly drained acid sands that were deposited on ancient marine terraces. Across Florida, pine flatwoods compose the most extensive ecosystem type and form the fuel matrix in which many other vegetation types are embedded. Fires originating in the flatwoods have strongly influenced the structure, composition, and juxtaposition of other fire-maintained and fire-influenced habitats, particularly cypress swamps, bays, marshes, and hammocks. See chapter 4 for additional discussion of pine flatwoods and other southern pine forest types.

Similar fire-maintained pine forests and savannas that occur on poorly drained acid sand substrates are found in western Cuba and on the Isle of Pines south of Cuba (Borhidi 1996). The pine species in Cuba are Caribbean pine and *Pinus tropicalis,* which appear to form vegetation complexes maintained by fire regimes identical to the regimes of slash pine and longleaf pine flatwoods, respectively. For example, like longleaf pine, *Pinus tropicalis* has a definite grass stage, increasing the probability that young individuals can survive frequent fires.

The pine rocklands, in contrast to the acid sands of the flatwoods, occur on alkaline limestone bedrock that forms a ridge running from north of Miami south into Everglades National Park. Pine rocklands occur elsewhere in southern Florida on outcrops of limestone, particularly in the Big Cypress National Preserve and on some of the Lower Florida Keys: Big Pine, Little Pine, No Name, Cudjoe, and Sugarloaf Keys. In some areas of the pine rocklands, exposed rock makes up 70 percent or more of the surface (Snyder and others 1990) (fig. 7-2). Although somewhat elevated from surrounding wetlands, the water table in the pine rocklands, like in the pine flatwoods, is at or near the surface during the wet season.

The pine species in both subtropical flatwoods and rocklands is the southern variety of slash pine (var. *densa*), which might be somewhat more fire tolerant than its northern Florida counterpart (var. *elliottii*). South Florida slash pine is described as having a fire-tolerant grass stage, similar to longleaf pine, that provides young individuals some protection from low-intensity surface fires (Little and Dorman 1954). The trait, however, is not nearly as pronounced as in longleaf pine. Seedlings of south Florida slash pine over a year old do develop a thicker stem and bark than the northern variety. This may offer south Florida slash pine relatively greater protection, but they possess nothing like the true grass stage of longleaf pine. The other pine species typical of Florida's flatwoods, longleaf pine and pond pine, do not extend into the subtropical zone.

As a mature tree, south Florida slash pine has thick protective bark that insulates the tree and high open branches that facilitate heat dispersal. Portions of the outer layer of the platelike bark also peel off as they are heated by a passing flame front, serving to dissipate heat. Like all of the fire-tolerant southern pines, south Florida slash pine continuously flushes new needles throughout the growing season. This gives trees the capacity to survive 100 percent crown scorch from burns occurring during the growing season, provided the buds are not killed. In contrast, equivalent levels of scorching from fires during the dormant season are more likely to lead to death of the tree, either directly or through stress-induced beetle infestation.

Figure 7-2—Pine rockland vegetation on Big Pine Key, Key Deer National Wildlife Refuge, Florida, consists of an overstory of south Florida slash pine, a diverse underwood of tropical hardwoods and palms, and a ground cover of grasses and forbs. Photo by Ronald Myers.

Other than the marked floristic differences between pine flatwoods and pine rocklands, they are quite similar: an open overstory of pines and a continuous ground cover of herbaceous and low shrubby fuels. Fire behavior and the historical range of variability of their fire regimes are nearly identical: frequent (every 1 to 5 years), low-intensity surface fire that can occur at any time of the year. Large severe fires are associated with predictable drought, primarily in March, April, May, and June, but can also occur at other times of the year. Smaller, and perhaps patchier, fires occur at more humid and flooded times of the year. Locally intense fires may be associated with fuels accumulated from tropical storm damage.

In general, the majority of the larger pines survive the fires, with mortality occurring in clumps possibly associated with areas of high fuel accumulation or greater intensity generated by interacting flaming fronts. Intensity being equal, pine mortality is greatest after fall and winter burns. Likewise, seedlings and saplings survive fires in isolated clumps that may be associated with gaps created when adult trees were killed by previous fires.

In spite of similar fuel characteristics and dominance by the same pine species, the developmental trajectories of pine flatwoods and pine rocklands in the absence of fire are markedly different. On better-drained sites, long-unburned flatwoods develop into evergreen hardwood hammocks dominated by oaks, particularly live oak. On poorly drained sites they develop into evergreen bays (red bay, sweet bay, loblolly bay, and other genera). The changes are relatively slow because propagules must arrive from pockets of hammock or bay forests persisting in fire shadows, drains, or hollows. In contrast, pine rocklands, in the absence of fire, take on the character of tropical hardwood forests (hammocks) within a few fire-free decades as the species in the extant shrub layer reach tree stature, and as other tropical hardwoods invade from tropical hammocks scattered through the rocklands. Once a significant midstory of hardwood vegetation forms in both pine flatwoods and rocklands, a fire can become lethal and severely damage the pine overstory. In such cases, postfire recovery of the pines depends on seed sources, environmental conditions for regeneration, and subsequent return of frequent fires. In the continued absence of fire, such a site rapidly becomes a dense hardwood forest dominated by either temperate or tropical species.

Unlike many Western forests maintained by understory fire regimes, southern Florida's pinelands did not experience broadscale changes due to 20th century fire suppression efforts. The loss of flatwoods and rocklands habitats has been largely due to urban development and associated fragmentation rather than successional changes brought on by successful fire suppression. Although fire prevention efforts and suppression activities were implemented in southern Florida, woodsburning has a long tradition throughout Florida and the Southeastern United States. Where fuels are uninterrupted across broad landscapes, fires are common. Fires are set by ranchers, hunters, arsonists, and by accident. The exclusion of fire has been more pronounced in fragmented, developed landscapes along the coasts; in relatively small parks, preserves, and refuges; and in areas where suppression equipment had ready access.

Management Considerations—Both the pine flatwoods and pine rocklands in southern Florida have been impacted by the interaction of fire and the spread of non-native species. Remnant rocklands that burn infrequently have been invaded by Brazilian pepper, a subtropical shrub that, once established, will persist in the shrub layer even if fire is reintroduced. Many pine rocklands are also threatened by Burma reed, an exotic grass whose spread is facilitated by frequent burning. The acid sandy soils of wetter pine flatwoods are susceptible to invasion and complete dominance by melaleuca. Melaleuca invasion can lead to a mixed fire regime consisting of some understory burns and some crown fires that are nonlethal to melaleuca but lethal to the pines. Both pineland types are threatened by cogongrass, a pantropical grass known for its close association with fire. In stands of pine that have escaped fire for several cycles and in other woodland vegetation, Old World climbing fern is becoming a problem by creating ladder fuels that carry fire into tree canopies. Fires in the pinelands that normally would stop at the margins of flooded hardwood and cypress swamps can burn into these vegetation types when their canopies are covered with the fern. Burning mats of the light-weight fern break free and are kited away by convection columns, igniting spot fires well downwind from the main fire.

Because of a long history of burning, first indiscriminate then prescribed, fire has been much more readily accepted as an essential land management activity in Florida than it has been elsewhere. Florida law allows the State to conduct hazard reduction burns on unoccupied private wildlands, and it recognizes prescribed burning as a right of the landowner. Prescribed hazard reduction burns and ecological management burns began in the Everglades in the 1960s. Today, the National Park Service's Big Cypress National Preserve burns more acreage annually than any other National Park Service unit in the United States, and more than any other publicly administered wildland in Florida.

Failure to burn either pineland type at frequent intervals (2 to 7 years) leads to rapid fuel buildup, changes in the vegetation structure, changes in species composition, and eventual habitat loss. Selection of an appropriate fire intensity depends on ecological and management objectives. Burns can be conducted at any time of the year. With fires of equal intensity, the pines are more susceptible to stress from scorching, other damage, and death from fall burns. Dry season burns are more effective in removing duff and exposing the mineral substrate, but in the pine rocklands, deep burns may occur in organic matter that has accumulated within cracks in the rock. These may result in root damage and death of the pines, but also they may effectively reduce the density of shrubs on long unburned sites.

Many of the understory species respond favorably to growing season burns, particularly when occurring at the transition between dry and wet seasons. Generally, managers favor winter burns for fuel reduction and growing season burns for ecosystem maintenance.

Pondcypress Wetlands

Vegetation Dynamics—Cypress-dominated vegetation types in southern Florida cover roughly 800 square miles (2,000 sq. km) (Wade and others 1980). Two variants of cypress occur in southern Florida: baldcypresss and pondcypress (var. *nutans*). Baldcypress grows in floodplain forests, around shores of large lakes, and in the interior of large cypress strands (broad vegetated drainage depressions), all of which burn at intervals of centuries. Pondcyress grows in frequently burned savannas, shallow depression ponds called cypress domes or cypress heads, and on the periphery of cypress strands (fig. 7-3).

Although the role of fire in the dynamics of cypress-dominated wetlands is poorly understood (Ewel 1995), it is known that nonlethal, understory fire regimes prevail in the pondcypress savannas and woodlands rather than the baldcypress forests. The state-transition model (fig. 7-4) illustrates that fire regimes of various types play a role in the dynamics of cypress-dominated wetlands, with understory fire regimes responsible for monospecific pondcypress forests and savannas, including what are known locally as cypress domes, strands, and dwarf forests (Wade and others 1980).

Cypress domes occur in circular depressions or ponds; cypress strands form in elongated shallow drainage channels. Dwarf cypress forests occur on shallow soils over limestone bedrock and may include domes, strands, and savannas. In both domes and strands, trees increase in stature from the periphery to the center or midline of the depression, giving the vegetation a domed or ridged appearance.

Numerous theories involving the interplay of soil depth, hydroperiod, water depth, and fire have been postulated to explain the cause of the domed and ridged appearance. Fires are generally more frequent around or along the periphery of domes and strands, and one study concluded that this "differential marginal fire theory" accounts for the doming and ridging (McJunkin 1977). As one enters a dome or strand, tree height increases, crown closure becomes more complete, and fine herbaceous fuels are either no longer continuous or are submerged. This, coupled with longer hydroperiods toward the interior, limits the influence of fires to those times when protracted drought coincides with fires originating in the surrounding vegetation, such as pine flatwoods, wet prairie, or cypress savanna or woodland. The longer interval between

Figure 7-3—Cypress-dominated landscape with cypress domes, woodlands, and savannas, Big Cypress National Preserve, Florida. Photo by Ronald Myers.

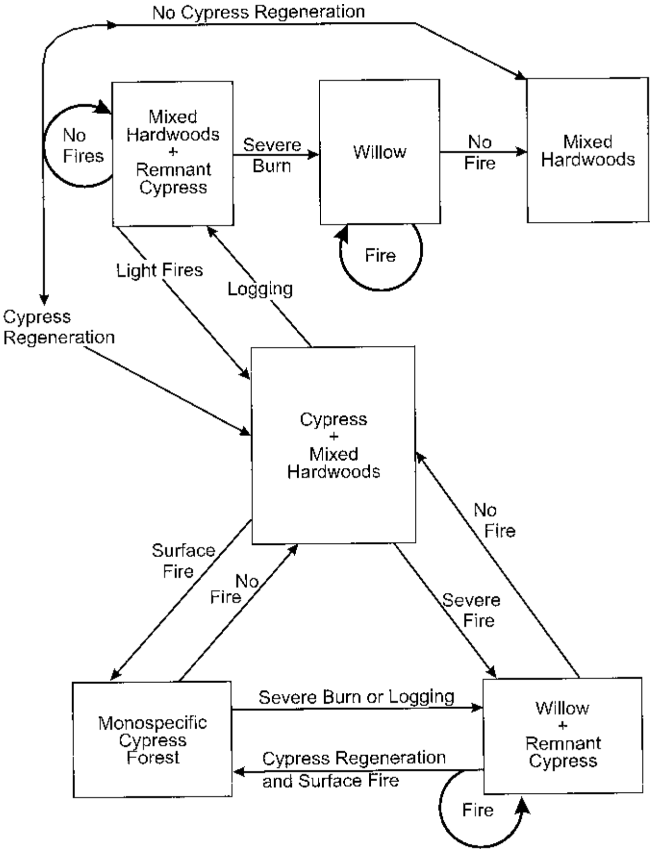

Figure 7-4—Generalized succession scheme for showing the role of different fire regimes in cypress wetlands and related vegetation types (Gunderson 1984).

fires coupled with increased flooding allows the accumulation of organic soils.

During those rare instances when fire penetrates to the interior of domes and strands, the organic matter may be consumed and the fires become lethal to the cypress causing stand-replacement, even though there is no flaming front. Such fires usually recur on the order of centuries. When they occur more frequently, open water or herbaceous marsh vegetation is present rather than forest. Some domes have an open water center giving them an appearance of a doughnut from the air. Similarly, the midline of strands may possess a series of ponds. Rare, severe fire events that consume organic soil probably play a role in the creation of these ponds.

Dwarf cypress forests and savannas occur on shallow sandy or muck soils overlying limestone bedrock. Density of stunted cypress trees varies from a few to nearly 700 trees per acre, with a ground cover of grasses, sedges, and forbs serving as the principal carrier of the surface fires (Wade and others 1980). Dwarf cypress forests form a continuum grading into cypress strands and domes as soils deepen, hydroperiods lengthen, and maximum water depth increases. Productivity and hydroperiod mediate the accumulation and density of the herbaceous fuel bed, which to a large degree controls the recurrence of fires. Return intervals may range from 5 to 20 years in dwarf cypress forests.

Little is known about the fire tolerance of pondcypress, but its bark has good insulation properties relative to other native wetland species, and the trees seem quite tolerant of low-intensity surface fires (Hare 1965). Limited studies have shown that pondcypress is much more likely to survive surface fires than competing hardwoods and shrubs (Ewel and Mitsch 1978), and years of observing wildfires and prescribed burning in the Big Cypress National Preserve attest to pondcypress' ability to withstand fire. Some trees, particularly those in dwarf cypress forests, possess the ability to resprout from their swollen buttresses if top-killed. Observations also point to the importance of fire in checking the encroachment of swamp hardwoods and permitting the establishment of cypress seedlings (Gunderson 1977).

After a long fire-free interval, pondcypress domes and strands develop a hardwood understory. In the continued absence of fire, the pondcypress is replaced by a hardwood-dominated swamp or bay. Once the shrub layer forms a continuous fuel bed under the cypress, conditions for lethal fires exist with the potential to eliminate the cypress overstory. It has been postulated that many of the bayheads (clumps of evergreen hardwood shrubs and trees) in the Everglades represent former cypress domes that were burned out during the droughts of the late 1930s (Wade and others 1980). If such fires include ground fuels, all woody vegetation would be removed; depending on the depth of the burn this would create open water, marsh, or willow, or permit re-establishment of cypress, provided there was a seed source. Without a ground fire, the hardwoods would resprout creating either a deciduous hardwood swamp dominated by red maple and Carolina ash, or a bay dominated by red bay, sweetbay, and loblolly-bay.

Shallower pondcypress wetlands are susceptible to invasion by melaleuca following fire. There are a number of examples where a single high-intensity fire in pondcypress that included scattered mature melaleuca led to the replacement of pondcypress by melaleuca.

Management Considerations—Most prescribed burning in cypress is limited to the savannas and woodlands that have herbaceous fuel beds. This effectively reduces the hazard of fire getting into deeper domes and strands during droughts. Burning during exceedingly dry periods is usually avoided to prevent excessive damage to trees on organic soils and to avoid extended smoke problems. However, preventing or avoiding "muck" fires will eventually lead to the loss of deeper water cypress forests and their habitats as they succeed to hardwood swamp forests.

Cabbage Palmetto Savannas and Forests

Cabbage palmetto-dominated savannas and forests are most prevalent on soils having a calcareous layer (shell, limestone) near the soil surface. The role of fire in creating and maintaining palm-dominated communities is poorly understood, but cabbage palmetto is tolerant of fire (Myers 1990a). Its single terminal bud, even as a seedling, is well protected from fire. The absence of a fire-sensitive surficial cambial layer in monocot stems allows palms to survive considerable charring, even some consumption of the stem. The palms appear able to withstand higher intensity fires than slash pine. Cabbage palmetto is frequently a codominant with live oak in hardwood hammocks. Where cabbage palmetto is the sole dominant, it may indicate that either the pine forest or live oak forest had been eliminated in a severe fire or series of fires, or perhaps that the pines had been harvested. In the absence of fire, oaks invade palm forests, eventually becoming the dominants.

Live Oak Forests

Live oak is generally considered a non-fire type or climax of pine and palmlands in the absence of fire. Live oak forests, however, do burn. Saplings and small trees have the ability to both resprout and to send runners forming clones. Large trees are protected by a relatively thick bark. Live oak litter produces a particularly compact fuel that limits flame length. It rarely burns without the momentum of a heading fire. The litter also holds moisture making it fire resistant except during periods of extreme drought. When it burns, flame lengths are short (<2 feet). Intense fire with some crowning may occur where the oak forest has expanded over a continuous saw palmetto fuel bed such as in long unburned pine flatwoods or dry palmetto prairies (Huffman and Blanchard 1991). Live oak forests are commonly embedded in flatwoods, prairies, and palmlands on sites somewhat protected from the frequent fires that normally spread through the matrix vegetation. For example, oak forests are frequently found on the leeward side of wetlands, ponds, lakes, and wet drainages where they would rarely be exposed to direct heading fires. The importance of these “fire shadows” was first pointed out by Harper (1911). Live oak forests that have gone for extended periods without being exposed to fire may exhibit a higher diversity of broadleaved trees and shrubs including other oaks and red maple. At more southerly and coastal locations in the Florida peninsula, this component takes on a more tropical character. Where oaks have expanded into other vegetation types due to fire exclusion, repeated growing season burning, undertaken before the summer rainy season, is most effective in killing oaks and restoring historic landscape patterns (Huffman and Blanchard 1991).

Mixed Fire Regimes

Major Vegetation Types

Melaleuca forests, woodlands, and savannas are an unusual wooded vegetation type dominated by an introduced Australian tree, melaleuca. It occurs in both Florida and Hawaii. In Florida, melaleuca is an aggressive invader of fire-prone wet prairies, shallow marshes, wet pine flatwoods, and cypress swamps (Meskimen 1962; Myers 1983); in Hawaii, it is a component of upland plantations composed of a variety of other introduced species. Although the Hawaiian melaleuca plantations are subject to anthropogenic ignitions creating significant fire control problems, it is in Florida where melaleuca seems most at home and appears to be gaining control of the fire environment (fig. 7-5). By the mid-1990s nearly 500,000 acres (202,400 ha) were infested with melaleuca, and of that, 40,000 to 50,000 acres were considered “pure” monospecific forests (Ferriter 1999).

Fire Regime Characteristics and Vegetation Response

The fire regime mediating Florida’s melaleuca forests varies from one characterized by low-intensity surface fires in savannas, with some torching of individual trees, to high-intensity crowning fires in denser stands. Regardless of the fire intensity, little or no mortality occurs to any of the trees beyond the seedling stage (Myers and Belles 1995). The bole of melaleuca is protected by thick spongy bark. The outer bark layers carry fire into the crown, consuming branches and leaves and releasing seed from serotinous capsules. New branches sprout from the bark-protected larger stems. Fire in stands of melaleuca containing any mature capsule-laden individuals leads to the spread of the melaleuca forest into susceptible habitats nearby, resulting in a shift from a fire regime controlled by surface fuels to one dominated by aerial fuels. This is a fire regime heretofore unknown in the Florida environment and is likely to result in significant changes to wetland habitats, especially the species composition. Once melaleuca gets a foothold in a pine or cypress dominated habitat, the shift from a low-intensity to high-intensity fire regime results in the mortality of the native pine and cypress and subsequent conversion to melaleuca. This regime of varying fire intensity makes it difficult to categorize the melaleuca fire regime in table 7-1. Understory burns occur in melaleuca savannas. In mixed stands of

Figure 7-5—Torching melaleuca during prescribed research burn in a melaleuca-invested wet prairie, Big Cypress National Preserve, Florida. Photo by Holly Belles.

melaleuca and cypress or pine, the fires are lethal to cypress or pine but not to melaleuca. In pure melaleuca forest, high-intensity crowning fires are not lethal to the main stem of the trees. The combination of limited stem mortality and high-intensity fire is unusual in North American ecosystems. Placing melaleuca forest in the mixed fire regime is a compromise between low mortality and high intensity.

Management Considerations—Controlling the spread of melaleuca is a major concern and challenge on public lands in southern Florida. Successful control involves a strategy of first targeting outlying individuals and populations, then treating mature trees individually with herbicide, followed by prescribed burning after released seeds have germinated. Seedlings and saplings less than 3 feet tall are generally killed in these burns. Larger individuals will resprout (Myers and Belles 1995).

Stand-Replacement Fire Regimes

Major Vegetation Types

Vegetation types ranging from hardwood forests to grasslands in subtropical Florida, Hawaii, Puerto Rico, and the Virgin Islands are characterized by stand-replacement fire regimes.

Grassland vegetation types are considered to be stand-replacement regimes because the dominant aboveground vegetation is burned and replaced. When grasslands possess an overstory of trees—that is, a savanna type—the fire regime becomes an understory fire regime if repeated fires are primarily nonlethal to the overstory (see chapter 1). In Florida, the prairie types, known as dry and wet prairies, have ground cover and surface fuels that are identical to that found in pine flatwoods or pondcypress savanna without the overstory trees. The lack of a tree overstory, which determines whether a fire regime type is stand-replacement or understory, may be related to a history of frequent (nearly annual) burning or other aspects of fire and land use. Whether open pine, cypress or prairie, the fuel characteristics, burn conditions, and fire behavior are nearly identical, and the ground cover vegetation responses are the same.

In many tropical environments, including Hawaii, Puerto Rico, and the Virgin Islands, there is an interplay between lethal and nonlethal stand-replacement fire regimes, one fueled by grasses, the other by forest fuels. Fires originating in agricultural fuels, usually non-native range grasses, burn up to and penetrate forest edges killing trees and allowing grasses to encroach at the expense of forest. Once within the forest, these fires create feedbacks in future fire susceptibility, fuel loading, and fire intensity that favor grass fuels—a process, which if left unchecked, has the potential to transform large areas of tropical forest into shrubland, savanna, or grassland (D'Antonio and Vitousek 1992; Koonce and Conzales-Caban 1990).

The stand-replacement fire regime type also characterizes the fires that occasionally occur in mangrove vegetation, along with frequent fires in Florida's salt and freshwater marshlands, and fires that occur in a number of successional stage communities between marshes and swamp forest. These successional communities are dominated by willows, ashes, and bays.

Grasslands and herbaceous wetlands are common fire-maintained vegetation types in both Florida and Hawaii, with notable differences between the two locations. In Hawaii virtually all of the grasslands are dominated by introduced exotics, such as thatching grass, Natal redtop, molasses grass, broomsedge, fountain grass (fig. 7-6), and Columbian bluestem. These fuels have created altered fire regimes to which the

Figure 7-6—The introduced fountain grass retains dead leaves due to its bunchy structure and is highly flammable. It is an aggressive invader that readily replaces native plants especially where it carries fire into less flammable native vegetation. It invades lava beds creating continuous fuels where fuel breaks formerly existed. Photo by Jim Brown.

native Hawaiian flora are not adapted (Smith and Tunison 1992). In some cases, the nonnative grasses form the understory of introduced pine and eucalyptus plantations. In Florida a few introduced grasses are causing problems in pinelands, but Florida's extensive wet and dry prairies are native fire-maintained ecosystems.

In subtropical Florida, most native herbaceous vegetation types are associated with wetlands. They include salt marsh, sawgrass marsh, wet prairie, and miscellaneous broadleaved herbaceous marshes. The vegetation type known as dry prairie or palmetto prairie represents the ground cover vegetation of pine flatwoods or palm savanna without the trees. It is more common in the transition zone between temperate and subtropical vegetation in Florida and is on the periphery of the region discussed in this chapter.

Fire Regime Characteristics and Vegetation Response

Florida Freshwater Marshes and Wet Prairie—Freshwater marsh and wet prairie vegetation types include dense and sparse sawgrass marshes, wet prairie, marl prairie, spikerush flag marsh, beakrush flag marsh, and American white waterlily marsh (Kushlan 1990). Fire in these marsh types affects species composition and may limit or reduce peat accumulation. The sharp demarcation between different marsh types frequently indicates the boundary of a past burn (Kushlan 1990). The fire regime and fire effects in these herbaceous wetlands result from the interplay of hydroperiod and fuels, which together determine whether the fires are lethal or nonlethal to the dominant species, and whether or not fire is a succession-initiating process. In most cases, the aboveground vegetation is consumed, but the fires are usually not lethal to the dominant species that make up the fuel. These species simply resprout from underground buds, tubers, or rhizomes.

Two conditions can create lethal fires and initiate a vegetation change (Herndon and others 1991). One is when fire coincides with severe drought and consumes some, or all, of the organic substrate, destroying root systems and underground regenerative organs of the dominant species. The second occurs when water levels rise faster than vegetative regrowth after a burn and the site remains flooded long enough to cause the death of the vegetation. In either case, vegetation is replaced primarily by species present in the seed bank.

Sawgrass, once covering several million acres, is the dominant marsh vegetation type in the Everglades. Dense sawgrass marsh occurs on organic soils, while sparse sawgrass marsh is found on marl soils. Fuel loadings in dense sawgrass are sufficient to allow fires to burn over standing water, and lightning ignitions are common. These wet season fires go out when they burn into other marsh types. During the dry season, fires may burn through sparse sawgrass stands and wet praires with low fuel loads by smoldering through a dry algal mat called periphyton, composed mostly of filamentous blue-green alge, that forms over the substrate surface when the marsh is flooded. During severe droughts, organic soils may be consumed in both dense sawgrass and in many of the deeper water marsh sites. Sawgrass marshes and wet prairies can burn every 3 to 5 years (Wade and others 1980). In the absence of fire, sawgrass will succeed through a willow stage to hardwood bay vegetation.

Wet prairies are the least flooded of Florida's marsh vegetation types. The dominant species are identical to those in pine flatwoods and cypress savannas. Soils vary from periphyton-derived marls to sands. Dominants include maidencane, cordgrass, beakrush, and hairawn muhly. The fire frequency is 2 to 5 years.

Flag marshes are named after herbaceous broad-leaved marsh species that have a flag appearance. They have a long hydroperiod and only burn during severe droughts. These fires may consume organic soils.

Except for the deepest water marshes, all types are susceptible to invasion by melaleuca. Dense sawgrass marshes and wet prairies are particularly susceptible.

Florida Salt Marsh and Mangrove—Although salt marshes occur in both the temperate zone and the tropics, fire mediates the tension zone between mangrove and salt marsh only in the tropics. In subtropical Florida, salt marsh is wedged between mangrove on the seaward side and freshwater marsh on its inland edge. Freezing temperatures probably have some influence on the juxtaposition of mangrove and salt marsh, but fires originating either in the salt marsh or further inland in freshwater marsh, control the inland advance of mangrove. When intense fires are stopped by mangrove, the outer fringe of trees is killed and the marsh expands (Wade and others 1980). Under moderate burn conditions the mangrove acts as a firebreak. Similar fire dynamics probably occur between marshes and mangrove in Cuba and the Bahamas.

Florida's salt marshes are dominated by black rush, gulf cordgrass, sand cordgrass, and inland saltgrass, mixed with a number of species found in fresh water marshes, notably sawgrass and cattail. Because salt marsh is under tidal influence and relatively isolated from human activities, a large proportion of ignitions are lightning caused. Fires supported by high fuel loadings frequently burn over standing water. Fire behavior varies considerably depending on the dominant species contributing to the fuel. Cordgrass fuel loadings are frequently as high as 22 tons/acre (49 t/ha) with a fuel bed depth of 8 feet. Beakrush fuel loadings and fuel bed depths are half that amount. Saltgrass fuel loadings vary considerably, and the fuel bed depth is only 1 to 2 feet (Everglades NP FMP 1991).

Mangroves rarely burn, but they are influenced by fire in seasonal environments such as in Florida. Mangroves are a tropical and subtropical forest type growing in brackish to high salinity coastal sites that have weak wave action. Four species of mangrove are found in Florida: red mangrove, black mangrove, white mangrove, and buttonwood. Each tends to be indicative of different zones of salinity or tidal influence. Lightning may be an important factor in the structure and dynamics of mangroves by creating numerous circular holes of dead and dying trees that may develop into patches of more flammable herbaceous vegetation. Fire is responsible for checking the encroachment of mangrove into salt marsh, and it is not uncommon to find red or white mangrove scattered through long unburned fresh water marshes. They have even been observed in the understory of cypress domes.

Florida Coastal Prairies—Coastal prairies are closely related to salt marsh; they occur in southern Florida, the Bahamas, and Cuba. They are less frequently inundated than salt marshes, but are sometimes flooded with salt or brackish water. Species composition includes saltgrass, seaside tansy, and batis. Coastal prairies are maintained by a combination of fire and hurricanes (wind damage and storm surge). If fire is absent for several decades, coastal prairies develop into buttonwood forests (Craighead 1971). The extent of coastal prairies may be a function of past clearing for charcoal production followed by cattle grazing and associated frequent burning. Coastal prairies on the Zapata Peninsula in Cuba have been contracting at the expense of buttonwood since the area was made a national park and the burning associated with cattle operations was curtailed (Myers 1999). See chapter 4 for additional discussion of freshwater and salt marshes.

Florida Tropical Hardwood Forests—Hardwood forests in southern Florida are usually islands imbedded in a matrix of marsh, prairie, or savanna. Fires burning in tropical hardwood forests (called hammocks in Florida), hardwood swamps, and bays or bayheads likely originate in the more easily ignited matrix fuels. The forest islands usually serve as effective firebreaks with fires burning only at their periphery. Fires have the opportunity to enter these ecosystems during extreme droughts, where they may cause conversion to earlier successional stages or shifts to marsh vegetation. Depending on fuel and weather conditions, vegetation structure, and type of substrate, the fires may be low-intensity surface fires, ground fires that burn out organic soils, or crowning fires moving through dense low shrubs, palmettos, or trees. Many of the tropical hardwoods have the ability to resprout if top-killed, but fires are lethal if the organic substrate is consumed. It is not uncommon for fires burning through grass fuels to go out at night as humidity recovers, but will hold over in hardwood forests, igniting the grass fuels the next day.

Hawaiian Forests and Grasslands—Fire regimes in Hawaiian native forest were probably always stand-replacement. Lava flows are the most common cause of natural ignitions, but resulting fires probably have not been an important evolutionary force shaping the characteristics of the vegetation. Some tree species such as koa and a'ali'i show some tolerance to fire, but these may represent preadaptations from their noninsular evolutionary environment (Smith and Tunison 1992). Most of Hawaii's native vegetation is extremely sensitive to fire. Historically, lightning may have caused some ignitions; today, lightning ignitions may be more common due to the prevalence of exotic grass fuels. Human sources, however, are by far the most frequent cause of fires.

The litter fuels in native Hawaiian forests, which range from dry forest to montane rainforest, are generally not sufficient to carry fire. The woody vegetation itself is not flammable. The presence of flammable grasses is essential for fires to spread. The grass fuels tend to be available throughout the year. Fire may have played a natural role only in the seasonal montane environment where, in places, native grasses and shrubs form a continuous fuel bed (Mueller-Dombois 1981).

Vegetation Dynamics—Although the vegetation, and thus fire regimes, of Hawaii changed dramatically in the later part of this century, some changes actually began over 1,600 years ago when the Polynesians colonized the Islands. The Polynesians encountered vegetation that was not fire prone, but, like indigenous peoples elsewhere, they used fire to manipulate the vegetation to plant crops, to facilitate travel, and to stimulate native grasses used for thatch such as pili grass. Early Euro-American visitors to the Islands reported encountering open savannas and grasslands at lower elevations and observed Polynesian burning practices (Kirch 1982).

Fire frequency and fire size greatly increased in the late 1900s when non-native grasses were introduced and spread (fig. 7-6). The invasion of fire-prone alien grasses coincided with the removal of feral goats from areas like Hawaii Volcanoes National Park. The grasses provided the fuel that carried fire into nonfire-adapted vegetation leading to the death of the native vegetation and its replacement by the grasses. Many fires are confined to mesic and dry forest habitats; however, grasses are also encroaching into some rainforest habitats and subalpine ecosystems. The prevalence of fire in these areas has been increasing (Mueller-Dombosis 1973).

Management Considerations—Managing fires in Hawaii is problematic. Fires of all intensities, timing, and sources are destructive to the Islands' native ecosystems, and most fires should be aggressively suppressed even if they result from natural ignitions. Prescribed fire has potential as a tool to reduce alien grass fuels and create firebreaks to prevent fires from entering sensitive native vegetation. Evidence indicates that prescribed fire may have a limited application in the restoration of a'ali'i shrublands and koa forests or woodlands because fire stimulates resprouting in both of these species. Prescribed fire might be useful in managing habitat for the endangered Hawaiian goose or nene (*Nesochen sandwicensis)* (Smith and Tunison 1992).

Forests of Puerto Rico and the Virgin Islands—Unlike Cuba and Hispaniola, which have native fire-maintained pine and palm forests and savannas and fire-maintained herbaceous marshes and wet prairies, Puerto Rico and the U.S. Virgin Islands do not possess any significant fire-adapted native vegetation types. Native forest types of Puerto Rico and the Virgin Islands include mesic forests on the windward sides that grade into rainforest or montane cloud forest with increasing elevation. The leeward lowlands support dry forest types. Fires occur in the dry forests, but the sparse accumulation of litter fuels only supports low intensity surface fires that generally go out at night as the humidity rises. If the thin-barked trees are top-killed, many have the ability to resprout, probably from an adaptation to drought, not fire. Where disturbances have created grassy openings, higher intensity fires may cause significant damage to the forest cover particularly at the forest edge. Repeated burning favors the grasses at the expense of forest. Similar fire damage occurs on more mesic sites where agricultural fires may escape and encroach into the forest during dry periods. This pattern of burning is more prevalent and a greater problem in other tropical areas where slash and burn (swidden) agricultural practices are common.

Severe fires have occurred in hurricane-damaged tropical dry and seasonal forests on the Yucatan Peninsula in Mexico (Whigham and others 1991). The potential for similar damage and subsequent fires exists in all of the tropical regions covered in this chapter.

Notes

Kevin C. Ryan

Chapter 8: Global Change and Wildland Fire

Global change, the combined effect of human activity on atmospheric and landscape processes (Vitousek 1994), affects all aspects of fire management. Scientists have documented changes in the global carbon cycle due to increases in atmospheric carbon dioxide (CO_2), changes in biogeochemical cycling due to increased nutrient deposition (for example, nitrogen), and changes in land use and cover. These changes are expected to continue for the foreseeable future (IPCC 1996a,b).

Changes in the global atmospheric chemistry are attributed to biomass burning and industrial processes. These alterations in the chemical makeup of the atmosphere are predicted to have a significant impact on biogeochemical processes and Earth's radiation balance, the so called "greenhouse effect." These changes in the chemical composition of the atmosphere and Earth's energy balance can be expected to modify precipitation, temperature, humidity, and vegetation development—all affecting fire management. In addition, historic changes in patterns of land use—roads, subdivisions, timber harvesting, farming, and ranching—have altered vegetation and fuels, affecting the potential ignition, spread, and severity of wildland fires. Continued migration of people into wildlands further complicates prescribed fire management and wildfire suppression.

Because of the complex interactions of all these processes, it is difficult to make definitive estimates about the rate and, in some cases, even the direction of change. However, given current knowledge, the anticipated changes can be expected to increase the pressure on fire management organizations. This chapter examines the complexity of global change and the possible influences on vegetation and fire management.

Changes Over Time

Vegetation and fire regimes have been in a constant state of flux throughout geologic time. Climate has changed throughout the millennia (Bradley 1999). New species have evolved as others became extinct, with climate and herbivory as the dominant influences. In the current geological epoch, the Holocene (0 to 10,000 years before the present), activities of humans have increasingly influenced vegetation and fire. Numerous climate fluctuations during this time include the Medieval Warm Period, AD 900 to 1350, and the Little Ice Age, 1450 to1900. Average temperatures have varied by as much as 5.4 °F (3 °C) over periods as short as a few decades (Bradley 1999; Mann and others 1999). Human societies (Ahlstrom and others 1995; Lipe 1995) and the prevalence of fire have been significantly affected by these changes (Bonnicksen 1999; Clark 1990; Swetnam 1993).

Prior to the arrival of Europeans in North America, native people routinely used fire to drive game animals and manage vegetation near encampments (Barrett and Arno 1982; Bonnicksen 1999; Boyd 1999; Clark and Royall 1995, 1996; Pyne 1982). In some regions Native Americans developed large agrarian communities where vegetation was extensively altered. Although the degree to which fire was used to initiate and maintain agriculture is uncertain, agriculture and harvesting of biomass for energy did lead to substantial change in fire regimes and vegetation in some areas. Across much of the landscape, lightning was the primary source of ignition. And fire spread was only hindered by the availability of fuels, weather, and natural barriers. Forest and rangeland sites developed with characteristic fire regimes and vegetation. Landscapes developed with a characteristic mosaic of stands of varying age, structure, and species composition. Fauna developed life cycle and behavior patterns tuned to these landscape patterns. While early Euro-American settlers may have seen many desirable features in existing patterns of vegetation, these features were not static but represented only a point in time in the development of North American vegetation (Betancourt and others 1990; Bradley 1999; Bonnicksen 1999; Delcourt and Delcourt 1987; Prentice and others 1991; Woolfenden 1996).

Since Euro-American settlement across the continent, fragmentation of the landscape resulting from agriculture, mining, and urbanization (Bahre 1991; Baker 1992; Veblen and Lorenz 1991) has significantly altered the fire potential of many ecosystems. This transformation of the vegetation prevented fires that formerly swept across prairies and steppes into adjacent forests (Gruell 1985). Domestic livestock have reduced the availability of fine fuels for fire spread. Grazing and fire exclusion together have led to the replacement of grasslands by shrublands in some areas (Wright and Bailey 1982). The introduction of exotic species has led to substantial changes in the species composition and fire potential of many ecosystems, particularly in arid and semiarid areas (Billings 1990). Timber harvesting has led to unnatural patterns of vegetation, modified fuel beds, and altered fire severity.

One of the most significant changes in land use in the 1900s was the suppression of wildfires. Fire suppression has led to changes in species composition and vegetation structure, and it has led to a significant buildup of fuels (Arno and Brown 1989) and increased forest health problems (Mutch 1994). The shift from ranching to ranchettes and urban encroachment on wildlands (Riebsame and others 1997) is also leading to a buildup of fuels. The result is that fires, though apparently less frequent than in the 1800s, are now often larger and more severe (Agee 1993; DeBano and others 1998; Sampson 1997) than formerly. When fires do occur they can result in serious threat to life and property. Larger, more severe fires also have greater potential to adversely affect postfire vegetation composition and structure, as well as soils and water, cultural resources, and air quality, as described in other volumes of the Rainbow Series ("Effects of Fire on Soil and Water," "Effects of Fire on Cultural Resources and Archaeology," and "Effects of Fire on Air").

Emissions from industrial processes, burning of fossil fuels, and slash-and-burn agriculture in the tropics have increased the concentrations of greenhouse gasses (GHG) in the atmosphere particularly in recent decades (Tett and others 1999). Chief among these is carbon dioxide (CO_2), but water vapor (H_2O), ozone (O_3), methane, nitrous oxides (NO_x), and various chlorofluorocarbons (CFCs) are also important (IPCC 1996a). Carbon dioxide has risen from approximately 270 ppm in the preindustrial atmosphere to around 365 ppm. While there is not universal agreement that increases in GHG have caused temperatures to rise, the observed 20th century warming reversed a millennial cooling trend. The 1990s were the hottest decade in the millennium, and 1998 was the hottest year (Mann and others 1999). There is growing scientific consensus that we are experiencing a greenhouse warming effect (IPCC 1996a).

Changes in the atmosphere due to GHG are expected to alter global weather patterns and significantly change regional climate. Due to the complexity of general circulation models used to predict climate change, there is much uncertainty as to the magnitude of effects of increased GHG on regional climate. Globally, average annual temperatures are expected to increase on the order of 2 to 8 °F (1.1 to 4.5 °C), depending on location. At this time estimates of regional climate changes are more tentative than estimates of global change, but increases are expected to be greater at high latitudes, in mid-continent regions, and in fall and winter (IPCC 1996a). The growing season may be extended by 1 to 2 months depending on latitude and altitude. Average annual precipitation may increase as much as 20 percent, but little summer rain is expected in much of North America's interior. Maritime climates may be wetter than today, but it is uncertain if increases in precipitation will be adequate to compensate for higher temperatures (Franklin and others 1991). Because continents are expected to warm up more rapidly than oceans, the interiors of the continents are expected to experience major drought by the middle of the next century (IPCC 1996a; Rind and others 1990).

Climate is generally defined as the 30-year average weather for a location. While temperature change (that is, global warming) is the major focus, the atmosphere, hydrosphere (water), cryosphere (ice), and biosphere (flora and fauna) are mechanistically coupled, and their interactions affect the relations between climate and wildland fire and between vegetation and fire regimes (K. Ryan 1991) (fig. 8-1). If the expected global warming occurs, the increase in CO_2 and changes in precipitation will alter growth and competitive interactions of plant communities. This will result in changes in ecosystem structure and species composition.

Climate change, therefore, will directly affect the frequency and severity of weather favorable to fire spread. Changes in plant communities will affect fire regimes indirectly by altering the physical and chemical properties of fuels. Fire is a major source of mortality in many communities, but in turn it creates gaps for new species. As a result, changes in fire regimes will modify the rate at which communities respond to climatic change.

Climate, Weather, and Fire Interactions

Regional fire potential is strongly coupled to regional climate. This connection between weather and fire potential is a fundamental part of all models and training in fire behavior, so clearly any significant climate change will affect the frequency and severity of conditions suitable for the ignition and spread of fires. In addition, each type of fuel has characteristic physical and chemical properties that affect flammability, and these properties vary with climate and weather. On the other hand, predicted temperature increases are not expected to have large direct effects on fire potential (fig. 8-2) but are expected to be correlated with a number of other changes that will affect fire potential (table 8-1). One of these changes is drying rate. Each type of fuel has its own drying rate, and its moisture content varies according to site moisture history. Decreased relative humidity can be expected to result in lower fine fuel moisture (fig. 8-2) and more rapid fire spread, whereas higher humidity can be expected to result in higher fuel moisture and less rapid spread. Therefore, if humidity patterns are altered, the moisture and combustion properties of fuels will change correspondingly.

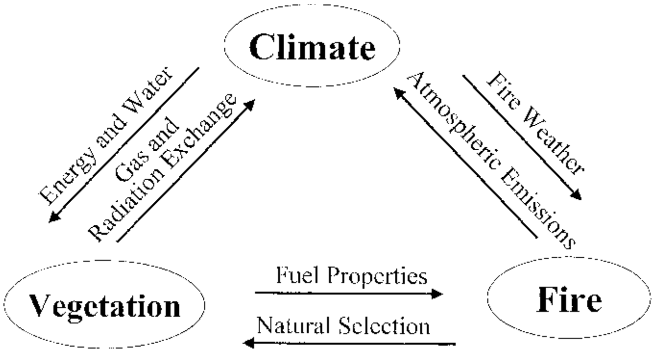

Figure 8-1—Climate, vegetation, and fire are dynamically coupled. Any change in one factor will lead to changes in the other two.

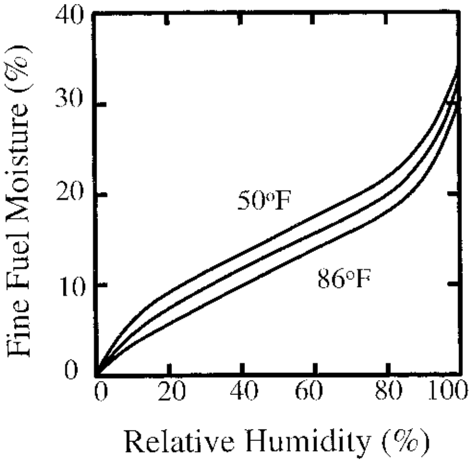

Figure 8-2—Fine fuel equilibrium moisture content as a function of relative humidity and temperature.

Increases in the frequency and severity of drought and an extension of the length of the fire season (Rind and others 1990; Wotton and Flannigan 1993) will result in more severe fires and increased consumption of long time-lag fuels (logs, duff, and organic soils). Wind is also a major factor in determining how fast fires spread. If the frequency of high winds changes, the potential for large fires will also change. Lightning is a major source of ignitions, and increased temperature, precipitation, and evaporation will change thunderstorm patterns. As a result, the frequency of lightning-caused fires is expected to increase some 30 to 70 percent depending on location (Price and Rind 1994). Further, much of the increase is expected to come during periods of moisture deficit.

Based therefore on climate projections, increases are expected in the length of the fire season (Wotton and Flannigan 1993), the frequency of lightning fires (Price and Rind 1994), the frequency of drought (Rind and others 1990), and area burned (Flannigan and Van Wagner 1991; Stocks 1993) in much of Canada and the United States; but some regions are expected to experience a decrease in fire activity (Bergeron and Flannigan 1995). In addition to these direct influences

Table 8-1—Fire interactions with the climate/weather system.

Climate/weather influences on fire	Fire influences on climate/weather
Relative humidity	**Carbon dioxide** (CO_2)
Wind (speed, persistence, extremes)	**Carbon monoxide** (CO)
Drought (frequency, persistence)	**Methane** (CH_4)
Length of fire season	**Water vapor** (H_2O)
Lightning (dry vs. wet)	**Particulates** (Pm 2.5, Pm 10)
Dry cold fronts (frequency)	**Nitrous oxides** (NO_x)
Blocking high pressure (persistence)	**Ammonium** (NH_4)
	Trace hydrocarbons
	Trace gasses (including VOC)

of weather on fire, storms are predicted to be more severe in the altered climate (IPCC 1996a). If so, increased wind damage to forests could greatly increase available fuels.

Complete combustion of biomass results in the production of CO_2 and H_2O, but combustion is rarely complete, and a variety of other chemical species are produced (table 8-1) (Crutzen and Goldammer 1993; Goode and others 1999; Hao and others 1996; Levine 1996; Ward and Hardy 1991). Globally, biomass burning is a major source of several chemical species in the atmosphere. Many of the compounds released by burning are greenhouse gasses. Particulate matter (for example, Pm 2.5, Pm 10) can produce a local short-term cooling effect by reducing solar heating. Particulates can also result in reduced precipitation (Rosenfeld 1999).

Biomass burning contributes to the overall problem society faces in managing greenhouse gasses and providing for clean air. See the volume "Effects of Fire on Air" in the Rainbow series for a state-of-the-knowledge review of the fire management issues associated with fire's impact on local and regional atmospheric conditions.

Climate and Vegetation Interactions

Climate is considered the principal determinant of vegetation distribution throughout the world (COHMAP Members 1988; Neilson 1995; Woodward 1987). Solar radiation, temperature, humidity, precipitation, and wind all affect the physiological ecology of plants (Bazzaz 1996), thereby affecting their ability to complete life cycles and sustain populations (table 8-2). Vegetation, therefore, is governed by the cumulative history of climate, vegetation, and disturbance processes, and as climate changes, the distribution of the world's vegetation will change. Moreover, the pattern and severity of disturbance, especially fire, will also change (Overpeck and others 1990, 1991; K. Ryan 1991).

Climate-controlled relationships between vegetation structure and species composition occur similarly along both altitudinal and latitudinal gradients. Changes occur along these gradients such that every 1,640 foot (500 m) increase in altitude is roughly proportional to a 171 mile (275 km) increase in latitude (Hopkins bioclimatic law) (McArthur 1972). Given

Table 8-2—Climate/weather system interactions with vegetation.

Climate/weather influences on vegetation	Vegetation influences on climate/weather
Solar energy	**Albedo**
Temperature	**Evapotranspiration** (H_2O)
Relative humidity	**Photosynthesis** (O_2)
Precipitation (timing, amount)	**Respiration** (CO_2, H_2O)
Atmospheric chemistry	**Methane** (CH_4)
Wind (direction, speed, extremes)	**Convection**
	Advection
	Desertification

time to establish equilibrium following an average annual temperature increase of 6.3 °F (3.5 °C), vegetation zones in the Rocky Mountains can be expected to shift approximately 2,167 feet (630 m) up mountain slopes (fig. 8-3), or 213 miles (350 km) farther north. The rate of vegetation movement associated with the shift in isotherms is several times faster than known species migration rates (Davis 1990; Gates 1990). Massive shifts in biome boundaries should be expected (IPCC 1996a; King and Neilson 1992; Neilson 1993; Overpeck and others 1991). In many cases, species will not be able to migrate and populations will become fragmented (Peters 1990).

Greenhouse changes will affect numerous biochemical processes that will alter ecological relationships (Joyce and Birdsey 2000; Schimel and others 1999). Photosynthesis, respiration, decomposition, and nutrient cycling will all be affected (Agren and others 1991; Bazzaz 1996; Long and Hutchin 1991; Mooney 1991). Responses of these four major physiological functions are interdependent and vary with temperature and atmospheric chemistry. However, they each exhibit different responses to changes in temperature. For example, the temperature response curves for photosynthesis and respiration differ, and an increase in CO_2 will have a major impact on photosynthetic rates. So the current balances within individual plants or communities cannot be projected into the future. Substantial increases in water use efficiency (ratio of the amount of CO_2 assimilated during photosynthesis to the amount of H_2O transpired) may result from increased atmospheric CO_2 (Bazzaz 1996; Houghton and others 1996; Mooney and others 1991; Strain 1987). As a result plant growth may accelerate greatly. However, all living cells respire, and the respiration rate increases with temperature (M. Ryan 1991).

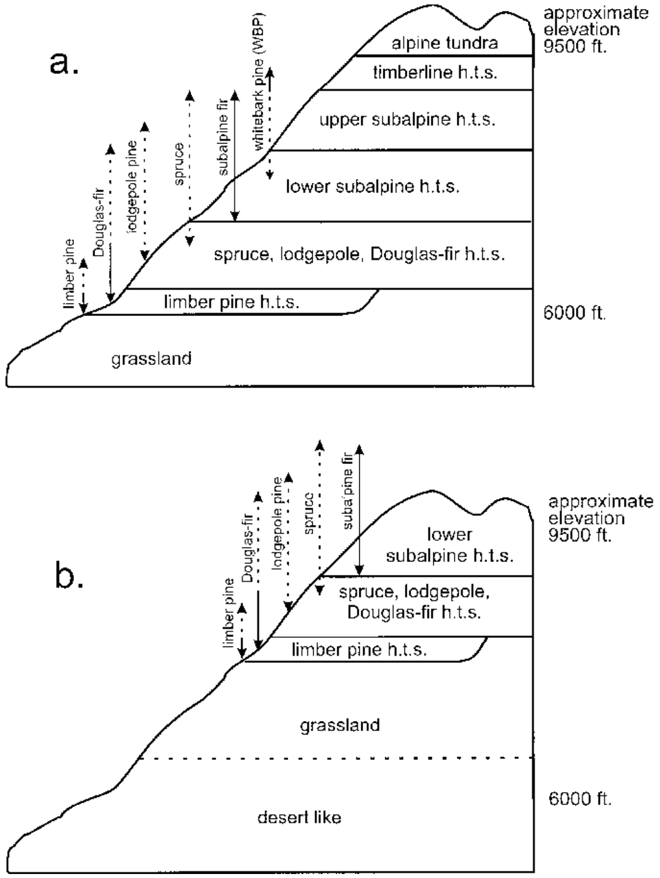

Figure 8-3—Current vegetation zones in the Bitterroot Mountains of Montana and Idaho as a function of elevation (Arno 1980) (A). Projected vegetation zone changes associated with warmer annual temperatures associated with a doubling of CO_2 in the atmosphere (B). This simple one-dimensional projection does not take into account the many dynamic interactions but does illustrate the relative magnitude of possible shifts in vegetation zones.

Because woody plants have more nonphotosynthetically active living tissue (for example, large root systems and sapwood) than herbaceous plants, temperature-influenced changes in forests and woodlands are expected to be relatively large compared to grasslands (IPCC 1996b), and mature forests are more likely to be severely affected than young forests (M. Ryan 1991; Waring and Running 1998). The loss of carbon during respiration will increase with temperature, thereby potentially reducing the effect of increased water use efficiency. If increased CO_2 alters carbon to nitrogen ratios of plants, then decomposition, nutrient cycling, and insect and disease resistance will be altered (Vitousek 1994).

Regional climate dominates the zonation of vegetation, but microclimate, soils, life cycle processes (for example, germination and growth), and ecological interactions such as competition, herbivory, and fire strongly affect the external morphology and physiological ecology of communities within vegetation zones. All of these can be expected to change in response to changes induced by greenhouse gasses.

Studies have not addressed interspecific and intraspecific interactions that affect growth rates, allocation (that is, how a plant's growth is allocated between leaves, roots, fruiting, and so forth), and community relationships (Joyce and Birdsey 2000; Mooney 1991). However, given the complexity of species traits, it is unreasonable to expect current community relationships to remain unchanged in the future (Delcourt and Delcourt 1987; Foster and others 1990; IPCC 1996a,b). For example, temperature, moisture, and photoperiod exert strong controls over phenology and growth.

Species are adapted to a range of seasonal patterns. Significant changes in these seasonal patterns can lead to asynchronous development, which can lead to reproductive failure and growth loss (Grace 1987). Also, height growth and foliage biomass have been shown to increase at elevated CO_2 levels (Kramer and Sionit 1987). If canopy structure changes, the light available for understory plants will be altered, and elevated CO_2 partially compensates for low light (Cure and Acock 1986). Because not all species sustain enhanced growth for long periods of time, the effect on the competitive relationships between overstory and understory species is uncertain. Water use efficiency varies by species, and some species respond to elevated CO_2 by increasing root to shoot ratios. Thus, competition for water and nutrients will change. Species with the C_3 photosynthetic pathway (for example, woody plants and "cool season" grasses) show greater increases in growth at elevated CO_2 than plants with C_4 pathway (for example, "warm season" grasses) (Houghton and others 1996; Smith and others 1987). Cheatgrass, an exotic C_3 grass in the Western United States, is especially responsive to elevated CO_2 (Smith and others 1987). Climate change is expected to favor early successional species assemblages over later ones (Bazzaz 1996).

Climate not only affects regional vegetation, but vegetation in turn affects both regional climate and microclimate (fig. 8-1, table 8-2). The character of surface vegetation affects the amount of solar energy absorbed versus reflected. Evapotranspiration from actively photosynthesizing foliage contributes substantial amounts of water vapor to the atmosphere, potentially affecting local precipitation. Both living and dead vegetation produce CO_2 during respiration and release a variety of other compounds to the atmosphere. Some of these compounds are greenhouse gasses (for example, CH_4), and some contribute to air quality problems such as regional haze and smog (for example, trace hydrocarbons).

Without question, global change has affected interspecific relationships and will continue to do so, likely at an accelerated rate. The effects will likely cascade throughout ecosystems. For example, increased water use efficiency of upland plants can be expected to reduce stream flows (Running and Nemani 1991), thereby affecting aquatic systems.

Interspecific relationships are too complex and poorly understood to be predicted. Given this complexity of the interactions, managers and policy makers are not likely to have significantly improved scientific bases for their actions, and many changes are likely to go undetected until major shifts occur.

Fire and Vegetation Interactions

The species composition of a community is determined by the successive birth and death of the individual community members. How fast vegetation responds to changing climate depends on species life histories, migration rates, and rates with which suitable regeneration gaps are created. Fire has played and will continue to play a significant role in determining vegetation physiognomy, structure, and species composition in the world's temperate and boreal ecosystems (Agee 1993; Crutzen and Goldammer 1993; DeBano and others 1998; Rundel 1982; Wright and Bailey 1982).

Fire is a major cause of plant mortality. For example, fire preferentially kills trees of short stature or thin bark (fig. 8-4). Likewise, fire creates gaps that new individuals colonize. Thus, changes in fire may greatly accelerate vegetation's response to changing climate. Such interaction of climate, vegetation, and fire has influenced the presence and rate of most ecosystem processes in forest and rangeland settings (Heinselman 1981). For example, fire return intervals influence the distribution of life forms and regeneration modes present on a site (Noble and Slayter 1980). The composition and structural integrity of some ecosystems are so strongly influenced by the fire regime that they are considered to be "fire dependent" (DeBano and others 1998; Habeck and Mutch 1973; Turner and Romme 1994; Wright and Bailey 1982). The severity of fire, which depends on the amount and type of biomass present and weather conditions at the time of the fire, exerts a strong influence on plant survivorship and regeneration (fig. 8-5). Therefore, altered fire regimes under a future of global climate change can be expected to accelerate vegetation changes on the landscape (King and Neilson 1992; Overpeck and others 1991;

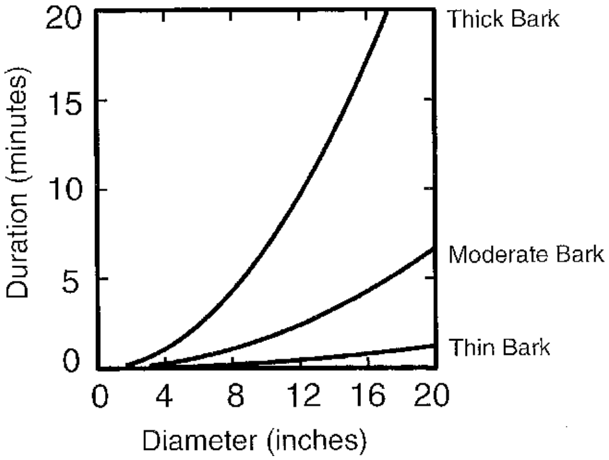

Figure 8-4—Variation in predicted stem mortality as a function of bark thickness and fire duration (adapted from Peterson and Ryan 1986). Area above a bark thickness curve implies cambium death, area below implies survival.

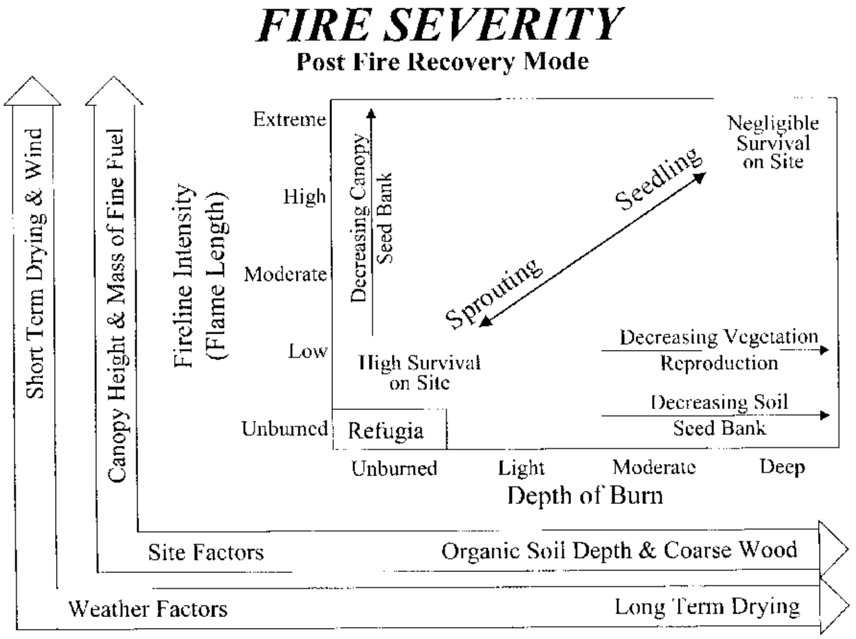

Figure 8-5—Postfire vegetation recovery varies with fire severity (adapted from Ryan and Noste 1985). In this concept of fire severity, the y-axis represents heat pulse above the fire and the x-axis represents the heat pulse down into the soil.

Romme and Turner 1991; K. Ryan 1991; Weber and Flannigan 1997).

Fire exerts selective pressure both at the individual plant and community level (Noble and Slatyer 1980; Rowe 1983). Short fire cycles favor species that endure fire by juvenile sprouting or evade fire by storing seed in the soil, invading from offsite, and having short life cycles. Intermediate fire cycles favor species that resist fires when mature or evade fire by storing seed in the canopy, but sprouting and invasion by offsite colonizers also occur. Long fire cycles favor species that typically avoid fire. Such species exhibit low resistance to fire injury and regenerate predominantly by seed. If the fire return interval is reduced to a period less than the time to sexual maturity, then a species will no longer be able to complete its life cycle on the site and could be lost from the site. The ensuing rate of reseeding will depend largely on the size of the area burned and the mobility of the seed.

The quantity, chemistry, and size distribution of fuels will change as species, growth patterns, and decomposition change (table 8-3). For example, high temperatures, drought, and nutrient shortages may lead to stress-induced mortality (King and Neilson 1992; Waring 1987) and early leaf senescence, thereby accelerating fuel accumulation. Also, an increase in carbon to nitrogen ratios will reduce decomposition (Agren and others 1991) and modify the role of fire as an agent of decomposition and nutrient cycling (Gosz 1981; Rundel 1982).

Table 8-3—Fire interactions with vegetation.

Vegetation influences on fire	Fire influences on Vegetation
Biomass ("loading" mass/area)	**Survival** (resistance to fire injury)
Bulk Density (mass/volume)	**Regeneration** (seeding vs. sprouting)
Size Distribution (surface area/volume)	**Injury** (stress and loss of vigor)
Chemistry (volatiles vs. nonvolatiles)	**Competition** (light, water, nutrients)
Live vs. dead ratio	**Community dynamics**
Shading/exposure	**Structural composition**
Strata (surface vs. overstory)	
Continuity (horizontal and vertical)	

Uncertainty of Interactions: Can We Predict the Future?

Interactions between climate, vegetation, and fire are complex and uncertain; thus, expectations for fire management are general and tentative. We can hypothesize how change in one factor will change another, but in actuality several climatic forcing factors will change simultaneously and initiate many internal adjustments within individual plants and communities. The relative abundance of species may shift because some are less adapted to the climate-altered site. Some species may regenerate but will be unable to successfully complete their life cycle given new climate and fire regimes. For example, redstem ceanothus, a valuable wildlife forage species in the Northern Rocky Mountains, and similar species that rely heavily on seed stored in the soil, sometimes for centuries, could be eliminated from sites by regeneration failure resulting from new climatic extremes, particularly early season drought and severe fire. If changes in climate and fire regimes lead to extensive species losses on a site, then migration of species from offsite will be accelerated. Species with wide ecological amplitude should be favored over those with narrow, specific habitat requirements. Regeneration strategies best suited to unstable conditions should also be favored. The additional environmental stresses and the increased frequency and severity of disturbance will likely favor the expansion of exotic and invasive species (Baskin 1998; Hogenbirk and Wein 1991).

Given changes in climate, soils, nutrients, and fire, many endemic populations will not be able to compete and successfully complete their life cycles on their current sites. They will become locally rare or extinct unless they are able to colonize new areas. Some species, particularly those that predominantly reproduce vegetatively or from seeds stored in the soil, are not highly mobile. While they may regenerate prolifically following site disturbance, they are less likely to take advantage of climatic-induced disturbance off site. These species should be slow to migrate to new areas that are within their ecological amplitude.

Given the altitudinal shift in life zones, numerous alpine species will become locally rare or endangered because there is no higher zone into which they can migrate (Franklin and others 1991; Peters 1990; Romme and Turner 1991). Similarly, subalpine species such as whitebark pine will be lost from all but the higher mountain ranges. Poor soil development will retard the migration of subalpine species into the former alpine zone, but montane species should migrate freely to higher elevations. If high temperatures and moisture stress severely limit productivity, they could threaten the continued existence of low elevation forests (IPCC 1996b). The advance of dry woodland and steppe species into these forests may be slowed by their lack of shade tolerance, but they should invade readily on sites disturbed by fire. Increased temperature and drought can be expected to increase the decomposition of peat soils and increase their susceptibility to fire (Hungerford and others 1995). The current boundaries between temperate and boreal forests, and between boreal forests and tundra, are expected to shift northward but not necessarily at the same rate.

In general, climatic change may be expected to result in improved habitat conditions at the cooler-wetter limits of a species' range and poorer conditions at its warmer-drier limits. However, many communities exist as "habitat islands" isolated by ridges or valleys or surrounded by cultivation and urban areas (Peters 1990; Peters and Darling 1985). These form effective barriers against species migrations. The rate of climatic change may be much more rapid than species' ability to migrate (Davis 1990; Gates 1990).

Understanding the potential impacts of climate change on vegetation and fire will require a level of integration not previously attempted in ecosystem studies (Mooney and others 1991). Several authors (Agren and others 1991; Franklin and others 1991; Keane and others 1997; Neilson 1993; Overpeck and others 1991) have attempted to understand the complex interactions by using process-based computer models to simulate long-term ecosystem changes in response to changes in climate. Keane and others (1997) provided the most comprehensive treatment of fire and climatic interactions on biogeochemical cycling. They used the Fire-BGC and FARSITE (Finney 1998) models to simulate changes in stand structure, species composition, and water and gas exchange over a 250-year time span in Glacier National Park, Montana. For model comparisons they simulated four fire management scenarios: (1) current existing climate and complete fire exclusion, (2) current existing climate and recent historical fire frequencies, (3) future expected climate and complete fire exclusion, and (4) future expected climate and expected future fire frequencies. Their results indicate that, because fire tends to maintain younger forests and younger forests have lower respiration, the Glacier National Park landscape respires less carbon to the atmosphere with periodic fires, even after taking fire emissions into account (table 8-4). Smoke emissions nearly doubled in the future climate/fire scenario (4), but these fluxes were small relative to those from autotrophic and heterotrophic respiration in unburned forests (scenario 3). Future climate was predicted to result in more frequent and severe fires.

These results are for one ecosystem, and results are likely different for other ecosystems, especially where fire has not played such a strong historic role in vegetation development. However, the prediction of reduced atmospheric flux of greenhouse gasses

Table 8-4—Annual carbon flux (thousand tons C/year) on the McDonald and St. Mary drainages, Glacier National Park landscape averaged across the 250 year simulation period. Table adapted from Keane and others (1997).

Carbon sources[a]	No fires, current climate (1)	Historical fires, current climate (2)	No fires, future climate (3)	Future fires, future climate (4)
Heterotrophic respiration (HR)	820	768	942	810
Autotrophic respiration (AR)	1,168	1,087	1,466	1,128
Total respiration (TR=HR+AR)	1,989	1,855	2,409	1,938
Total fire emissions	0	15	0	24
Total carbon emissions	1,989	1,871	2,409	1,962

[a] Carbon, expressed in units of 1,000 tons/year, can be converted to Gg/year if multiplied by 0.9072. Gg is a gigagram (10^9 grams).

associated with periodic fire illustrates that, because of the complex interactions among ecosystem functions, ecosystem responses may be counterintuitive.

Considerable uncertainty still exists as to how far and how fast climate will change. The autecology of many species is poorly known so it is not possible to make quantitative determinations of how they will respond. Because future climate and vegetation are uncertain, it is not possible to quantify changes in fire potential. Considerable research is needed before we can confidently predict the magnitude of climate change, its effects on vegetation and fire, and feedbacks to the climate system. Given the complexity of the problem, it is unreasonable to expect significantly better information in the near future. Given the magnitude of potential implications to fire management, long-range planning should recognize the need for greater resources in fire management (K. Ryan 1991; Stocks 1993).

Global change is a fundamental fact that natural resource managers must face. The direction and magnitude of climate change over the next few generations are uncertain, particularly at the regional level. But the continued changes in land use are likely to affect fire management regardless of the degree of climate change. Given that weather patterns and atmospheric chemistry are likely to change, and given the introduction of exotic species, management activities based on the goal of restoring the historic range of variation may not succeed (Millar 1997). Active manipulation of wildlands and their disturbance regimes may be necessary to try to maintain the continued presence of numerous species (Peters 1990; Sampson 1997).

Notes

James K. Brown

Chapter 9: Ecological Principles, Shifting Fire Regimes and Management Considerations

This chapter presents a broader, more fundamental view of the ecological principles and shifting fire regimes described in the previous chapters that have important implications for ecosystem management. Also included are strategies and approaches for managing fire in an ecosystem management context and sources of technical knowledge that can assist in this process. Research needs are also described. The ecological fundamentals that underlie the effects of fire on flora and fuels can be described under four broad principles:

1. Fire will occur with irregular pattern depending on climate.
2. Diversity of species and vegetation pattern depends on fire diversity.
3. Fire initiates and influences ecological processes such as regeneration, growth and mortality, decomposition, nutrient fluxes, hydrology, and wildlife activity.
4. Humans exert a commanding influence on ecosystems by igniting and suppressing fire.

Ecological Principles

Fire Recurrence

Fire as a disturbance process on wildlands has occurred as long as vegetation has been present on earth. The history of fire can be traced through charcoal fragments back to the Paleozoic Era, several hundred million years ago (Agee 1993). Lightning that can start fires occurs at a mind boggling rate. Approximately 8 million strikes per day occur globally (Pyne 1982). Human ignitions were common historically and continue to be common today. Wildland fires will continue to happen; the important questions about fire occurrence are when, where, and of what severity?

The frequency of historical fire varied widely across North America depending on climate. Fire return intervals typically ranged from 2 to 5 years in ecosystems supporting abundant cured or dead fine fuels such as the Southern pines, Southwestern ponderosa pine, and oak savanna. They ranged from 5 to 35 years for dry site conifers, shrublands including California chaparral, and most grasslands; 35 to 200 years for mesic site Western and Northern conifers; 200 to 500 years for some Eastern hardwoods and wetter site conifers; and 500 to 1,000 years for extremely cold or wet ecosystems such as alpine tundra and Northwestern coastal spruce-hemlock forests.

Our knowledge of fire frequency is largely based on tree ring analyses and postfire stand ages, which only allow a glimpse of fire history over the past several hundred years—a rather short climatic period. Nonetheless, it provides a basis for understanding the recurrence of fire that can be useful in planning. Keep in mind that climate could indeed change and in turn influence the occurrence of fire and the nature of vegetation response.

Historically, fires have occurred at irregular intervals, largely determined by climate. Dendroclimatological studies in western Canada (Johnson and Larsen 1991) and the United States (Swetnam 1993) have shown that climatic cycles within cycles sometimes influence fire frequency. For example, in giant sequoia forests, precipitation was the most important influence on fire occurrence over periods of years such as the recurrent episodes of the climatic phenomena El Nino and La Nina (Swetnam and Betancourt 1990). However, temperature was the most important influence on fire frequency over periods of decades to centuries. In both cases fuel moisture content was probably the important fuel property most influenced by climatic trends in precipitation and temperature. A study of presettlement fire frequency regimes of the United States (Frost 1998) suggests that patterns of fire recurrence, termed "fire periodicity," can be considered as regular or irregular. For fire regimes having high fire frequencies (average fire-return intervals of 0 to 10 years), individual fire occurrences were considered nonrandom because they clustered around a mean fire frequency. For fire frequencies greater than 10 years, individual fires occurred irregularly or in a random pattern.

Biodiversity

Biodiversity is broadly defined as the variety of life and associated ecological processes that occur in an area. This variety is sometimes broken down into genetic, species, and ecosystem components (Salwasser 1990). In dealing with vegetation, it is convenient to think of the spectrum of components as being plant, community, and landscape. The landscape can be viewed as a mosaic of patches, which are plant communities typically described as vegetation types, successional stages, stands, and age classes.

Fire regime types influence biodiversity in various ways (Duchesne 1994). In forest ecosystems, understory fire regimes have the greatest influence on biodiversity within plant communities because the understory vegetation is more affected by fire than the overstory. Stand-replacement fire regimes substantially influence biodiversity across the landscape by affecting the size, shape, and distribution of patches. Mixed fire regimes probably have the most influence on biodiversity within plant communities, but also affect patch characteristics or between community diversity. In grassland ecosystems, fire frequency and seasonal timing largely determine biodiversity.

Biodiversity can be increased by fire in many ecosystems and reduced by eliminating fire (Keane and others, in press). Variability of fire regimes in time and space creates the most diverse complexes of species. Thus, landscapes having fires with high variability in timing, intensity, pattern, and frequency tend to have the greatest diversity in ecosystem components (Swanson and others 1990). The phrase "pyrodiversity promotes biodiversity" coined by Martin and Sapsis (1992) aptly summarizes this concept. However, biodiversity can be reduced when fires occur much more frequently than happened under the historical fire regime. An understanding of the underlying relationships provides a basis for managing fire to meet conservation of biodiversity goals.

Plant Response to Fire

Chapter 2 explained the many adaptive traits that allow plant species to survive fire. In fact, many species depend on fire to continue their existence. Traits such as thick bark, fire resistant foliage, and adventitious buds allow plants to survive low to moderate intensity fires of relatively short duration. Traits such as fire stimulated germination, belowground sprouting parts, and serotinous cones allow plants to reproduce following high severity fires. For any particular plant to survive and persist, its adaptive traits must be compatible with characteristics of the fire and the timing of its occurrence. Fires can vary in intensity, duration, severity, seasonal timing, and frequency. Other factors, especially weather and animal impacts, can greatly affect whether a species can reproduce and continue its existence following fire. Grazing by ungulates can influence postfire successional patterns and flammability of future fires (Smith 2000).

Fire severity and intensity have a large influence on composition and structure of the initial plant community following fire. Fire intensity mostly influences survival of aboveground vegetation. Fire severity accounts for both upward and downward heat fluxes; thus, it is a better indicator of initial postfire flora and other fire effects. For example, when moisture contents of the forest floor fuels are high, a surface fire may burn at high intensity yet not damage sprouting tissues in the duff layer and mineral soil. Conversely, under low forest floor moisture contents, a surface fire may burn at low to moderate intensity yet consume the forest floor and damage many sprouting tissues. As a general rule, burned areas tend to return to the same flora that was there before fire (Christensen 1985; Lyon and Stickney 1976). However, fires of high severity create opportunities for new plants to establish from offsite seed. Large, high severity burns can be slow to recover depending on available seed sources. Fires of low severity are followed by a strong sprouting response except where annuals are the dominant vegetation.

The timing of fire including both seasonality and frequency is crucial to managing for conservation of biodiversity. This aspect of fire management can be easily overlooked because of emphasis on controlling fire and meeting air quality constraints. Seasonal timing of fire is important because it largely determines fire severity and related mortality. It particularly affects reproduction of herbaceous plants and shrubs. For example, in some ecosystems spring and summer fire may produce abundant postfire flowering while late summer and fall fires may produce little. Perennials in Texas survive spring fire, but annuals are harmed if fire occurs before seed is produced (Chandler and others 1983). Evidence suggests that to maintain long-term (decades) diversity in a tall grass ecosystem, fire should be applied at different times of the year to achieve successful seedling establishment and productivity for a variety of plants (Bragg 1991).

Fire frequency is a particularly important consideration in short fire return-interval regime types because a period of several years to perhaps a decade can be critical for survival of some species. Frequent fire regimes that allow control of shrubs are critical to maintaining grassland ecosystems (Wright and Bailey 1982). Many rare and threatened species have declined with reduction of fire frequency (see Greenlee 1997). Some fire dependent species in the Southeastern United States seem to require a 1 to 3 year fire return-interval (Frost 1995). In contrast, local species extinctions can occur with fires that occur too frequently, although it is generally accepted that locally rare plants have greater chances of surviving on landscapes having diverse vegetation communities and structure created by diverse disturbance histories (Gill and Bradstock 1995). A problem today is that plants adapted to short fire return-intervals can be harmed by fires burning with high intensity and severity in accumulated fuels that resulted from long fire-free periods (Sheppard and Farnsworth 1997).

Community and Landscape Responses to Fire

Species diversity within a vegetation community such as a stand or a patch depends on the collection of species in the community, their adaptive traits, the timing of fire, and the nature of fire as it moves through the community. The spatial arrangement of fuels and individual plants can be important to survival, particularly where fuels are unevenly distributed. Variable fire weather can also influence survival. Concentrations of live or dead fuels can generate much greater fire intensities and severities on relatively small sites. This could enhance or reduce diversity depending on the community. For example, in a Douglas-fir forest, localized fuel concentrations may result in fire-created gaps or holes in the canopy. This would create structural diversity and stimulate understory vegetation, a typical response to fire in a mixed fire regime (fig. 9-1). However, in a ponderosa pine forest, excessive mortality to highly valued old growth trees could be a consequence.

Ecosystems and plant communities are considered to be fire dependent when their continued existence depends on recurrent fire. Where fires occur regularly and frequently, such as in African savannas, open pine communities, and Mediterranean shrublands, they may remain stable for millennia (Chandler and others 1983). Repeated fires in fire-dependent communities maintain a dynamic process that creates diversity across the landscape, but if fire is excluded, biodiversity would probably diminish (Chang 1996). It has been argued that fire-dependent communities have evolved flammable characteristics that help ensure repeated fires and the cycle of renewal (Mutch 1970). However, the evolutionary argument remains unsettled (Chang 1996, Christensen 1993b).

Stand-replacement and to some extent mixed regime fires create patches on the landscape of differing dominant vegetation and stand structures (fig. 9-2). Patches can vary greatly in size and shape depending on the biophysical features of the landscape and fire behavior. Winds of variable speed and direction can cause fire behavior to create a variety of fire shapes. Terrain and landforms, rather than other fire influences, primarily determine patch dynamics in heavily dissected landscapes (Keane and others, in press). For example, fires in the nonmountainous boreal forests were typically large (often well over 10,000 acres) but medium to large (100 to 10,000 acres) in conifer forests of western mountains (Heinselman 1981). Even in large fires in mountainous terrain, fire severity can vary considerably within the burn, leaving a patchy distribution of fire effects (Turner and Romme 1994). Generally, on landscapes characterized by large stand-replacement fires, the pattern is naturally coarse grained. On landscapes supporting smaller stand-replacement fires, the pattern is finer grained. On landscapes having understory fire regimes, occasional trees are killed, creating gaps. This leaves a fine grained pattern in the overstory such that the notion of patches is not as helpful for describing landscape

Figure 9-1—A mixed severity fire burned through this Douglas-fir stand in Yellowstone National Park killing about half of the trees leaving gaps and large openings in the canopy.

Figure 9-2—Stand-replacement fire sustained during low wind speeds by burning in heavy accumulations of dead surface fuels, Yellowstone National Park.

diversity. In these fire regimes, considerable structural diversity can exist within communities.

As time since last fire increases, succession advances all stands to similar communities gradually reducing structural diversity (Keane and others, in press). Extending fire-free periods also increases the likelihood of larger fires, hence larger patches and less patch diversity (Bonnickson and Stone 1982; Heinselman 1981; Swetnam 1993). In whitebark pine forests, Murray (1996) found that lack of fire created high elevation landscapes with high mean patch size and low diversity. Romme (1982) found that fire control policies tended to reduce landscape richness and patchiness and increase evenness in Yellowstone National Park, although in some situations, exclusion of fire actually increased landscape diversity. Knowledge of fire regimes can help managers choose alternative land practices involving fire that favor landscape diversity compatible with natural ecosystems.

Ecological Processes

Fire is an ecological process that triggers an amazing network of other processes and associated conditions. To explain this network, it can be helpful to categorize fire effects into first and second orders. First order effects are the immediate actions of fire and include plant mortality, consumption of organic material, creation of smoke, and changes to the physical-chemical environment. Second order effects are many and depend on the nature of first order effects and the postfire environment, especially soil, weather, and animal activity. For example, here is a partial list of second order effects:

1. Change in microclimate
2. Increase in range of soil temperatures
3. Change in soil nutrients and microbial activity
4. Regeneration of vegetation
5. Succession and new vegetation patterns
6. Change in plant growth rates and competitive interactions
7. Altered wildlife habitat and activity of invertebrates and vertebrates
8. Changed water storage capacity and pattern of runoff

Plant mortality, regeneration, and growth are fire effects of obvious importance to land managers because they determine the characteristics of flora and fuel that are readily observable as succession proceeds. Less apparent but nonetheless important, especially to the pattern of fuel change, is the decomposition process that involves fire, insects, and pathogens in varying roles.

Successional Pathways

The classical concept of succession was based on the perception that plant communities evolved over time toward a final climax state that remained stable indefinitely. However, modern ecologists have rejected this concept and now view succession as a dynamic process that can move in alternative directions under the influence of periodic disturbance and never reach a stable end point (Christensen 1988). A useful method of portraying succession utilizes the multiple pathways approach (Connell and Slayter 1977; Kessell and Fischer 1981) where successional classes or stages are linked along pathways converging to one or several somewhat stable late-successional community types. Successional classes are described by vegetation type and structural stage. The number of succession classes, pathways, and time steps between classes can vary depending upon knowledge and the application. This approach allows fire of varying severity and other disturbances such as grazing and silvicultural cuttings to be incorporated in the conceptualization of successional processes.

Time is a key element in understanding succession (Wright and Heinselman 1973) and explaining it to others. Some plant communities such as mesic and wet site grasslands regain their former composition and structure within only 1 or 2 years after disturbance (fig. 9-3). For other ecosystems, some compositional change may continue to occur long into the future. Forest and shrubland communities vary greatly in the time necessary for recovery to a mature condition. In understory fire regimes, vegetation usually recovers rapidly. Structural changes are small or fine-grained and may not be readily apparent. In stand-replacement fire regimes, a young forest condition may appear within 20 or so years. But it could take several times longer in large severe burns where tree seed sources are limited.

Decomposition

Fire, insects, and pathogens are responsible for the decomposition of dead organic matter and the recycling of nutrients (Olson 1963; Stoszek 1988). Fire directly recycles the carbon of living and dead vegetation. The relative importance of fire and biological decomposition depends on site and climate (Harvey 1994). In cold or dry environments biological decay is limited, which allows accumulation of plant debris. Fire plays a major role in recycling organic matter in these environments. Without fire in these ecosystems, nutrients are tied up in dead woody vegetation. In forests, tree density and understories thicken causing increased competition and moisture stress. In turn,

Figure 9-3—One year after a prescribed fire in a mountain big sagebrush community, this mesic site recovered to domination by perennial grasses and forbs, Caribou National Forest, Idaho.

this increases the likelihood of mortality from insects and diseases leading to increased dead fuels, higher intensity fires, and possibly volatilization of more nutrients. In grassland ecosystems where both fire and grazing are excluded, thatch or dead herbaceous litter accumulates, which depresses herbage yields and the number of plant species (Wright and Bailey 1982). Fire can help control encroaching shrubs and trees; increase herbage yield, utilization of coarse grasses, and availability of forage; and improve habitat for some wildlife species.

Fire both creates and consumes fuel. It increases available fuel by killing shrubs and trees, which leads to falldown of dead material into the surface fuel complex. Moisture contents of dead fuels average much lower than live fuels, which also increases fuel availability. Insects and diseases perform similar roles. They both kill vegetation, which creates available fuel, and decompose organic matter. Fire in some circumstances enhances the opportunity for insect and disease attack. For example, bark beetles may overwhelm fire-injured conifers, and wood rotting organisms may invade fire-scarred deciduous trees. A complex interaction that is not well understood exists between insects and disease organisms, fire, and the environment. However, we do know that fire, insects, and pathogens evolved together as vital components of ecosystems.

Fuel Accumulation

Fuel accumulation is a term often used loosely to indicate an increasing potential for fire to start, spread, and intensify as the time since the last fire increases. Generally, in ecosystems where annual biomass increment exceeds decay, total vegetative biomass increases steadily with time because photosynthesis is an ongoing process. Fuels accumulate but not necessarily in a steady fashion (Brown 1985a). On forested sites much of the annual biomass increment is tied up in live tree boles where it is unavailable for combustion. In grasslands and forests having short fire intervals, fuels increase regularly over time as biomass increases. However, in medium to long fire interval conifer forests, available fuel, and fire potential may decrease as a postfire stand develops, then increase as the stand becomes old and overmature (Brown and See 1981).

Fuel accumulation and associated fire potential depend on fuel quantity as well as other important fuel properties such as compactness and continuity (vertical and horizontal). To be useful for estimating fire behavior, fuel quantity must be expressed by size classes for live and dead components. In a given vegetation type, fuel quantity, size distribution, dead-to-live ratio, and continuity are the important properties that change as succession progresses. Generally, fuel quantities accumulate to greater levels on the more productive sites in grassland, shrubland, and forest ecosystems (Brown and See 1981; Wright and Bailey 1982). In forest ecosystems much of the dead fuel exists as coarse woody debris, which includes pieces larger than 3 inches in diameter and sometimes larger than 1 inch diameter (Harmon and others 1986). The more productive sites grow larger trees, which eventually become coarse woody debris. An important consideration in management of temperate ecosystems is that coarse woody debris be recognized for the many roles it plays. It contributes to biodiversity by being part of the life cycle of macroinvertebrates, soil mites, insects, reptiles, amphibians, birds, and mammals (McMinn and Crossley 1996). It is a source of nutrients, habitat for terrestrial and aquatic life, and fuel for wildfire (Harmon and others 1986). As a fuel its most significant feature is that it becomes rotten wood, which prolongs burnout and allows fire to persist on site for long periods. Historically, large fires occurred because fire remained smoldering in rotten wood and duff for extended periods until low fuel moistures combined with high wind speeds to support intense, fast spreading fires.

Flammability increases as dead-to-live ratios increase. As fuels accumulate through growth and mortality of plants, flammability thresholds may be reached that allow fires to increase greatly in intensity. Surface fires become crown fires in conifer forests, and shrub communities burn intensely as a single fuel complex.

Fuel continuity is important because it partly controls where a fire can go and how fast it travels. In grasslands and open shrublands, heavily grazed areas and areas of low productivity form discontinuous fuels that limit spread of fire, which can be a critical obstacle to use of prescribed fire. In forests, existence of ladder fuels from understory vegetation allows surface fires to reach into the crown canopy. If the canopy is mostly closed, crown fire can readily develop under adequate wind speeds. Open canopies do not support crown fires. Increased fuel continuity can account for changes in fire severity from understory to mixed and from mixed to stand-replacement. Many options are available to land managers for altering fuel continuity through manipulation of vegetation.

Effects of fire on fuel arise basically two ways: first, reducing fuel through consumption, and second, increasing fuel by killing vegetation. Both processes affect several properties of fuel and fire potential. Initially dead surface fuel loadings are reduced, also lowering the dead-to-live ratio. If substantial amounts of shrubs, small conifers, and limbs and foliage of larger conifers are killed by fire but not consumed, they will contribute to surface fuels in the years ahead as they accumulate on the ground. Fire greatly influences fuel continuity by creating vertical and horizontal gaps within and between surface fuels and crown fuels.

Accumulation in Forests—Live and dead fuels, as well as small and large diameter fuels, can follow different patterns of accumulation. Typically, live herbaceous and shrub fuels increase following fire during early stages of stand development. Then as tree canopies close, live herbaceous and shrub fuel quantities tend to decrease on mesic sites (Habeck 1976; Lyon and Stickney 1976). However, a decrease in biomass may not occur where understories contain shade tolerant species. Fine dead fuels from foliage, bark flakes, twigs, and cured herbaceous vegetation become incorporated in the forest floor. Once crown canopies close, the amount of litter fuel remains fairly constant as newly fallen litter is offset by older litter moving into the duff layer. Duff quantities continue to increase for some time until equilibrium with decay is reached. This period varies widely from approximately 5 years in Southeastern United States (McNab and others 1978) to well over a hundred years in some boreal ecosystems.

Dead branches and tree boles accumulate on the ground in response to natural mortality and factors causing downfall (Brown 1975). Mortality factors such as fire, insects, disease, canopy suppression, and wind and snow damage impact stands in a rather haphazard manner. Thus, accumulation of downed dead fuel often occurs in an irregular pattern that is correlated poorly with stand age (Brown and See 1981).

Conifer crown fuels increase regularly; however, likelihood of crown fire may increase then decrease as the lower canopy level grows further above surface fuels. Eventually, crown fire potential increases again when surface fuels increase and understory conifers become ladder fuels. Shade tolerant species tend to have more foliar biomass than intolerant species due to their longer needle retention and higher crown densities (Brown 1978; Keane and others 1999). Because of their shade tolerance they can fill in crown canopy gaps and develop into understory ladder fuels.

Fuels critical to fire spread differ considerably between short and long fire interval fire regime types (Brown 1985a). In short fire interval forests, fine fuels such as grass, live shrubs, and needles create flammable understory fuels even in forests with vastly different decomposition rates such as in longleaf pine and ponderosa pine. The substantial quantity of fine fuels coupled with long periods of suitable burning conditions largely account for the understory fire regime. In long fire interval forests the forest floor and accumulated coarse woody debris are critical fuels. They burn with considerable heat release over a relatively long duration resulting in extensive mortality to overstory trees. They ignite other surface and aerial fuels and serve as excellent receptors of spotting embers that often allow fire to move in a leap frog fashion. Fire intervals and environments differ considerably between long fire interval types such as cedar-hemlock forests on warm moist sites and subalpine and boreal forests on cold, dry sites. Nevertheless, in both cases accumulated forest floor and downed woody fuels support stand-replacement fire particularly during extended dry periods (Romme and Despain 1989).

Accumulation in Shrublands and Grasslands—On many grasslands, grazing eliminates most of the annual production so fuel accumulation is inconsequential. In the absence of grazing, fuel quantities depend primarily on annual production, which varies substantially by site potential and annual precipitation (Wright and Bailey 1982). Fuel loading may increase for several years after a fire as some slow responding grassland communities recover. Frequently, however, productivity is increased within 1 or 2 years following fire (Wright and Bailey 1982). Herbaceous litter accumulates in some grassland ecosystems but only marginally in others. Ratios of accumulated litter-to-current production typically range from 0.25 to 0.50 (Reinhardt and others 1997).

In shrub and shrub/grass ecosystems young communities generally have a low dead-to-live ratio. Flammability depends largely on grass and sedge fuels. As shrubs become senescent or undergo mortality, dead stemwood accumulates, which significantly increases potential flammability. Dead fuel quantities tend to increase with time since last fire or with age of plant community as suggested for chaparral, however, not in a uniform nor readily predictable fashion (Paysen and Cohen 1996). Besides age, other factors such as drought, winter kill, insects, and disease can cause periodic dieback that creates substantial dead fuel quantities. As cover and height of shrubs such as sagebrush increase, fire intensity and rate of spread potential increase markedly (Brown 1982).

Human Influences

People are part of ecosystems and certainly have exerted a major, far reaching influence on fire across the landscape. Indian burning was common throughout the United States and Canada. Pyne (1982) quotes Henry Lewis as saying, "To simply note that all Indians used fire to modify their environments is no more an ecological generalization than to note that all farmers used plows." The extent of Indian burning varied considerably, however, depending on locale and population movements (Boyd 1999; Pyne 1982). Indian burning greatly extended grasslands especially in the Eastern and Midwestern United States. Most of the coastal plain from Massachusetts to Florida to Texas was savanna. Western valleys and foothills were maintained as grasslands and open forests (Gruell 1985).

Considerable debate exists about the relative importance of Native Americans and lightning in maintaining historical fire regimes (Barrett and Arno 1982; Frost 1998; Keane and others 1999). The relative importance of Native American fires was probably greater in topographically complex areas where fire compartments were smaller and where lighting ignitions were infrequent (Frost 1998). Also debated is whether anthropogenic burning should be considered part of the native or natural fire regime (Arno 1985; Kilgore 1985). Fires set by Indians were often of different seasonality, frequency, and landscape pattern than those started by lightning (Frost 1998; Kay 1995). Indian and lightning-caused fire existed for thousands of years, a short evolutionary period but a long time for plant communities to adjust to fire disturbance. This long period of fire on the landscape argues strongly for accepting both sources of ignition in considerations of Euro-American presettlement fire history used to guide management of ecosystems.

Efforts to suppress fires were modest at first relying on wet blankets and buckets around dwellings and campsites (Pyne 1982). Modern suppression capabilities relying on sophisticated communications, rapid attack, specialized equipment, and many fire fighters are a far cry from the early 1900s. Fire protection has succeeded in reducing the extent of fire and increasing fire intervals. Chandler and others (1983) suggested that as protection succeeds, fire intervals become greater and flammability increases. Then, more protection is needed to keep burned acreage down. A given protection effort and annual burned area will eventually reach equilibrium. Since the 1980s, the costs of protection and greater understanding of the role of fire have led to more hazard reduction and ecosystem maintenance rather than just protection.

For the past 100 years or so, human use of fire—earlier termed controlled burning and now prescribed fire and wildland fire use—has met with considerable controversy politically and within land management organizations. "Light burning" (understory fire) was once widely applied in the southern pines and ponderosa pine type especially in California. However, the perceived threat to effective organized fire control largely curtailed the program on publicly owned lands (Pyne 1982). Some benefits of controlled burning were still recognized, especially hazard reduction and preparation of seed beds for regeneration. In the West justification for prescribed fire was fuel reduction, namely slash burning. This single purpose use of prescribed fire resulted in short-term successes but long-term failure to optimize societal objectives for forests (Agee 1993).

More recently, the concept of ecosystem management has led to a much wider understanding of the ecological role of fire and its importance in the functioning of ecosystems. Concerns over air quality, control of fire, and costs, however, remain as major constraints on the application of prescribed fire and wildland fire use. The responsibility to see that fire is properly managed as a component of the ecosystem is now greater than ever because land managers have the power to delay and exclude fire as well as an understanding of fire's important ecological role.

Shifting Fire Regimes

Chapters 3 through 7 clearly show that fire regimes have shifted from what they were historically across most of the United States and southern Canada. In a comprehensive assessment of burning in the contiguous United States, Leenhouts (1998) estimated that approximately 10 times more area must be burned than at present to restore historical fire regimes to nonurban and nonagricultural lands. The greatest departure from historical fire regimes is in the Rocky Mountains where only a small fraction of the pre-1900 annual average fire acreage is being burned today (Barrett and others 1997). Kilgore and Heinselman (1990) estimated that the greatest detrimental effects of fire exclusion were in short interval fire regimes of the Rocky Mountains. In contrast, in long fire regimes, the effects of fire protection have not had a significant influence. In the Canadian and Alaskan boreal forest limited protection due to remoteness has maintained fire regimes essentially as they were historically.

Extensive grazing by domestic stock that reduces fuels, and fragmentation by agriculture and human developments, have also contributed to shifting fire regimes. Lengthened fire return intervals have resulted in changes of minor to major consequence to vegetation and fuels by increasing wildfire severity and decreasing species and structural diversity. A comparison of historical and current fire regimes in the Interior Columbia River Basin of about 200 million acres showed that fires have become more severe on 24 percent of the area (Morgan and others 1998) (see fig. 5-1 in chapter 5 of this volume). Fire severity was unchanged on 61 percent of the area. Fires were less frequent on 57 percent of the area, unchanged on 33 percent, and more frequent on 10 percent of the land area. Fire protection, reduced fine fuels from grazing, decreased fuel continuity from human development, and in some cases exotic plants are the most probable causes (Chang 1996; Keane and others 1999). Further analyses of changes in fire regimes and condition classes of vegetation are currently under way for the United States (Hardy 1999).

Forests and Woodlands

Changes in forest composition and structure due to shifting fire regimes have been widely documented. Generally, shade-intolerant species are being replaced with shade-tolerant species. Stand densities are increasing with development of multiple layer canopies. Outbreaks of insects and occurrence of root diseases appear to be worsening (Stewart 1988). The greatest impacts have occurred in the understory fire regime types typified by ponderosa pine and longleaf pine ecosystems (fig. 9-4). Although these two ecosystems experience widely different climates, they share the same end results of fire exclusion made worse in some locations by selective harvesting of old growth trees. Where fire regimes have shifted, growth and vigor of trees is reduced, insect and disease mortality is increased, and understory fuel loadings and continuity increased so that wildfires tend to be of high intensity, killing most or all of the overstory pine. Diversity of understory herbs and shrubs is decreased. The loss or depletion of the pyrophytic herb layer is considered to be one of the unrecognized ecological catastrophes of landscape history (Frost 1998). The extent of the problem is greater in ponderosa pine where relatively little prescribed fire has been applied. Although prescribed fire is widely applied in the South it has largely been used only for rough (accumulated understory fuels) reduction during the dormant season. Thus, lack of seasonal fire diversity in the southern pine types has limited plant diversity.

In mixed fire regime types such as coastal and inland Douglas-fir, whitebark pine, red pine, and pinyon-juniper, the results of fire exclusion have created the same problems as found in understory fire regimes. Mixed fire regimes are experiencing considerably less nonlethal understory fire than in the past (Brown and others 1994). The mixed fire regime is shifting toward a stand-replacement fire regime that favors more shade tolerant species and less landscape diversity.

In stand-replacement fire regimes, fire intervals have generally lengthened; however, the effects of this vary widely depending largely on presettlement fire return intervals and accessibility for fire suppression efforts. For example, in the lodgepole pine/subalpine fir type, which dominates the Selway-Bitterroot Wilderness, presettlement stand-replacement fire was 1.5 times more prevalent than during the recent period (Brown and others 1994). The presettlement fire return-interval was approximately 100 years. In the same type in Yellowstone National Park, characterized by a fire return-interval of about 300 years, the area burned probably has not differed between presettlement and recent periods (Romme and Despain 1989).

The age distribution of marginally commercial and noncommercial forests such as those in wilderness areas and parks is shifting to an abundance of older stands (Brown and Arno 1991). Succession is increasing the shade tolerant component of stands, making a major species shift likely if fire continues to be

Figure 9-4—A stand-replacement fire supported by accumulated dead surface fuels and live ladder fuels from dense understory trees occurred in this understory fire regime type killing the old growth ponderosa pine, Yosemite National Park.

excluded. In the case of western aspen more than half of the type has been lost (Bartos 1998), much of it due to successional replacement by conifers (Bartos and others 1983). Fire protection policies have resulted in the fire cycle in aspen shifting from about 100 years to 11,000 years; thus, if this degree of fire exclusion continues, the loss in biodiversity will be considerable. In jack pine forests the more shade tolerant balsam fir is gradually assuming dominance aided by natural deterioration and harvesting of jack pine.

Fuel accumulation patterns vary widely in coniferous stand-replacement fire regime types. Mature forests may support abundant or relatively little available fuel. However, as fire intervals are allowed to increase and stands become over mature, downed dead woody fuels and live ladder fuels from shade tolerant understory conifers can be expected to dramatically increase. The result will still be stand-replacement fire but at higher intensities, which will tend to propagate larger fires in spite of suppression efforts. This trend could lead to fewer but larger fires burning during severe fire weather years, causing less diversity in patch size and age (Keane and others 1999).

Grasslands and Shrublands

Grassland fire regimes have shifted dramatically from the presettlement period. Many ecologists consider the reduced frequency and extent of fires on rangelands due to fire protection to be among the most pervasive influences in the United States by non-Native Americans (Pieper 1994). The shift to woody plant domination has been substantial during the past hundred years. Grazing and possibly climate changes have acted with reduced fire to give a competitive advantage to woody plant species. Some woody plants such as honey mesquite become resistant to fire, develop fuel discontinuities, and reduce spread of fire. In time, recovery following fire favors shrubs over perennials (Archer 1994). This can alter the composition of ecosystems to the point that a return to the grassland type becomes nearly impossible or impractical (Brown 1995).

Historically, fires were more frequent in Eastern than in Western grasslands. High productivity of biomass was maintained in the tallgrass prairie by frequently occurring fire that recycled accumulated thatch. A diverse composition was probably favored by variable frequency and seasonality of fires (Abrams and Gibson 1991; Bragg 1991). Western grasslands appear to have generally experienced fire less frequently (Gruell 1985; Wright and Bailey 1982) but still frequently enough to hold back invasion of woody plants.

Fire regimes have shifted to too much fire in the drier portions of the sagebrush-steppe ecosystem that occupies over 100 million acres in Western United States. Fire frequency has increased in many areas due to invasion of cheatgrass and medusahead, introduced annuals that cure early and remain flammable during a long fire season. Increased fire frequency exerts strong selective pressure against many native plants (Keane and others 1999). A contrasting situation exists for the more mesic mountain big sagebrush type where decreased fire frequency and encroachment by conifers is causing a reduction in herbaceous and shrub vegetation (fig. 9-5).

Managing Fire

Fire is an integral component of ecosystems that can affect all aspects of ecosystem management. Fire regimes have shifted as a result of human influences and may continue to shift with clearly detrimental results in some ecosystems. Land managers need to know how to plan and carry out fire management strategies that successfully incorporate the ecological role of fire. Constraints on managing prescribed fire and smoke make it difficult to achieve resource goals, while protection against wildland fires allows development of undesirable ecological consequences (Brown and Arno 1991). Overcoming this predicament requires that land managers and the public alike recognize the role of fire in the functioning of ecosystems and in meeting varied resource objectives.

Strategies and Approaches

Vegetation and fire management objectives should be derived from broader ecosystem management goals to achieve desirable fire effects. Determining objectives, and the strategies and approaches for achieving them, can be simple to complex depending on land ownership and direction provided by the owners. For example, a small woodlot owner may simply want to reduce fire hazard, in which case fuel reduction objectives can be clearly stated and, if appropriate, a prescribed fire conducted to reduce the unwanted fuel. Where the direction is ecosystem management, a goal recently adopted on many Federal and some State lands (Salwasser 1994), a more elaborate process may be required to determine objectives and strategies.

Figure 9-5—Without disturbance, this sagebrush/grass community being encroached by Douglas-fir will eventually become a closed canopy forest with sparse understory vegetation, Deerlodge National Forest, Montana.

To steer this process, a guiding principle or goal for ecosystem management is to provide for conservation of biodiversity and sustainability of ecosystem composition, structure, and processes (Kaufmann and others 1994). This involves molding a management plan based on an understanding of ecosystem processes. An element missing or minimally considered from many past planning efforts was the landscape of varying scales. For this a perspective is needed that involves consideration of ecological processes across a hierarchy of land units (Hann and others 1993).

The setting of goals and objectives starts out broadly with a goal specifying the future condition of the ecosystem or a particular tract of land. This desired future condition is a vision for the future and not an objective for management action (Kaufmann and others 1994). An assessment of the ecosystem, resource potentials, and needs of people is a prerequisite for setting the desired future condition. From this, more specific objectives can be derived for managing fire. They should be specified in terms that can be monitored. Different approaches may be appropriate for doing an assessment and setting the desired future condition and the ensuing management objectives.

Consider the planning task by three types of land use zones (Arno and Brown 1989):

- **Zone I – wilderness and natural areas** objectives call for allowing fire to play its natural role to the greatest extent possible. Fire objectives may vary depending on whether it is a wilderness or natural area intended to preserve a particular condition or process.
- **Zone II – general forest and range management**, where the need to provide resource values means a wide range of vegetation and fire objectives will be appropriate.
- **Zone III – residential wildlands**, where the natural role of fire will be constrained considerably and fuel management is the primary objective.

Two occasionally troublesome facets of setting goals and objectives in Zones I and II that rely on knowledge about the ecological role of fire involve the “historical range of variability” and the goal orientation of “process versus structure.”

Historical Range of Variability

The historical range of variability (also called natural range of variability) in ecosystem components can be used to help set desired future conditions and fire management objectives. It can serve as a basis for designing disturbance prescriptions at varying spatial scales and help establish reference points for evaluating ecosystem management (Morgan and others 1994). Reference points to past functioning of ecosystems can be interpreted from various sources such as historical records, palynology, natural areas, archival literature and photographs, GIS data layers, and predictive models (Kaufmann and others 1994; Morgan and others 1994). Historical fire regimes of forest ecosystems are often characterized by determining age distribution and areal extent of seral classes across a large landscape and dating fire scars to determine fire return intervals. These techniques provide a snapshot of ecosystem conditions that covers the past 100 to 400 years. Pollen analysis can extend this period but with less precision about disturbance events (Swanson and others 1993). Estimation of historic fire frequencies in grasslands and shrublands is more problematical because of a lack of fire scars and easily determined age classes. It relies largely on historical accounts of human activities.

To what extent should knowledge of the historical range in variability be relied upon to help establish goals and objectives? This depends largely on soundness of the ecological knowledge and other ecosystem issues such as human needs and threatened and endangered species (Myers 1997). A strong argument can be made that knowledge of historical fire should be used as a guide for understanding landscape patterns, conditions, and dynamics, but not necessarily for creating historical landscapes. Knowledge of historical variability provides a basis for bringing the range of existing conditions in a landscape within the historical range (Swanson and others 1993).

A scientifically based rationale underlies the use of historical variability as a guide for managing biodiversity. Native species evolved and adapted to natural disturbance events over at least the past 10,000 years. Numerous ecological studies emphasize the close dependence of species on disturbance regimes (Swanson and others 1993). Genetic diversity (Frankel and Soule 1981) as well as landscape diversity are maintained through disturbance regimes. Where fire regimes have shifted markedly, species and landscape diversity have declined.

Concerns and limitations to using historical variability as a guide to managing ecosystems (Morgan and others 1994; Swanson and others 1993) are:

1. Difficulty interpreting past variability due to insufficient data.
2. Degree to which past and future environmental conditions may fall outside the established range of historical conditions. For example, the possibility of future climate change due to global warming is a significant concern.
3. Extent to which the range of ecosystem conditions desired by society differs from historical variability.

The natural range of variability can be determined and applied with reasonable confidence in high frequency fire regimes of forests. In understory fire regimes, considerable data on fire frequency often can be obtained by consulting published accounts or conducting studies of fire intervals on fire scarred trees. Variability of fire-return intervals can be quantified and compared with recent fire history to determine whether a significant departure has occurred (Brown 1993). In long interval stand-replacement fire regimes of some forests and tundra, estimates of the historical range of variability are more difficult to establish with certainty because of the limited number of disturbance events that can be studied. Perhaps the best technique for measuring fire regime characteristics in this situation utilizes satellite and GIS technologies to map vegetation pattern (Morgan and others 1994), an approach requiring considerable resources.

A question that often arises in interpreting fire history especially concerning wilderness and other natural areas is how Indian ignitions should be treated (see Lotan and others 1985). The prevailing thought seems to be that because Indian burning occurred over a long period, ecosystems were adjusted to fire effects from human and lightning ignitions combined and this reflects historical fire regimes. Disturbance history can only be readily and reliably measured for the past 200 to 400 years. Variability in climate, vegetation composition, and disturbance patterns has been substantially greater over the past several thousand years than over just the last 400 years. But land managers need a consistent basis on which to plan, and using measurable fire history is a practical approach. The concept of the historical range of variability can be valuable in understanding and illustrating the dynamic nature of ecosystems and in evaluating current ecosystem health.

Process Versus Structure Goals

Process and structural goal setting approaches are important to management of Zone I lands. These concepts originated with establishment of wildernesses and natural areas where the goal was to manage for naturalness. The proper role of fire in wilderness and natural areas has been characterized in terms of process-oriented and structure-oriented goals (Agee and Huff 1986). Expressed simply, do we want a natural fire regime (process) or rather the vegetation that a natural regime would have created (structure) (Van Wagner 1985)? The answer to this may always involve some degree of debate because of philosophical differences over the concept of natural (Kilgore 1985). In practice, both approaches or a mixture of the two may be appropriate depending on circumstances. Practical aspects such as costs, fire safety considerations, and size and boundaries of the ecosystem will often determine the most appropriate approach.

A strictly process-oriented goal is probably only appropriate in large wilderness areas. The process goal approach modified by practical considerations will usually be necessary.

In understory fire regimes where surface fuels have accumulated to the point that high intensity fire is likely, a structure-oriented goal is the best approach to ultimately achieve natural conditions. After fuels have been reduced using a prescription for low severity fire to avoid killing the overstory, a process goal of allowing natural ignitions can be followed if it will maintain the understory fire regime (Bonnickson and Stone 1985). Structural goals will continue to find application in understory fire regime types to restore and even maintain the natural role of fire. The structural goal approach is probably the best for management of threatened and endangered species. It may also be more efficient and esthetically pleasing (Agee and Huff 1986).

Mixed fire regime types in wilderness areas present variable, complex landscape patterns that can make structural goals difficult to achieve. Fire frequencies in the mixed type typically range from 35 to 100 years. In some localities fire has been absent long enough that fuels and stand structures appear to be falling outside the range of historical variability (Arno and others 2000). In such cases, where accumulated surface fuels and naturally occurring ignitions would favor stand-replacement fire, structural goals aimed at retaining a portion of the overstory may be appropriate to restore the mixed fire regime. If excessive fuels have not accumulated, process goals seem to be the most reasonable.

Another consideration in wilderness areas, regardless of whether structural or process goals are chosen, is when and where to use prescribed fire to meet wilderness objectives. In the contiguous United States 75 percent of Congressionally classified Wilderness areas, which occupy half of the classified wilderness land area, are too small to maintain natural fire regimes by relying strictly on natural ignitions (Brown 1993). Constraints such as concern over escape fire, lack of lightning-caused fires, conflicting wilderness goals, and air quality regulations will require prescribed fire to restore fire and mimic natural processes. Decisions to use prescribed fire must be ecologically based, but also with the realization that exacting solutions to mimicking natural fire processes are probably not feasible. Neither the determination of fire history nor applications of prescribed fire are precise undertakings.

For residential and commercially zoned lands (Zones I and II), structural goals are the most appropriate. Clearly definable and measurable end points are being sought. For example, specific conditions such as tree species and size, stand age distribution, patch size, stimulation of shrubs, increased forage production, and reduced fuel quantities may be desirable objectives.

Landscape Assessment

Managing biodiversity and for sustainability of ecosystem components and processes requires a landscape perspective. Small ecosystems are found within larger ecosystems, individuals occur within communities, and short-term processes are nested within longer term processes (Kaufmann and others 1994). The various scales fit into a hierarchical structure that determines patterns of diversity for an area (Bourgeron and Jensen 1993). A major challenge to setting vegetation and fire objectives in the context of ecosystem management is evaluating and interpreting the ecological significance of multiple scales. Vegetation scales range from individual plants, communities, seral stages, potential vegetation types, to the biome level.

Species and individual plant communities are dealt with using a fine filter approach. Traditionally, assessments of fire effects and other environmental impacts have been done on a project basis using fine and mid scale evaluations. The coarse scale aspects of ecosystems have been largely neglected. The coarse filter approach, which deals with higher scale levels such as aggregations of communities, can operate with relatively little information, yet be an efficient way to meet biodiversity goals (Bourgeron and Jensen 1993; Hunter 1990; Kaufmann and others 1994). A single ecosystem can be too small to hold viable populations of all its species, especially large predators. Thus, the coarse filter approach is best used on assemblages of ecosystems such as watersheds and mountain ranges. Both approaches are necessary to evaluate all facets of an ecosystem and meet the goals of ecosystem management (Hann and others 1993a).

Assessment of landscape and ecosystem properties can be undertaken with varying degrees of sophistication and effort. Some of these planning efforts, which are evolving through trial and error, are mentioned as examples. During the past decade agencies such as the U.S. Forest Service and Bureau of Land Management have undertaken landscape analyses on extremely large areas such as the 200 million acre Upper Columbia River Basin (Keane and others 1996) and smaller areas such as the Pike and San Isabel National Forests and Cimarron and Comanche National Grasslands in Colorado (U.S. Forest Service 1997) and the 130,000 acre Elkhorn Mountains and 46,000 acre North Flint Creek Range in Montana (O'Hara and others 1993). Details of these landscape evaluations varied but they followed three general steps (Hann and others 1993b):

1. Characterize the general composition, structure, and processes of the ecosystems and landscapes within the designated analysis area.
2. Analyze data to assess changes in structure and composition and relate the changes to previous management treatments.
3. Examine the ecosystem processes important for the area and their effects on ecosystem and landscape composition, structure, and rate of change.

Succession Modeling—Simulation of succession provides a means of predicting the long-term interaction of processes such as fire, insects, disease, and cutting of vegetation on landscapes of varying scale. Simulation can be helpful to managers and the public by helping them understand how ecosystems function and for evaluating different management alternatives. The wider availability of powerful computer capabilities has led to an increase in succession modeling efforts particularly for landscape applications. Manager-oriented computer models that simulate successional processes across large landscapes are faced with a tradeoff between realistic portrayal of ecological processes and utility of the model. Some models are too complicated to use without special training or assistance. Nonetheless, managers are increasingly using succession models in their planning while models are continually evolving and computer capabilities growing.

In choosing a model for a particular application, it is important for the temporal and spatial scales of the model to match the intended use (Reinhardt and others, in press). Models that operate over a period of decades are useful for scheduling treatments. For example, the Fire and Fuels Extension to Forest Vegetation Simulator (Beukema and others 1997) simulates fuel quantities, tree characteristics, and tree mortality in the event of a fire for single stands. Managers can use the model to help schedule thinnings and fuel treatments when potential fire behavior and fire effects on an area are deemed unacceptable (Reinhardt and others, in press).

Models that simulate fire effects over centuries are useful for providing targets for managers, for estimating the historic range of conditions, for evaluating implications of climate change, and for understanding possible long-term consequences of management actions. For example, CRBSUM was used to simulate landscape changes for different management scenarios in the Columbia River Basin (Keane and others 1996). Some of the current models that have been applied to assist land managers are summarized in appendix B.

Restoration of Fire

Restoration of fire is needed to varying extents in most ecosystems of North America to meet the holistic goals of ecosystem management. The need for restoration is most evident in high fire frequency regimes such as understory fire regime types and some grasslands and shrublands where fire has been excluded for several times longer than the average fire return interval. Although considerable knowledge supports the need for restoration of fire into wildland ecosystems, constraints and obstacles confront land managers (Brown and Arno 1991; Mutch 1994). Limited funding, air quality restrictions, concerns over escape fire, and inadequate public support can pose difficulties. Some breakthroughs in managing emissions and obtaining support have provided more latitude for prescribed fire programs (Mutch and Cook 1996).

Successful restoration involves clearly stated objectives, plans based on scientific knowledge of fire's role in the ecosystem, and adaptive learning from prescribed fire efforts. Adaptive learning is important because prescribed burning usually improves with experience. Prescription conditions and firing techniques may need to be modified to achieve objectives such as a given level of fuel reduction or to meet constraints such as holding overstory mortality to certain limits. Fire may not spread adequately under an initial prescription, thus requiring lower fuel moisture contents or higher wind speeds to be successful.

Restoration of fire can be undertaken on an entire ecosystem or on an individual plant community basis. Ideally, restoration of individual plant communities would be based on ecological considerations of the broader ecosystem of which they are a part. The extent of ecosystem assessment that is appropriate for planning restoration will depend largely on land ownership and direction given to management. For large land ownerships, restoration of entire ecosystems or large landscape areas is the soundest approach to manage landscape pattern and meet biodiversity goals. It also allows for effective placement of fuel treatments designed to disrupt fuel continuity and reduce threat of large fire occurrences. The steps undertaken by Keane and Arno (1996) to restore fire in the whitebark pine ecosystem may be useful in other situations including grasslands and shrublands. They recommend first, an inventory of landscape and stand characteristics at multiple scales; then, writing descriptions of the important processes of the landscape and stands. Landscapes and stands can then be prioritized for restoration treatment and selected based on inventory, description, priority, and feasibility. Treatments should be designed for each selected stand or landscape based on inventory and description information and implemented as efficiently as possible. Finally, treatments should be monitored to evaluate restoration success.

Restoration of fire in grasslands, shrub steppe, and savannas requires careful consideration of seasonal timing and frequency to assure that prescribed fires will spread at appropriate severities. Once woody plants have encroached to a point of dominating a site, it becomes difficult to get fire to spread with sufficient heat to kill aboveground stems such as oak in savannas (Huffman and Blanchard 1991) and juniper in sagebrush/grass communities. Perhaps the greatest obstacle to success lies with areas that have successionally lost the native mix of species and lack sufficient grass fuel to carry fire. Seeding of native species following fire may be necessary to restore a resemblance of former plant composition. Where conifers invade grasslands such as pinyon-juniper and inland Douglas-fir (Gruell and others 1986), successful spread of surface fire may require fuel enhancement work such as cutting numerous trees to create adequate surface fuels. Otherwise, crown fire may be required, which will necessitate a more flammable, narrow fire prescription that can limit burning opportunities.

Prescribed Fire and Silviculture

Prescribed fire and silviculture can go hand in hand for restoration of forest stands and ecosystems. Some consider prescribed fire to be a silvicultural technique even though it goes far beyond the usual goals of silviculture that are oriented to producing tree products and desirable forest stand structures. One debatable point is the extent to which it is desirable to have management mimic the kinds of stands and landscape structures that typified presettlement fire regimes. However, an understanding of similarities between characteristics of fire regime types and silvicultural stand structures can be helpful for integrating fire with silviculture to restore fire as a process and meet ecosystem management goals. The following description of stand structure and silvicultural practices based on a discussion by Weatherspoon (1996) applies to individual stands. Stands can be treated differently to manage landscape-level vegetation.

Even-Aged Stands—These stands originated naturally mostly from high-severity, stand-replacement fires that killed most of the trees. Silvicultural methods that produce even-aged stands include clear-cutting, seed tree, and shelterwood cutting. Shelterwood or seed trees are typically removed after regeneration is secured. Pile burning or broadcast burning is commonly used to reduce fuels and prepare sites for regeneration. Leaving snags, large downed woody material, and untreated patches in larger treatment units is important for meeting biodiversity goals.

Two-Storied Stands—These stands were associated with moderate to high severity fire typical of the mixed fire regime type. Retention shelterwood (also called irregular shelterwood or shelterwood without removal) is the silvicultural method for treating the stand. Prescribed underburning can often be practiced to manage fuels and create within-stand diversity. Once created, the stand would never be devoid of large trees because each regeneration cutting would be accompanied by retention of some overstory trees. Snags could be readily created.

Uneven-Aged Stands with Even-Aged or Even-Sized Groups—These were associated with low to moderate severity fires associated with the understory fire regime type and perhaps to some extent with the low severity end of the mixed fire regime type. Silviculturally this stand structure is mimicked with the group selection cutting method. Skillful prescribed underburning is required to apply the proper severity for maintaining this structure. Jackpot burning and two-stage burning under different prescription conditions may be appropriate.

Uneven-Aged Stands with Fine Tree Mosaic—These stands are characterized by three or more sizes and ages of all tree species distributed rather uniformly throughout the stand. This stand type is thought to have developed primarily with shade-tolerant conifers over long periods following stand-replacement fire. It is incompatible with frequent fires. The individual tree selection method is used to maintain this structure. This stand structure could be considered to represent open stands of ponderosa pine and longleaf pine. Ecologically, however, they fit better with the previous category of even-aged groups.

Understory Fire Regime Type

Restoration of the understory fire regime type requires application of frequent, low intensity fire, which has been excluded for excessive periods of time. Restoration approaches can vary considerably depending on stand and fuel conditions. The objective generally is to create more open stand structures consistent with historical disturbance regimes. A wide range of stand densities can be appropriate depending on site potential and silvicultural objectives. Various even-aged and uneven-aged stand structures can be utilized. Favoring the long needle pine component through regeneration and retention of old growth trees is frequently a high priority need. Often the major problem to overcome is excessive understory fuel accumulations particularly live ladder fuels, and buildup of duff around the base of desirable leave trees. Another consideration is burning to encourage the historical understory vegetation diversity. This requires burning during the growing season, which is a departure from the traditional application of prescribed fire during the spring, fall, or winter dormant seasons.

Conducting the first prescribed fire after a prolonged period of no fire must be done cautiously to avoid flare-ups in sapling thickets or rough that might kill desirable trees. For ponderosa pine, thinning of dense understories and piling and burning slash before conducting a prescribed underburn may be necessary to reduce flammability and remove competitor species that might survive most prescribed fires (Fiedler and others 1996). However, too much caution where the understory consists of thick patches of fir will result in inadequate fire. Some fuel augmentation by cutting small fir can help carry the fire with adequate intensity to kill the fir. A series of prescribed fires aimed at gradually reducing the accumulated live and dead fuels may be necessary to return stands to where maintenance underburning is easily manageable (Sackett and others 1996). The best approach to restoration must be determined on a case by base basis, but it will usually require a combination of mechanical treatments and prescribed fire repeated over a period of years.

Mixed and Stand-Replacement Regimes

The mixed fire regime includes a wide range of stand structures and landscape patterns that result from highly variable fire severities. Individual fires may be of either nonlethal understory or stand-replacement severity, or a combination of both severities. Thus, managers have considerable latitude in designing prescribed fire and silvicultural activities (fig. 9-6). Although little guidance based on past restoration efforts exists, the best way to determine restoration objectives is on a large landscape basis because of the wide latitude in individual stand structures. The challenge is to provide a diversity of stand structures with retention of snags and some coarse woody debris in forest ecosystems and unburned patches in grasslands and shrublands. In wilderness and natural area management where fires have not been previously allowed, avoiding excessive stand-replacement due to accumulated fuels may be important.

Stand-replacement fire severities can be created from either severe surface fire or crown fire. Wildfires over prolonged burning periods can leave large proportions of both severities as observed in lodgepole pine (Brown and others 1994). High severity surface fires may be more readily prescribed and achieved than crown fires due to the higher risk and fewer burning opportunities for prescribed crown fires. Ecological effects of severe surface fire and crown fire differ. Crown fire consumes foliage that otherwise would fall and protect the soil. It can kill seeds in cones, redistribute nutrients in ash, and provide more chance for regeneration by offsite colonizers. Where silvicultural objectives are being pursued, an important consideration is avoidance of excessive fragmentation caused by intensive small-scale cutting and prescribed fire activities. Provision for snags and coarse woody debris is also important.

Grazing and Exotic Plants

Introduced exotic species and grazing are two major problems that can seriously interfere with efforts to restore fire as an ecosystem process. Well-intentioned

Figure 9-6—Aspen is being successionally replaced by fir, Bridger-Teton National Forest, Wyoming. Restoration will require a stand-replacement disturbance, which could be facilitated by cutting some of the conifers.

prescribed fire, and silvicultural and rangeland enhancement activities, can fail drastically unless grazing and exotic plants are anticipated and managed properly.

Grazing—Excessive grazing can be the biggest hindrance to successful use of prescribed fire where grass vegetation is a major component, particularly in western grasslands and shrub/grass vegetation types (Wright and Bailey 1982). It is more of a problem for bunchgrasses than rhizomatous grasses (Mack and Thompson 1982). Overgrazing in the absence of fire as well as following fire can reduce plant diversity. Grazing too soon following fire can eliminate or greatly reduce desirable vegetation. In grassland areas woody plants are competitively favored, which could defeat the purpose of burning to halt woody plant encroachment.

Depending on site potential and grazing pressure, grazing should be deferred 1 to 2 years following fire in ecosystems such as sagebrush/grass and semidesert shrub (Wright and Bailey 1982). In forests such as the aspen type, intensive grazing of sprouting plants by livestock and wild ungulates, especially elk, following prescribed fires can greatly retard plant recovery. Small prescribed burns are particularly vulnerable to overutilization because of concentrated grazing (Bartos and others 1991).

Grazing prior to a prescribed burn can easily reduce fine fuels to a point where fire will not spread successfully nor have sufficient heat to ignite or kill woody plants. At least 600 lb/acre of herbaceous fuel is needed for successful prescribed fire in grassland and grass/shrub vegetation (Wright and Bailey 1982).

Exotic Plants—Fire can create favorable sites for nonindigenous plant species to become established and flourish. If exotic plants already grow in or near areas that are candidates for prescribed fire, a potential problem exists. Aggressive exotic species can competitively exclude native vegetation. Severe fires that expose large areas of mineral soil are most apt to be invaded by exotic plants; if exotics are already established, their dominance may be accelerated. Lower severity burns are more resistant to proliferation of exotics because many native species sprout and quickly occupy the site.

Cheatgrass, a nonindigenous annual that dominates millions of acres, is an extreme example of a species favored by fire. Its invasion of the sagebrush-steppe vegetation type has led to increased frequency of wildfire due to abundant, early curing fine fuels. The result is permanent conversion to annual grassland and disruption of the historic fire regime (Whisenant 1990). Another problem with nonindigenous plants can occur from seeding nonnative grasses such as annual ryegrass on severely burned sites as part of wildfire rehabilitation efforts. This practice, which is intended to stabilize soils, can delay reestablishment of native species and possibly alter long-term community composition (Conard and others 1991).

A far different problem is caused by exotics such as Chinese tallow, which has invaded coastal marshes of the Southeast. Its invasion causes a shift from grass-dominated communities to a sparse forb-dominated community that is much less flammable and acts as a fire break. Consequently, once Chinese tallow gains dominance on a site, prescribed fire cannot be effectively used to control the exotic and encroaching woody plants. Thus, the grass-dominated marsh communities are reduced.

Fire Prescriptions

Ecosystem management has brought new challenges to the application of prescribed fire primarily due to the increased scale and complexity of some prescribed burning (Zimmerman and Bunnell 1998). Traditionally, prescribed fire was applied on small, relatively homogeneous units of a single land ownership. Prescribed fire will continue to be important for small-scale operations. But to meet some ecosystem goals, prescribed fire needs to be applied over extensive areas that contain a variety of vegetation communities and fuel conditions.

In designing fire prescriptions, a strong, clear connection is needed between ecosystem goals, resource objectives, and fire objectives. This helps assure that prescribed fire will accomplish the desired effects. It can also help in choosing proper technical aids for determining the prescription and in assuring fires are cost effective and safely conducted. Designing prescriptions through a visible, logical process can also demonstrate professional competence and promote credibility of those in charge of the prescribed fire activities.

Defining fire objectives boils down to specifying first order fire effects that describe what the burning should immediately accomplish (Brown 1985b). Treatment objectives need to specify: (1) how much of what kind of organic matter should be consumed, (2) what vegetation should be killed, and (3) what the size of burned and unburned patches should be. Constraints on achieving the treatment objectives must also be considered. These can be thought of as the fire effects that should be avoided. Controlling fire, managing smoke, and avoiding overstory mortality are the common constraints. Specifying objectives and constraints is a matter of declaring what the fire should accomplish and avoid. Both are fire objectives of sorts, so why regard them differently? One reason is that it helps in demonstrating an awareness of beneficial and undesirable aspects of fire and in explaining the prescribed fire plans to others.

Depending on resource objectives, the fire objectives may call for a wide or narrow prescription window. For example, the resource objective to restore fire as a process in a nonlethal understory fire regime type may only require that prescribed fire be able to spread with minimal mortality to the overstory, an objective that could be accomplished with a wide prescription window. The specific resource objective of attaining natural regeneration while retaining some large downed woody material may call for a fire objective that specifies exposure of 20 to 30 percent mineral soil without consuming more than half of the large downed woody material. This would require a narrow prescription window.

Occasionally, conflicts may arise between fire objectives and constraints. A common example is between the objective to reduce fuels by burning at low fuel moistures and the constraint to control smoke production. Conflict can arise between different objectives; for example, to expose a high percentage of mineral soil and to leave large downed woody material for other ecosystem benefits. When conflicts arise, compromise may prevent the fire from achieving the resource objectives. It is important to recognize those situations so a potentially unsuccessful prescribed fire can be avoided.

Many technical aids are available to assist in preparing fire prescriptions. Most involve prediction of information such as weather probabilities, fuel loadings, fuel consumption, fire behavior, tree mortality, and plant response. Two technical aids—both with user guides that can help in writing and explaining prescribed fire objectives and designing fire prescriptions—are relevant for applications across the United States and much of Canada. They are the Fire Effects Information System-FEIS (Fischer and others 1996) and the First Order Fire Effects Model-FOFEM (Reinhardt and others 1997).

FEIS—This is an easy to use, computerized knowledge management system that stores and retrieves current information as text organized in an encyclopedic fashion. FEIS provides fire effects and related biological, ecological, and management information in three major categories: plant species, wildlife species, and plant communities. The plant species category includes for each species, information on taxonomy, distribution and occurrence, value and use, botanical and ecological characteristics, fire ecology, fire effects, and references. A citation retrieval system can be searched independently by author and keyword. Although the system was originally developed to meet prescribed fire needs, it is now recognized as a valuable aid for obtaining information about species ecology for any application. It can be accessed through a U.S. Forest Service Web site:

http://www.fs.fed.us/database/feis

FOFEM—This system was developed to predict the direct consequences of fire, that is, first order fire effects. FOFEM computes duff and woody fuel consumption, mineral soil exposure, fire-caused tree mortality, and smoke production for many forest and rangeland ecosystems. An update is scheduled to add soil heating effects. FOFEM contains a fire effects calculator to predict effects of fire from the burning conditions and a prescribed fire planner to compute the burn conditions necessary to achieve a desired effect. Users may enter their own fuel data or use default values derived from fuel models provided for natural and activity fuels by many forest cover types. The model is implemented in a computer program available for use on a PC or Forest Service computer. To obtain a current version of the FOFEM software, contact the authors at the Intermountain Fire Sciences Laboratory, (406) 329-4800, or PO Box 8089, Missoula, MT 59807.

Research Needs

The goals of maintaining sustainability of all ecosystem components and processes and conserving biodiversity present new challenges to land management organizations. Knowledge of how ecosystems function and what they provide is essential to making informed environmental decisions. The following broadly stated research needs indicate the knowledge required for managing fire effects on flora and fuel that will contribute to maintaining sustainable ecosystems.

Characteristics of Fire Regimes

- What is the historical range of variability in fire regime characteristics especially fire frequency, seasonality, and severity for fire dependent ecosystems? This should be answered for multiple spatial scales because of the hierarchical structure of ecosystems.
- What are the limits to ecosystem patterns and processes that signal ecosystems are beyond the boundaries of the historical range of variability?
- To what extent has climate influenced fire regime characteristics in the past? How might anticipated climate change alter fire regime characteristics in the future?

Effects of Fire on Ecosystem Processes and Biodiversity

- What are the long-term effects of fire of varying frequencies and severities on nutrient dynamics and vegetation?
- How does fire of varying frequency, seasonality, and severity influence individual plant species and plant community development? The emphasis for research should be on rare species and other vegetation components where knowledge is lacking.
- What interactions between insects and diseases and fire characterized historical fire regimes, and how has this affected landscape patterns? How do these interactions change when ecosystems exceed the natural range of variability and when various management activities are applied?
- What is the interaction of different ecosystem scales on ecosystem processes and biodiversity? To what extent can coarse scale analysis account for ecosystem processes and biodiversity?
- What are the long-term effects of largely excluding fire from ecosystems that evolved under fire regimes?

Restoration of Ecosystems

- What approaches and methods involving wildland fire use, prescribed fire, silviculture, and grazing can be used to restore ecosystems to a semblance of the historical range of vegetation composition and structure while meeting the resource needs of society?
- What fuel management activities can provide an acceptable level of fire hazard and remain compatible with ecosystem goals, especially needs for coarse woody debris?
- How can nonindigenous plant species be managed in combination with prescribed fire and resource utilization activities to maintain biodiversity?

Development of Ecosystem Evaluation Methodologies

- Continue with development of simulation models and ecosystem evaluation techniques that can help in understanding and managing ecosystem dynamics. Succession and landscape models are needed that account for interaction of fire, vegetation, fuels, and climate.
- Fire effects models at small spatial and temporal scales are needed for rigorous fire effects hypothesis testing and as building blocks for models with larger temporal and spatial scales.
- Determine organizational approaches that allow complex ecosystem models requiring specialized skills and high speed computer facilities to be accessible to all land management organizations and units.

Notes

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Appendices

Appendix A: Common and Scientific Names of Plant Species

Common names mentioned in the text and scientific names follow the nomenclature of the U.S. Department of Agriculture (1999) PLANTS database except for some wet grassland species, which follow the Integrated Taxonomic Information System (http://www.itis.usda.gov/). For some species a second common name is shown because it is commonly used.

Common name	Scientific name
Trees and Shrubs	
a'ali'i	*Dodonaea viscosa*
Alaska-cedar	*Chamaecyparis nootkatensis*
alder	*Alnus* spp.
alligator juniper	*Juniperus deppeana*
American beech, beech	*Fagus grandifolia*
American elm, white elm	*Ulmus americana*
American mountain-ash	*Sorbus americana*
antelope bitterbrush	*Purshia tridentata*
Arizona pine	*Pinus ponderosa* var. *arizonica*
Arizona white oak	*Quercus arizonica*
Ashe juniper	*Juniperus ashei*
aspen, trembling aspen, quaking aspen	*Populus tremuloides*
Atlantic white-cedar	*Chamaecyparis thyoides*
baccharis	*Baccharis* spp.
baldcypress	*Taxodium distichum*
balsam fir	*Abies balsamea*
balsam poplar	*Populus balsamifera*
basin big sagebrush	*Artemisia tridentata* ssp. *tridentata*
basswood	*Tilia americana*
batis	*Batis maritima*
bayberry	*Morella* spp.
beaked hazel	*Corylus cornuta*
bear oak	*Quercus ilicifolia*
Bebb willow	*Salix bebbiana*
big sagebrush	*Artemisia tridentata*
bigleaf maple	*Acer macrophyllum*
bigleaf sumpweed	*Iva frutescens*
bitter cherry	*Prunus emarginata*
bitternut hickory	*Carya cordiformis*
black ash	*Fraxinus nigra*
black cherry	*Prunus serotina*
black cottonwood	*Populus trichocarpa*
black greasewood	*Sarcobatus vermiculatus*
black mangrove	*Avicennia germinans*
black oak	*Quercus velutina*
black spruce	*Picea mariana*
black walnut	*Juglans nigra*
blackbrush	*Coleogyne ramosissima*
blackgum, black tupelo	*Nyssa sylvatica*
blackjack oak	*Quercus marilandica*

Common name	Scientific name
blue huckleberry	*Vaccinium membranaceum*
blue oak	*Quercus douglasii*
blue spruce	*Picea pungens*
bluejack oak	*Quercus incana*
bog blueberry	*Vaccinium uliginosum*
bog labrador tea	*Ledum groenlandicum*
boxelder, Manitoba maple	*Acer negundo*
Brazilian pepper	*Schinus terebinthifolus*
buckeye	*Aesculus*
buckwheat tree	*Cliftonia monophylla*
bur oak	*Quercus macrocarpa*
bursage spp.	*Ambrosia* spp.
butternut	*Juglans cinerea*
buttonwood, button mangrove	*Conocarpus erectus*
cabbage palmetto	*Sabal palmetto*
California black oak	*Quercus kelloggii*
California red fir	*Abies magnifica*
California sagebrush	*Artemisia californica*
canyon live oak	*Quercus chrysolepis*
Caribbean pine, Honduras pine	*Pinus caribaea*
Carolina ash, pop-ash	*Fraxinus caroliniana*
cenzia, purple sage	*Leucophyllum frutescens*
chamise	*Adenostoma fasciculatum*
Chapman oak	*Quercus chapmanii*
chestnut oak	*Quercus prinus*
Chinese tallow	*Sapium sebiferum*
chokecherry	*Prunus virginiana*
coast Douglas-fir	*Pseudotsuga menziesii* var. *menziesii*
coast live oak	*Quercus agrifolia*
coastalplain staggerbush	*Lyonia fruticosa*
cocoplum	*Chrysobalanus icaco*
common persimmon	*Diospyros virginiana*
creeping barberry	*Mahonia repens*
creosotebush	*Larrea tridentata*
cypress	*Taxodium* spp.
cyrilla, swamp cyrilla	*Cyrilla racemiflora*
dahoon	*Ilex cassine*
digger pine, California foothills pine	*Pinus sabiniana*
Douglas-fir	*Pseudotsuga menziesii*
dwarf chinkapin oak	*Quercus prinoides*
dwarf huckleberry	*Gaylussacia dumosa*
eastern baccharis, groundsel-tree	*Baccharis halimifolia*
eastern cottonwood	*Populus deltoides*
eastern hemlock	*Tsuga canadensis*
eastern redcedar	*Juniperus virginiana*
eastern white pine	*Pinus strobus*
Engelmann spruce	*Picea engelmannii*
eucalyptus	*Eucalyptus* spp.
fetterbush	*Lyonia lucida*
flowering dogwood	*Cornus florida*
forage kochia	*Kochia prostrata*
fourwing saltbush	*Atriplex canescens*
Fraser fir	*Abies fraseri*
gallberry, inkberry	*Ilex glabra*
Gambel oak	*Quercus gambelii*

Common name	Scientific name
giant sequoia	*Sequoiadendron giganteum*
gooseberry, currant	*Ribes* spp.
grand fir	*Abies grandis*
gray birch	*Betula populifolia*
green ash	*Fraxinus pennsylvanica*
greenbriar	*Smilax glauca, Smilax* spp.
ground blueberry	*Vaccinium myrsinites*
hickory	*Carya* spp.
hoaryleaf ceanothus	*Ceanothus crassifolius*
honey mesquite	*Prosopis glandulosa*
horsebrush	*Tetradymia* spp.
incense-cedar	*Calocedrus decurrens*
interior live oak	*Quercus wislizenii*
interior ponderosa pine	*Pinus ponderosa* var. *scopulorum*
jack pine	*Pinus banksiana*
Jeffrey pine	*Pinus jeffreyi*
Joshua tree	*Yucca brevifolia*
juniper	*Juniperus* spp.
koa	*Acacia koa*
large gallberry	*Ilex coriacea*
laurel oak	*Quercus laurifolia*
leatherleaf	*Chamaedaphne calyculata*
live oak	*Quercus virginiana*
loblolly pine	*Pinus taeda*
loblolly-bay	*Gordonia lasianthus*
lodgepole pine	*Pinus contorta*
longleaf pine	*Pinus palustris*
lyonia	*Lyonia* spp.
manzanita	*Arctostaphylos* spp.
melaleuca	*Melaleuca quinquenervia*
mesquite	*Prosopis* spp.
mockernut hickory	*Carya tomentosa*
Mormon tea	*Ephedra torreyana*
mountain alder	*Alnus incana*
mountain big sagebrush	*Artemisia tridentata* ssp. *vaseyana*
mountain hemlock	*Tsuga mertensiana*
mountain-laurel	*Kalmia latifolia*
myrsine	*Myrsine quianensis*
myrtle oak	*Quercus myrtlifolia*
myrtle, wax myrtle, southern bayberry	*Morella cerifera*
noble fir	*Abies procera*
northern pin oak	*Quercus ellipsoidalis*
northern red oak	*Quercus rubra*
northern white-cedar, e. white-cedar	*Thuja occidentalis*
oneseed juniper	*Juniperus monosperma*
oak	*Quercus* spp.
Oregon white oak	*Quercus garryana*
Pacific madrone	*Arbutus menziesii*
Pacific ponderosa pine	*Pinus ponderosa* var. *ponderosa*
Pacific silver fir	*Abies amabilis*
paloverde spp.	*Cercidium* spp.
paper birch	*Betula papyrifera*
persimmon	*Diospyros* spp.
piedmont staggerbush	*Lyonia mariana*
pignut hickory	*Carya glabra*

Common name	**Scientific name**
pin cherry, fire cherry	*Prunus pensylvanica*
pin oak	*Quercus palustris*
pinyon pine	See singleleaf pinyon, true pinyon
pitch pine	*Pinus rigida*
pond cypress	*Taxodium ascendens*
pond pine	*Pinus serotina*
ponderosa pine	*Pinus ponderosa*
poplar	*Populus* spp.
post oak	*Quercus stellata*
rabbitbrush	*Chrysothamnus* spp.
raspberry, blackberry	*Rubus* spp.
red alder	*Alnus rubra*
red bay	*Persea borbonia*
red elderberry	*Sambucus racemosa* ssp. *pubens*
red mangrove	*Rhizophora mangle*
red maple	*Acer rubrum*
red pine	*Pinus resinosa*
red raspberry	*Rubus idaeus*
red spruce	*Picea rubens*
redberry juniper	*Juniperus erythrocarpa*
redstem ceanothus	*Ceanothus sanguineus*
redwood	*Sequoia sempervirens*
rhododendron	*Rhododendron* spp.
Rocky Mountain Douglas-fir	*Pseudotsuga menziesii* var. *glauca*
Rocky Mountain juniper	*Juniperus scopulorum*
Rocky Mountain lodgepole pine	*Pinus contorta* var. *latifolia*
Rocky Mountain maple	*Acer glabrum*
rosemary	*Ceratiola ericoides*
rusty staggerbush	*Lyonia ferruginea*
sagebrush	*Artemisia* spp.
salmonberry	*Rubus spectabilis*
sand live oak	*Quercus virginiana* var. *maritima*
sand pine	*Pinus clausa*
sand post oak	*Quercus stellata* var. *margaretta*
sand shinnery oak	*Quercus havardii*
Saskatoon serviceberry	*Amelanchier alnifolia*
saw palmetto	*Serenoa repens*
scarlet oak	*Quercus coccinea*
scrub oak	*Quercus dumosa*
scrub palmetto	*Sabal etonia*
shadscale	*Atriplex confertifolia*
sheep-laurel	*Kalmia angustifolia*
shore pine	*Pinus contorta* var. *contorta*
shortleaf pine	*Pinus echinata*
silver maple	*Acer saccharinum*
silver sagebrush	*Artemisia cana*
silverbell	*Halesia Ellis*
singleleaf pinyon	*Pinus monophylla*
Sitka spruce	*Picea sitchensis*
slash pine	*Pinus elliottii*
snakeweed	*Gutierrezia* spp.
snowbrush ceanothus	*Ceanothus velutinus*
sourwood	*Oxydendrum arboreum*
southern magnolia	*Magnolia grandiflora*

Common name	Scientific name
southern red oak, cherrybark oak	*Quercus falcata*
speckled alder	*Alnus rugosa*
spiny hopsage	*Grayia spinosa*
spruce pine	*Pinus glabra*
subalpine fir	*Abies lasiocarpa*
sugar maple	*Acer saccharum*
sugar pine	*Pinus lambertiana*
sugarberry	*Celtis laevigata*
swamp bay	*Persea palustris*
swamp chestnut oak	*Quercus michauxii*
swamp tupelo	*Nyssa biflora*
sweetbay	*Magnolia virginiana*
sweetgum	*Liquidambar styraciflua*
sweetpepperbush, poor man's soap	*Clethra alnifolia*
sycamore	*Platanus occidentalis*
Table Mountain pine	*Pinus pungens*
tamarack	*Larix laricina*
tanoak	*Lithocarpus densiflora*
tarbush	*Flourensia cernua*
Texas persimmon	*Diospyros texana*
thimbleberry	*Rubus parviflorus*
thinleaf alder	*Alnus incana* ssp. *tenuifolia*
threetip sagebrush	*Artemisia tripartita*
toyon	*Heteromeles arbutifolia*
trefoil	*Lotus* spp.
true pinyon, Colorado pinyon	*Pinus edulis*
turkey oak	*Quercus laevis*
Utah juniper	*Juniperus osteosperma*
varnish leaf	*Dodonea virginiana*
Virginia pine	*Pinus virginiana*
water oak	*Quercus nigra*
water tupelo	*Nyssa aquatica*
western hemlock	*Tsuga heterophylla*
western juniper	*Juniperus occidentalis*
western larch	*Larix occidentalis*
western redcedar	*Thuja plicata*
western white pine	*Pinus monticola*
white ash	*Fraxinus americana*
white bully	*Sideroxylon salicifolium*
white bursage	*Ambrosia dumosa*
white fir	*Abies concolor*
white mangrove	*Laguncularia racemosa*
white oak	*Quercus alba*
white sage	*Salvia apiana*
white spirea	*Spiraea betulifolia*
white spruce	*Picea glauca*
whitebark pine	*Pinus albicaulis*
willow	*Salix* spp.
willow oak	*Quercus phellos*
winterfat	*Krascheninnikovia lanata*
Wyoming big sagebrush	*Artemisia tridentata* ssp. *wyomingensis*
yaupon	*Ilex vomitoria*
yellow birch	*Betula alleghaniensis*

Common name	Scientific name
yellow buckeye	*Aesculus octandra*
yellow paloverde	*Cercidium microphyllum*
yellow-poplar	*Liriodendron tulipifera*
Grasses and Forbs	
alligatorweed	*Alternanthera philoxeroides*
American white waterlily	*Nymphaea odorata*
annual fleabane	*Erigeron annuus*
annual ryegrass, Italian ryegrass	*Lolium perenne* spp. *multiflorum*
arrowhead	*Sagittaria* spp.
arrowleaf balsamroot	*Balsamorhiza sagittata*
basin wildrye	*Leymus cinereus*
beakrush, Tracy's beaksedge	*Rhynchospora tracyi*
big bluestem	*Andropogon gerardii* var. *gerardii*
black grama	*Bouteloua eriopoda*
blackeyed Susan	*Rudbeckia hirta*
blue grama	*Bouteloua gracilis*
bluebunch wheatgrass	*Pseudoroegneria spicata*
blue-eyed grass	*Sisyrinchium campestre*
bluestem spp.	*Schizachyrium* spp.
bottlebrush squirreltail	*Elymus elymoides*
broadleaf arnica	*Arnica latifolia*
brome	*Bromus* spp.
broomsedge, broomsedge bluestem	*Andropogon virginicus*
buffalograss	*Buchloe dactyloides*
bulltongue arrowhead	*Sagittaria lancifolia*
Burma reed	*Neyrundia reynaudiana*
bush muhly	*Muhlenbergia porteri*
cane, switch cane	*Arundinaria gigantea*
cattail	*Typha* spp.
chairmaker's bullrush	*Schoenoplectus americanus*
chalky bluestem	*Andropogon capillipes*
cheatgrass	*Bromus tectorum*
cogongrass	*Imperata cylindrica*
Columbian bluestem, bush beardgrass	*Schizachyrium condensatum*
common camas	*Camassia quamash*
cordgrass	*Spartina* spp.
curly-mesquite	*Hilaria belangeri*
Curtis' dropseed	*Sporobolus curtissii*
cutgrass	*Zizaniopsis* spp.
deathcamas	*Zigadenus venenosus*
dropseed, sacaton	*Sporobolus* spp.
fescue	*Festuca* spp.
fireweed	*Epilobium angustifolium*
fountain grass	*Pennisetum sataceum*
gayfeather	*Liatris* spp.
glacier lily	*Erythronium grandiflorum*
golden brodiaea	*Brodiaea ixioides*
green arrow arum	*Peltandra virginica*
gulf cordgrass	*Spartina spartinae*
hairawn muhly	*Muhlenbergia capillaris*
heartleaf arnica	*Arnica cordifolia*

Common name	Scientific name
hydrocotyle, marshpennywort	*Hydrocotyle* spp.
Idaho fescue	*Festuca idahoensis*
Indian paintbrush	*Castilleja* spp.
Indian ricegrass	*Oryzopsis hymenoides*
Indiangrass	*Sorghastrum nutans*
inland saltgrass	*Distichlis spicata*
Lehmann lovegrass	*Eragrostis lehmanniana*
little bluestem	*Andropogon scoparius*
little bluestem, creeping bluestem	*Schizachyrium scoparium*
lupine	*Lupinus* spp.
maidencane	*Panicum hemitomon*
milkvetch	*Astragalus* spp.
molasses grass	*Melinis minutiflora*
muhly spp.	*Muhlenbergia* spp.
natal redtop, rose natalgrass	*Melinis repens*
needlegrass rush, black rush	*Juncus roemerianus*
panicum	*Panicum* spp.
pasque flower	*Anemone pratens*
pickerelweed	*Pontederia cordata*
pickleweed	*Salicornia* spp.
pili grass	*Heteropogon contortus*
pinegrass	*Calamagrostis rubescens*
pineland threeawn, wiregrass	*Aristida stricta*
pitcherplant	*Sarracenia purpurea*
pond lily	*Nuphar* spp.
prairie cordgrass	*Spartina pectinata*
prairie violet	*Viola pedatifida*
red brome	*Bromus rubens*
reed	*Phragmites* spp.
rough fescue	*Festuca scabrella*
rush	*Juncus* spp.
saltgrass	*Distichlis* spp.
saltmeadow cordgrass	*Spartina patens*
saltmeadow rush	*Juncus gerardii*
sand cordgrass	*Spartina bakeri*
sand dropseed	*Sporobolus cryptandrus*
Sandberg bluegrass	*Poa secunda*
sawgrass	*Cladium* spp.
seaside tansy	*Borrichia* spp.
sheathed cottonsedge	*Eriophorum vaginatum*
showy aster	*Aster conspicuus*
showy partridgepea	*Cassia fasciculata*
sideoats grama	*Bouteloua curtipendula*
sky blue aster	*Aster azureus*
slender bluestem	*Schizachyrium tenerum*
smooth cordgrass	*Spartina alterniflora*
spikerush	*Eleocharis* spp.
star-flowered Solomon’s seal	*Smilacina stellata*
strawberry	*Fragaria* spp.
switchgrass	*Panicum virgatum*
thatching grass	*Hyparrhenia rufa*
Thurber’s needlegrass	*Achnatherum thurberiana*

Common name	**Scientific name**
tobosa	*Pleuraphis mutica*
trumpet pitcherplant, yellow pitcherplant	*Sarracenia flava*
twinflower	*Linnaea borealis*
waterlily	*Nymphaea* spp.
western wheatgrass	*Pascopyrum smithii*
western yarrow	*Achillea millefolium*
wheatgrass	*Agropyron* spp.
wild columbine	*Aquilegia canadensis*
wild hollyhock	*Iliamna rivularis*
wild sarsaparilla	*Aralia nudicaulis*
wildrice	*Zizania* spp.
wiregrass	*Aristida* spp.
Mosses, Ferns, Cactus, and Lichens	
bracken fern	*Pteridium aquilinum*
cholla, prickly pear	*Opuntia* spp.
cup lichen	*Cladonia* spp.
dicranum	*Dicranum* spp.
knight’s plume moss	*Ptilium crista-castrensis*
mountain fern moss	*Hylocomium splendens*
Old world or small-leaf climbing fern	*Lygodium microphyllum*
prickly pear cactus	*Opuntia humifusa*
saguaro	*Carnegia gigantea*
Schreber’s moss	*Pleurozium schreberi*
sphagnum	*Sphagnum* spp.

Appendix B: Succession Simulation Models

FIRESUM–FIRE SUcession Model is a stand level, individual tree ecosystem process model developed for western conifers, especially ponderosa pine and whitebark pine, to simulate effects of different fire regimes on tree composition, stand structure, and fuel loadings (Keane and others 1990, 1989).

FVS–Forest Vegetation Simulation is a stand level mensurational model (Wykoff and others 1982).

FFE–Fire and Fuels Extension to the FVS is a stand level model for simulating surface fuel loadings, tree characteristics, expected fire behavior, and expected tree mortality (Beukema and others 1999). This model is particularly useful for growth, mortality, falldown, and decay of conifer trees.

Fire-BGC–The Fire Biogeochemical Mechanistic process model can be used to investigate stand- and landscape-level changes in ecosystem processes and characteristics in fire-dominated environments (Keane and others 1996). It is useful for evaluating effects of climate change.

SIMPPLLE–Simulating Patterns and Processes at Landscape Scales is a rule-based model designed for Northern Rocky Mountain forest types (Chew 1997). It starts at a coarse scale and adds fine scale only as needed to produce acceptable performance.

VDDT—The Vegetation Dynamics Development Tool is a nonspatial, deterministic model where successional pathways connecting successional stages are used to explore community dynamics (Beukuma and Kurz 1995). VDDT only simulates one vegetation type at a time but is useful and efficient for simulating disturbance and succession on mid-scale to fine-scale landscapes. This model can be readily used by managers and is undergoing development for national applications.

CRBSUM–The Columbia River Basin Succession Model was used to simulate landscape changes for the Interior Columbia Basin Ecosystem Management Project (Keane and others 1996). It incorporates disturbance as a stochastic process and models succession for individual landscape pixels. CRBSUM 2 was created from CRBSUM to improve the simulation of fire processes over time and space (Keane and Long 1998).

LANDSUM–the Landscape Succession Model was derived from CRBSUM to operate on a polygon level rather than pixels. This allows it to be used at finer scales of resolution (Keane 1999).

FIREPAT–Fire Pattern Succession Model attempts to more realistically model fire by simulating fire ignition and size to compute number of pixels disturbed by fire (Keane and Long 1998). It operates at a coarse scale and models succession similar to CRBSUM.

INTELAND–Integrated Terrestrial Landscape Model simulates natural processes in boreal forest ecosystems (Gauthier and others 1998). It was designed as a GIS-based model to help define natural system baselines for disturbance regimes, vegetation dynamics, wildlife species composition, and landscape diversity.

Appendix C: Glossary

The terminology here was derived from the following references: fuels and fire behavior from Agee (1993), Brown and others (1982), and Ryan and Noste (1985); fire occurrence from Agee (1993), Johnson (1992), and Romme (1980); and plant reproduction from Allaby (1992), Helms (1998), and Sutton and Tinus (1983).

Fuels

Fuel comprises living and dead vegetation that can be ignited. It is often classified as dead or alive and as natural or activity fuels. Fuel components refer to such items as downed dead woody material by various size classes, litter, duff, herbaceous vegetation, live foliage, live shrub stems and so forth.

Kinds of Fuel

dead fuels: Fallen dead vegetation such as downed woody material, litter, duff, and organic soils and dead upright vegetation such as cured grasses, forbs, and dead attached shrub stemwood.

live fuels: Living plants. Especially important components include tree crowns, shrubs, grasses, forbs, and ferns.

natural fuels: These result from plant growth and death, loss of foliage, branch breakage, and tree blowdown.

activity fuels: These fuels result from human activity such as logging, thinning, chaining, and herbicide use. It usually refers to residues from cutting operations.

down, dead woody fuels: Dead twigs, branches, stems, and boles of trees and shrubs that have fallen and lie on or near the ground (Brown and others 1982). Wood includes sound and rotten components.

litter: The top layer of the forest floor (01 soil horizon), including freshly fallen leaves, needles, fine twigs, bark flakes, fruits, matted dead grass and a variety of miscellaneous vegetative parts that are little altered by decomposition. Litter also accumulates beneath rangeland shrubs. Some surface feather moss and lichens are considered to be litter because their moisture response is similar to dead fine fuel. In grasslands, litter is the accumulated dead herbaceous material usually referred to as thatch.

duff: Partially decomposed organic matter lying beneath the litter layer and above the mineral soil. It includes the fermentation and humus layers of the the forest floor (02 soil horizon).

organic soils: The deep layers of organic matter that frequently develop in poorly drained areas such as bogs, swamps, and marshes.

Fuel Properties

loading: The weight per unit area of fuel often expressed in tons per acre or tonnes per hectare. Dead woody fuel loadings are commonly described for small material in diameter classes of 0 to 0.25, 0.25 to 1, and 1 to 3 inches and for large material greater than 3 inches.

fuel continuity: A qualitative description of the distribution of fuel both horizontally and vertically. Continuous fuels readily support fire spread. The larger the fuel discontinuity, the greater the fire intensity required for fire spread.

total fuel: The amount of biomass that potentially could burn.

available fuel: The amount of biomass that will burn under a given set of conditions. Moisture content and fuel size are the primary determinants of availability. Arrangement and compactness of fuel may also determine availability.

fuel moisture content: This is expressed as a percent or fraction of oven dry weight of fuel. It is the most important fuel property controlling flammability. In living plants it is physiologically bound. Its daily fluctuations vary considerably by species but are usually above 80 to 100 percent. As plants mature, moisture content decreases. When herbaceous plants cure, their moisture content responds as dead fuel moisture content, which fluctuates according to changes in temperature, humidity, and precipitation.

Fire Behavior

type of fire: Refers to the fuels that are primarily supporting the fire namely surface fires, ground fires, and crown fires.

surface fires: These fires burn in litter and other live and dead fuels at or near the surface of the ground mostly by flaming combustion.

ground fires: These burn in the organic material below the litter layer mostly by smoldering combustion. Fires in duff, peat, dead moss and lichens, and punky wood are typically ground fires.

crown fires: These burn in the crowns of trees and shrubs usually ignited by a surface fire. They are common in coniferous forests and chaparral type shrublands.

fireline intensity: Also called Byram's intensity, this is the rate of energy release per unit length of the fire front expressed as BTU per foot of fireline per second or as kilowatts per meter of fireline. This expression is commonly used to describe the power of wildland fires.

flame length: The length of flames in the propagating fire front measured along the slant of the flame from the midpoint of its base to its tip. It is mathematically related to fireline intensity and tree crown scorch height.

total heat release: The heat released by combustion during burnout of all fuels in BTU per square foot or kilocalories per square meter.

fire duration: The length of time that combustion occurs at a given point. It relates closely to downward heating and fire effects below the fuel surface as well as heating of tree boles above the surface.

ground char: A qualitative measure of a fire's heat pulse downward into the soil. It is determined by visually judging the extent of fuel consumption, charring, and changes in soil texture (Ryan and Noste 1985). It is also referred to as **burn severity** and **depth of burn**, which is a quantitative expression of depth of forest floor consumed by fire. It is largely determined by fire duration and characteristics of the soil including the forest floor.

fire severity: A qualitative measure of the immediate effects of fire on the ecosystem. It relates to the extent of mortality and survival of plant and animal life both aboveground and belowground and to loss of organic matter. It is determined by heat released aboveground and belowground. Ryan and Noste (1985) describe a method for rating fire severity based on flame length and depth of burn.

Fire Occurrence

The definitions here were based on a review of fire history terminology at a fire history workshop (Romme 1980) and phraseology by Agee (1993).

fire cycle: A fire return interval calculated using a negative exponential distribution, applied using current age-class structure on the landscape. It is the average stand age of a forest characterized using the negative exponential distribution.

fire frequency: A general term referring to the recurrence of fire in a given area over time. It is sometimes stated as number of fires per unit time in a designated area. It is also used to refer to the probability of an element burning per unit time (Johnson 1992).

fire rotation: The length of time necessary for an area equal in size to the study area to burn and is equal to the **fire cycle**.

mean fire-return interval: The arithmetic average of all fire intervals in a given area over a given time period.

Plant Reproduction

axil: The upper side of the point where a leaf meets a stem, or a branch meets another branch or the main stem of a plant.

bulb: An underground storage organ that bears roots on its lower surface and fleshy leaves above. It provides a means of reproduction in perennials.

burl: A mass of woody tissue or wartlike structure fomed on stem, branch, or root; has numerous buds, which rarely develop further.

caudex: A largely underground woody stem base of an otherwise herbaceous perennial forb that produces leaves and flowering stems.

corm: An underground storage organ bearing adventitious roots and scale leaves. It may function as an organ of vegetative reproduction in perennials.

epicormic branch: A shoot arising spontaneously from an adventitious or dormant bud on the stem or branch of a woody plant often following exposure to increased light levels or fire.

lignotuber: A woody organ of food storage and regeneration forming a swelling of stem material, more or less at ground level, that originates from the axils of cotyledons or early seedling leaves.

rhizome: A creeping stem, not a root, growing beneath the surface consisting of a series of nodes with roots commonly produced from the nodes and producing buds in the leaf axils.

root collar: Loosely, the point along the main stem-root axis at which the primary vascular anatomy changes from that of a stem to that of a root, usually applied to trees. Transition point between stem and root. It may be clearly or vaguely apparent.

root crown: A mass of woody tissue from which roots and stems originate, usually applied to shrubs and herbaceous plants; can be considered as the point at which root and stem meet.

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RMRS
ROCKY MOUNTAIN RESEARCH STATION

The Rocky Mountain Research Station develops scientific information and technology to improve management, protection, and use of the forests and rangelands. Research is designed to meet the needs of National Forest managers, Federal and State agencies, public and private organizations, academic institutions, industry, and individuals.

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*Station Headquarters, Natural Resources Research Center,
2150 Centre Avenue, Building A, Fort Collins, CO 80526