CHAPTER 11

Northeastern Plateaus Bioregion

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> The only means for taking deer wholesale was by kupi't (firing) and was practiced in late summer, about the middle of August. When deer were sighted on a hill, a group of hunters hastened there, some on either side of the mountain. They started fires, working them around until the band was completely encircled. This accomplished the fires were brought closer, constricting the circle until the animals were bunched on the crest on a hill where they could be shot conveniently. Deer firing was ordinarily executed without undue noise, but if a bear were accidentally caught in the fire, a great hue and cry was set up to warn others and to inform them which direction the animal was headed.
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> *Fire use by Northern Paiute Indians, as told to* ISABEL T. KELLY, 1932

Description of the Bioregion

Northeastern California landscape is a mixture of vast arid basins and uplands, and forested mountain ranges interspersed with both fresh water and alkaline wetlands. The entire bioregion is significantly influenced by the rain shadow effect of the Cascade Range to the west. Three ecological unit subsections are treated in this chapter: (1) Modoc Plateau Section (M261G), (2) northwestern Basin and Range Section (342B), and (3) extreme northern portion of the Mono Section (341Dk). The Cascade Range defines the western edge of the Modoc Plateau, and the Sierra Nevada defines the western boundary for the southern portion of the northwestern Basin and Range and Mono Sections. There is no one overriding feature that cleanly depicts this bioregion to the north and east as the political borders of the states of Oregon and Nevada, respectively, define the boundaries (Map 11.1).

Physical Geography

The Northeastern Plateaus bioregion extends east of the southern Cascade Range and south along the east slopes of the Sierra Nevada. This region represents California's portion of the Great Basin. The climate is dry, cold, and continental. Volcanism dominates the geology, topography, and geomorphic processes of the area. The most conspicuous feature of the region is the Modoc Plateau, a high, flat terrain characterized by basalt plains and volcanic shields (MacDonald and Gay 1966). The topography is extremely abrupt and elevations, ranging from 1,204 m (3,950 ft) to 3,016 m (9,892 ft), can change quickly. The Pit River, with headwaters in the Warner Mountains and, until 1877, Goose Lake, traverses the Modoc Plateau from east to west where it enters the Cascade Range and flows into the Sacramento River (Pease 1965). The Modoc Plateau is also scattered with seasonal ponds and pluvial lakebeds. Several rivers, including the Susan, which drains into the Honey Lake basin, and the Carson and Walker in the northern Mono section, flow from west to east and transect the steppe.

Several vegetation zones occur in the Northeastern Plateaus bioregion. The general sequence, from the lowest elevations to the highest, are: sagebrush steppe (includes Wyoming big sagebrush and western juniper series zones), lower montane (ponderosa pine series zone), mid-montane (lower-elevation white fir series zone), upper montane (upper-elevation white fir series), and subalpine (whitebark pine series zone). The bioregion as defined in this publication is too dry and cold to support red fir *(Abies magnifica)* and mountain hemlock *(Tsuga mertensiana),* and only a few Douglas-fir *(Pseudotsuga menziesii)* or sugar pines *(Pinus lambertiana)* are found on the extreme western edge of this bioregion. Several species reach the extent of their ranges in this area, as detailed below (Smith and Davidson 2003).

The volcanism that shaped the area is fairly recent, dating from the Pliocene to Pleistocene eras, and includes volcanic ash deposits, lava flows, talus jumbles, obsidian deposits, and lava rims. Soils that have weathered in place, particularly at the lowest elevations of the region, are often poorly

developed, shallow, and rocky. Clays weathered from basalt are a typical soil texture component in many locations. River deposits include Mollisols and Aridisols, and lakebed deposits include silt, sandstones, and diatomite (Young et al. 1988). At higher elevations in the bioregion, most residual soils are Mollisols, often weathered from basalt parent materials. Soils transported to lower-lying areas via alluvial processes are often Vertisols. Soil temperature regimes are mesic, frigid, and cryic, and soil moisture regimes are mostly xeric and aridic.

Climatic Patterns

Climate data in northeastern California is somewhat inaccurate because of historically few and somewhat widely spaced long-term climate stations confounding the difficulty in interpolating the marked rain shadow effects from topographic features. The PRISM climate model (Daly et al. 1994) uses extrapolations from weather station data points, and if the data are sparse then errors result in the extrapolations. In arid climates such as northeastern California, slight differences in precipitation and temperature show up in vegetation expression more readily than in wetter climates, and current climate models sometimes fail to predict this. We expect this will change in the next couple of decades as there are now more and spatially denser automated stations that should result in better predictive climate, vegetation, and fire models. In the meantime, structured observations of vegetation on zonal sites (sites reflecting the regional climate more than local topography or soils) are the best way to infer the climate of this bioregion.

FIRE CLIMATE VARIABLES

The bioregion is buffered from Pacific storms by being in the rain shadow of the Cascade Range and Sierra Nevada. Most of the precipitation occurs between October and May; with most occurring between November and April as snow. The summer months from May to October are mostly rainless with the exception of summer thunderstorms. Summer thunderstorms can be locally significant and are the source of lightning ignitions. The range of annual precipitation in the Northeastern Plateaus bioregion is from 17.8 to 121.9 cm (7–48 in) (Daly et al. 1994). The overall bioregional average is 43.2 cm (17 in). However, local orographic enhancement boosts precipitation from 91.4 to 121.9 cm (36–48 in) in the northern Warner Mountains. Temperatures range from January minimums of −11°C to −2°C (12°F–28°F) to July maximums of 21°C to 32°C (69°F–89°F). The majority of fires occur from June to September, and certain fire weather conditions can affect fire ignitions and fire behavior during those months.

WEATHER SYSTEMS

There are four basic fire weather patterns that can significantly affect fire behavior and natural ignitions in northeastern California during the May-to-October fire season. These are: (1) the Pre-Frontal Winds, (2) Lightning with Low Precipitation, (3) Moist Monsoon, and (4) Strong Subsidence/Low Relative Humidity patterns. The pre-frontal and subsidence patterns mostly affect fire behavior and spread, whereas the lightning with low precipitation and moist monsoon patterns are important for their fire ignition potential.

Pre-Frontal Winds Pre-frontal wind events are frequent in springtime and again in late summer and fall. They are of most consequence in the latter period, when both live and dead fuel moistures are low. The pre-frontal weather pattern occurs in advance of spring or fall cold fronts, or upper low-pressure troughs. Tight pressure gradients aloft produce moderate to strong south to west winds, which can surface strongly on the lee side of the Cascade Range and Sierra Nevada. As the downward-moving air masses lose elevation, they gain temperature and lose humidity due to adiabatic compression. These phenomena are combined with moderate to strong wind speeds of 24 to 56 km h^{-1} (15–35 mi h^{-1}), with gusts as high as 80 km h^{-1} (50 mi h^{-1}). This pattern usually occurs between 5 and 10 times a year, with one or two significant events during the fall season of most years. These conditions can lead to rapid fire spread and extreme fire behavior.

Lightning with Low Precipitation This weather pattern typically affects northeastern California to varying degrees, once or twice per fire season, although it can occur as many as four or five times in a single year. Though not a term generally recognized by meteorologists, local fire and resource mangers use "dry lightning" to describe this pattern because it includes episodes of thunderstorms that are more likely to cause fires than others. The pattern is most common in July and August, but can occur from June through mid-September. It sets up when the dominant high pressure aloft becomes oriented from roughly northern Nevada down through Arizona. When the main high-pressure center is in northern Nevada, it draws mid-level moisture northwestward, tracking along the western flank of the high-pressure mass toward northern California. Often, this moisture transport roughly parallels the Sierra Nevada, with the greatest moisture from near the crest eastward. When relatively cool disturbances aloft are caught up in this larger-scale flow pattern they can act as "triggers," destabilizing the mid and upper layers enough to generate high-based thunderstorms. Because the resulting cells have high bases, much of the precipitation associated with them evaporates before reaching the ground. The events usually result in many fire ignitions over a relatively short time, a situation that can be rapidly compounded by the gusty erratic downdraft winds associated with the thunderstorms (Rorig and Ferguson 1999, 2002).

Moist Monsoon The moist monsoon pattern is the "true" southwestern monsoon, and it is rare in the Northeastern Plateaus and the rest of northern California. The pattern is characterized by unstable air masses, high humidity, and fairly widespread wet thunderstorm activity. Although it is a

MAP 11.1. Northeastern Plateaus bioregion map. Note the inclusion of Carson Valley and the Pine Nut Mountains in the northern Mono sections of the Great Basin, southeast of Lake Tahoe.

common summer feature from Arizona/New Mexico northward into the Great Basin and southern Rocky Mountains, northeastern California usually lies just beyond its western fringes. About once in 10 to 15 years the pattern is displaced far enough westward to affect northeastern California. When it does so, sites can receive a total of 5 to 12.5 cm (2–5 in) of precipitation from thunderstorms over a span of 4 to 6 weeks. This occurred during the unusually wet July and August of 2003. Many wildfires can be ignited during this weather pattern. However, due to the accompanying precipitation they do not spread much, and therefore do not become a significant problem if standard levels of local firefighting resources are available.

Strong Subsidence/Low Relative Humidity A fourth pattern, strong subsidence/low relative humidity (RH) can, with enough duration, cause a significant increase in Northeastern Plateau fire potentials, even without much wind. The pattern occurs when a strong mid- and/or upper-level high-pressure area is centered to the west of northeastern California for a period of at least several days. High pressure over the area leads to diverging air at the surface. The surface divergence induces large-scale slow sinking of the mid/upper air mass over northeastern California. This phenomenon is called *subsidence* and it leads to adiabatic compression that warms and dries the lower atmosphere. Perhaps its most important effect is the hindrance of normal nighttime RH recovery. Non-subsidence pattern recovery occurs on lower slopes and in drainage bottoms. The RH recovery ranges from 35% to 45%, and may reach 55%. In a subsidence situation, which is most common in August or September and only pertains to mid-slopes and above, the daytime minimum RH usually drops to 4% to 12%, but nighttime recovery is very low, reaching only the 15% to 30% range. Dead fuel moistures drop; live fuels become more stressed; and fires ignite, spread, and spot more easily. Fire seasons characterized by large fires and/or extreme fire behavior are often preceded by one to several years of drought.

LIGHTNING

Lightning strikes are most common from June through August and peak in July; though they may occur from May through September (van Wagtendonk and Cayan 2007). Ignition probability is highest in late July through September as fuel moistures are typically at their lowest. Total number of lighting strikes from 1985 to 2000 was 73,187 covering 2,034,103 km^2 with 22.49 yr^{-1} 100 km^{-2}. The top three lightning-impacted areas by total number of strikes (in parentheses) and number $year^{-1}$ 100 km^{-2} within this bioregion are: (1) Devil's Garden (8,475) 19.76; (2) Fredonyer Peak (6,876) 24.91; (3) Tule Mt. to McDonald Peak (6,697) 25.06; and (4) Shaffer and Skedaddle Mountains to Eagleville (6,395) 21.08. Lightning occurrence is not linearly related to an increase in elevation in this bioregion. The distance lightning can travel in this area ranges from 180 km (112 mi), which is the longest at the lower end range of all bioregions, to 340 km (211 mi), which is second to the Southeast Desert bioregion.

Ecological Zones

The quantity of available moisture from precipitation or soil is the primary determinant of vegetation and fire regimes in the Northeastern Plateaus bioregion. Major zones include: sagebrush steppe, lower montane, mid-montane, upper montane, and subalpine zones.

The most widespread zone is the sagebrush steppe, which occurs across the driest areas in the bioregion, typically on the extensive flatter, lower-elevation areas. This zone is predominately shrub steppe, shrub desert, juniper woodlands and pinyon-Sierra juniper woodlands in the northern Mono section (Vasek and Thorne 1988, Young et al. 1988). The sagebrush steppe extends from California across the Intermountain Region into Wyoming, occupying a total of 44.4 million ha (110 million ac) (West 1983). It is generally characterized by an overstory dominated by short to tall forms of sagebrush with an understory of forbs and grasses.

Three coniferous forest-dominated zones occur with increasing elevation and precipitation associated with mountains or topographic relief. The lower montane zone is dominated by ponderosa pine *(Pinus ponderosa)* and mixed ponderosa pine/Jeffrey pine *(Pinus jeffreyi)* woodlands and forests. The mid-montane zone contains mixed forests of ponderosa pine, Jeffrey pine, and various amounts of white fir *(Abies concolor)*. The upper-montane zone is defined as the area where the climate supports the higher-elevation white fir forests in pure stands and in mixes with western white *(Pinus monticola)*, ponderosa, Jeffrey, and Washoe *(Pinus washoensis)* pines. The subalpine zone occurs in limited locations in the highest elevations of the Warner Mountains, dominated by whitebark pine *(Pinus albicaulis)* woodlands with some white fir, lodgepole pine *(Pinus contorta* ssp. *murryana)*, and western white pine.

Overview of Historic Fire Occurrence

Prehistoric Period

The Northeastern Plateaus bioregion appears to have been a dry region for more than a million years since the building of the Cascade Range began to block moisture from Pacific winter storms. Over this period the assemblages of vegetation across the bioregion appear to have remained relatively stable, though oscillating from grasslands to sagebrush and juniper to pine woodlands and forests depending on the trends in effective moisture as the climate changed between glacial and interglacial periods (Adam et al. 1989). During the Wisconsin glacial interval (the last major glaciation), several periods of extreme cold created conditions where Grass Lake, at only 1,537 m (5,043 ft) in the Cascade Range to the north of Mt. Shasta, appears to have been above treeline. The last of these cold periods appears to have occurred between 18.9 and 17.6 ka (thousands of years before present) (Hakala and Adam 2004). This coincides with glacial advances in several areas of the south Warner Mountains where the more extensive evidence of glacial activity is from the Pine Creek and

Owl Creek drainages. What is now Paterson Lake was then covered with glacial ice (Osborn and Bevis 2001).

We are aware of only two paleoecological studies looking at the association of climate, fire, and vegetation in this bioregion: one by Wigand et al. (1995) in the Eagle Lake area on the southwestern edge of the bioregion, and one by Minckley (2003) at Patterson and Lily Lakes in the Warner Mountains.

In the southeastern portion of the bioregion around Eagle Lake, it appears that most of the early and mid-Holocene (11.5–4 ka) was characterized by warm and very dry conditions. Vegetation appears to have oscillated between grasslands and shrublands during this period. Fire frequency appears to have been variable at this time with little charcoal produced. Juniper woodlands had their greatest expansion during the cool, moist period of 4 to 2 ka. This was accompanied by changes in the fire regime that indicated the greater available biomass would promote occasional extensive fires that would be followed by short-term expansion of grasslands that would then give way again to juniper woodlands. Eventually, a general increase in effective moisture over the last millennium appears to have increased vegetative productivity and promoted more frequent fires. As a result, the juniper woodlands gave way to the eastside yellow pine–dominated forests and woodlands that are characteristic of the area today (Wigand et al. 1995).

The Warner Mountains show warming and drying trends from 11.5 to 9 ka with sagebrush expansion similar to the rest of the Great Basin. However, fir expansion begins between 9 and 8 ka indicating greater effective moisture in contrast to other Great Basin locales. This also coincides with increased fire activity, as evidenced by charcoal influx to lakes in the Warner Mountains. Fir and pine continue to increase and modern forest conditions appear to assemble between 4.5 and 3 ka. Two other periods of high fire activity took place between 4 and 3 ka and from 2 to 1 ka. Interestingly, the mixed conifer vegetation at Lily Lake in the north Warners has been unusually stable in spite of the oscillations in fire activity. Here, variation in fire activity appears to have been independent of trends in vegetation. However, variation in fire activity, as indicated in charcoal influx, has been similar to variation in fire activity across the region (Minckley 2003). Recent work suggests that background levels of charcoal from lake sediments are an indicator of extralocal or regional fire activity (Whitlock et al. 2004). It may be that Lily Lake is capturing a more regional picture of fire activity and that local fires were of low intensity or patchy enough for locally produced charcoal to not overwhelm the regional picture (Minckley 2003).

Historic Period

In 1880, there were reported 16,000 beef cattle, 23,000 resident sheep, and 6,000 horses in Modoc County. By 1909, the reported numbers had increased to 44,000 beef cattle, 76,500 resident sheep, and 15,000 horses (Pease 1965, cited in Laudenslayer et al. 1989). In addition, transient sheep from neighboring areas of California, Oregon, and Nevada came through parts of the bioregion in the spring, summer, and fall. The intensive, largely unmanaged livestock grazing affected fire regimes by reducing fine fuels, reducing some plant life forms such as the perennial native bunchgrasses and palatable shrubs (e.g., antelope bitterbrush *[Purshia tridentata]*), and facilitating other life forms such as annual grasses and unpalatable trees and shrubs (e.g., sagebrushes and western juniper *[Juniperus occidentalis* var. *occidentalis]*) (Laudenslayer et al. 1989).

TABLE 11.1
Number and area of fires recorded on the Modoc National Forest, 1910–2005

	Total Number of Fires	*Fires ≥2000 Acres*	*Total Acres Burned*
1910–1919	64	2	212,652
1920–1929	65	6	57,847
1930–1939	73	7	74,847
1940–1949	78	7	91,733
1950–1959	49	12	162,487
1960–1969	34	2	13,087
1970–1979	65	10	282,736
1980–1989	47	6	37,058
1990–1999	123	10	109,381
2005	39	4	56,431

NOTE: These values include human-caused and lightning ignitions (data on file, Modoc N.F., Alturas, CA).

Ponderosa and Jeffrey pine stand structures were generally more open at the time of Euro-American settlement. Many early accounts of settlers, soldiers, surveyors, and others mention stands composed of large, widely scattered trees (Laudenslayer et al. 1988). Fire was almost certainly one of the main factors that affected the forest structures observed in those early accounts. Historic period disturbances of logging, grazing, mining, agriculture, rail networks, highways, and so forth began in the region in the 1860s, and were in full swing by the 1920s. This area, however, was always more sparsely populated than the main California Gold Rush regions to the south and west.

Timber harvesting was always an important activity in the forested areas of this bioregion. The main species harvested were ponderosa and Jeffrey pine. Most of the early timber harvest was used locally. By the mid 1930s, the local timber industry became economically dependent on sales of timber outside the eastside pine region, much of which is included in the northeastern California bioregion. Fire suppression was actively pursued and promoted as a means of protecting the economic resource provided by the region's forests, and was probably effective in the flatter areas of the bioregion (Laudenslayer et al. 1989).

The number of fires and area burned from 1910 to 1999 in the Modoc National Forest is depicted in Table 11.1. Though only two fires were recorded to have burned greater than 800 ha (2,000 ac) from 1910 to 1919, the area burned was the

second largest in the 90-year record. Many small fires were started by trains, escaped warming fires, and fires set to clear land for agriculture. As railroad and highway traffic increased, so did the number of ignitions along travel corridors in this bioregion. The highest number of hectares burned between 1970 and 1979 were after several above-average years of precipitation in the early 1970s followed by drought in the latter part of the decade. During the time period between 1990 and 1999, 123 fires were recorded, the highest recorded for this area of the Northeastern Plateaus bioregion.

The Sugar Hill fire, probably the most famous large and destructive fire in early Modoc County history, was ignited by sparks from a train on the Southern Pacific Railroad on the west side of the North Warner Mountains on July 22, 1929 (Brown and Brown 1991). The fire started around one in the afternoon when relative humidity was zero with high temperatures and near-gale winds. The fire quickly spread to the east, and at the peak of the fire storm, "the fire swept a half mile across Lassen Creek Canyon in an immense sheet of flame from tree tops on one side to those on the other, without igniting the ranch buildings and haystacks in the canyon bottom" (Brown and Brown 1991). It was finally controlled by 600 men after burning 2,400 ha (6,000 ac) of forest in two days. Though Crane Creek sawmill was heroically saved, many of the mill workers' homes were destroyed in the blaze. Thought to be totally extinguished, the fire erupted from smoldering roots and burned an additional 200 ha (500 ac) on August 5 before being controlled by 300 firefighters. There was no loss of life, but the loss of timber that was to be cut and milled was significant.

NATIVE AMERICANS

Information on the use of fire by Native Americans in this bioregion, either prehistoric or during initial contact with settlers is rare. Deer firing *(kupi't)* was an effective tool used by the Northern Paiute to move and corral frightened mule deer *(Odocoileus hemionus)* into a waiting party of hunters (see full quotation at the beginning of this chapter, Kelly 1932). Pease (1965) questioned the local legend that Native Americans used fire to keep the forest understory clear of shrubs to increase game and facilitate hunting. Fire use by Native Americans, Pease (1965) argued, would have reduced understory forest vegetation from repeated burning. In the lower-montane sites, Native American fire use coupled with lightning ignitions would have favored herbaceous species over fire-sensitive shrubs such as antelope bitterbrush, which is a key browse plant of mule deer and a major food source of the Native Americans (see Sidebar 11.1). However in the mid-montane zone, repeated fire promotes fire-adapted shrub species (e.g., ceanothus and manzanita), which make wildlife and human travel difficult and are not preferred browse species of mule deer.

Current Period

Since the turn of the century, fire regimes have changed among many of the semi-arid plant associations in northeastern California. These changes are largely attributed to the reduction of fine fuels through livestock grazing, fire suppression, the introduction of highly flammable, exotic annual herbs, and climate change. Since the late 1870s, some plant associations within ecological zones burn more frequently and others less frequently than prior to this period. Where fires were historically more frequent and have currently become less frequent, fire severity has increased.

Major Ecological Zones

Sagebrush Steppe

The predominant sagebrush species are (1) big sagebrush *(Artemisia tridentata)*, which include subspecies Wyoming *(Artemisia tridentata* ssp. *wyomingensis)*, basin big *(Artemisia tridentata* ssp. *tridentata)*, and mountain *(Artemisia tridentata* ssp.*vaseyana)*; and (2) low *(Artemisia arbuscula)*; and to a lesser extent black sagebrush *(Artemisia nova)* (Rosentreter 2005) (Fig. 11.1). Series associated with the sagebrush steppe are western juniper woodlands, quaking aspen *(Populus tremuloides)*, riparian, and associations dominated by antelope bitterbrush and curl-leaf mountain-mahogany *(Cercocarpus ledifolius)*. Presettlement juniper communities were primarily shrub savannas, which usually occupy shallow rocky soils. Most present day juniper woodlands in northeastern California have become established since the late 1800s and have replaced sagebrush steppe communities where precipitation is sufficient to support western juniper. Generally this occurs in areas that receive a minimum of 30 cm (12 in) of precipitation. In addition, introduced plant species such as cheat grass *(Bromus tectorum)*, medusahead *(Taeniatherum caput-medusae)*, tumble mustard *(Sisymbrium altissimum)*, and the native western tansy mustard *(Descurainia pinnata)* have successfully invaded and replaced shrub steppe communities throughout this region resulting in a shift to annual grasslands.

The pluvial valley bottoms, primarily found in the Surprise Valley and Honey Lake areas, are occupied by the greasewood and shadscale series. Greasewood *(Sarcobatus vermiculatus)* is also found in the lowlands around Alturas and in the Klamath Basin around Tule Lake and Klamath Marsh (Young et al. 1988). These environments have soils that are too saline or precipitation that is too low for most species of sagebrush to be a dominant component. Predominant shrub species are fourwing saltbush *(Atriplex canescens)*, greasewood, hop-sage *(Grayia spinosa)*, and winterfat *(Kraschenninnikovia lanata)*. Tables 11.2 and 11.3 describe the fire regimes of important species of the shrub steppe.

FIRE RESPONSES OF IMPORTANT SPECIES

One of the largest changes in community composition following fire is the immediate reduction of the shrub layer. Shrubs, associated with sagebrush steppe plant communities across the bioregion are composed of fire-tolerant and fire-intolerant species (Tables 11.4 and 11.5). The composition of

SIDEBAR 11.1. ANTELOPE BITTERBRUSH

Antelope bitterbrush *(Purshia tridentata)* is the dominant understory species in pine forests throughout the Northeastern Plateaus bioregion. How it responds to fire is of concern to land managers because of its high value as a wildlife browse species (Guenther et al. 1993). Antelope bitterbrush provides 73% to 84% of mule deer's diet from June to July in the pumice zone of south-central Oregon (Gay 1998) and is a major portion of their winter diet where both species occur (Salwasser 1979, Gay 1998). It is also important as small-mammal food source (Vander Wall 1994) and habitat (Smith and Maguire 2004), as well as neotropical bird nesting habitat (Dobkin and Sauder 2004), and contributor to soil productivity via symbiotic nitrogen fixation (Busse 2000a, 2000b).

Antelope bitterbrush grows from the sagebrush steppe to the mid-montane zone in the 30 cm to 89 cm (12 in to 35 in) precipitation range, respectively. It is adapted to forested environments where fire burned with low intensity, (function of energy content of the fuel, the mass of the fuel consumed, and the rate of the fire spread [Agee 1993]), low severity (effect of fire on plants [Agee 1993]), and frequent fire-return intervals (4 to 24 years) (Arno 1976, Wright and Bailey 1982, McArthur et al. 1983, Bork 1985, Agee 1993, see also West 1968, Sherman and Chilcote 1972, Martin 1983, Vander Wall 1994). Historically very frequent fires (<10 years) probably limited the ability of bitterbrush to occupy a site and favored fire resilient herbaceous species, especially graminoids (Riegel unpublished data). Because of lower precipitation in the sagebrush steppe, plant distribution is more dispersed with greater distances between shrubs and herbaceous species and seed production is limited (Young and Clements 2002. Riegel unpublished data). The heterogeneity of vegetation structure and biomass production prior to fire suppression and livestock grazing allowed for some plants to survive a mixed fire regime with low to moderate fire intensities and longer fire-return intervals (>30 years). This allowed for the potential to develop 50- to 98-year-old plants (Adams 1975, Clements and Young 2001, Riegel unpublished data). The largest antelope bitterbrush specimen in California reported to date was from Janesville, at the edge of Sierra Nevada and Northeastern Plateaus bioregions Nord (1965). It was 128 years old, 3.7 m (12 ft) tall, and 6.1m (20 ft) across.

Livestock grazing from 1880 through the early 1920s promoted an increase in antelope bitterbrush cover by: (1) selectively reducing competition between the historic graminoid-dominated understory, allowing antelope bitterbrush to expand and occupy a greater percentage of the understory, and (2) reducing the major source of fine fuels—graminoids—needed for carrying fire through the understory to the detriment of bitterbrush (Weaver 1943, Adams 1975, Peek et al. 1978, Shinn 1980). These factors, coupled with above-average precipitation during this time period (Antevs 1938, Graumlich 1987, Miller et al. 1994), allowed antelope bitterbrush and ponderosa pine to become established in higher densities than during the presettlement fire regime (Agee 1933).

The ability of antelope bitterbrush to persist in fire-prone environments is a function of sprouting (Blaisdell 1950, Blaisdell and Muggler 1956, Martin and Johnson 1979, Clark et al. 1982, Martin 1982, 1983, Martin and Driver 1983, Hurley and Ratcliff 1986, Cook et al. 1994) and rodent seed caching (West 1968, Sherman and Chilcote, Martin 1983, Vander Wall 1994). The literature on antelope bitterbrush's response to fire is often confusing. Fire is commonly reported to be very destructive to antelope bitterbrush (Hormay 1943, Billings 1952, Wagstaff 1980). Some plants have been observed to sprout after burning under favorable conditions (Nord 1959, 1965), while some populations have exhibited abilities to sprout reasonably well following burning (Blaisdell 1950, Blaisdell and Muggler 1956, Martin and Johnson 1979, Clark et al. 1982, Hurley and Ratcliff 1986, Cook et al. 1994). Antelope bitterbrush growing in pumice soils of central Oregon and northeastern California has been reported to sprout infrequently following fire (Driscoll 1963, Nord 1965, Sherman and Chilcote 1972, Martin 1982, 1983, Martin and Driver 1983).

However, Busse et al. (2000) found 25% of a bitterbrush population that sprouted was still alive by year five. There are a variety of factors and their interactions that may contribute to antelope bitterbrush's ability to sprout: genetics, plant age, phenology, soil moisture, soil texture, fuel moisture, fuel loads, fire intensity, and fire severity. Given the genetic potential to produce sprouts from the basal crown (Alderfer 1977, Winward and Alderfer-Finley 1983, Agee 1993, Jabbes 2000), plants older than 5 years and younger than 20 to 40 years typically produce the most sprouts (Martin and Driver 1983, Simon 1990). Early-season spring burns have the highest potential for successful sprouting when: (1) soil moisture is high enough to reduce heat flux to the basal crown and roots and allow for sufficient regrowth, (2) moisture content of small fuel (woody material <0.6cm [1/4"in] in diameter, generally drying out in 1 hour, with a comparatively high surface area-to-volume ratio) is low enough for rapid fire (low residence time), (3) and moisture content of heavy fuels (large diameter wood such as large limbs, logs, and snags) is relatively high so that these fuels do not burn. These abiotic factors interact with plant phenology; burning early versus late in the growing season may yield higher sprouting success as the plants have a greater amount of time to resprout and resume photosynthesis if the basal crown was not killed and the plant has sufficient carbohydrate reserves to facilitate resprouting (Agee 1993). The percentage of successful sprouting appears to be inversely related to fire severity as measured by Blaisdell (1950, 1953), Nord (1959), and Driscoll (1963). Thus, dry fuels produce the highest heat yields, but high soil moisture reduces the heat flux to the crown and root tissues, mitigating the effect of intensity.

Establishment and recruitment of antelope bitterbrush occurs primarily by seed from rodent caching (West 1968, Sherman and Chilcote 1972, Martin 1983, Vander Wall 1994). Successful caches are made in the interspaces between shrubs where litter is sparse. Though a sprouting plant may produce flowers and seed the first year following fire, it takes typically 9 to 15 years to reach pre-burn production (Pechanec et al. 1954, Wright 1972, in Noste and Bushey 1987). Fire-return intervals shorter than 20 years yield the highest sprouting potential of antelope bitterbrush (Simon 1990) but result in lower density and cover, while fire-return intervals greater than 20 to 40 years allow for a sufficient population of seed bearing shrubs but only a small percentage may retain the ability to sprout (Simon 1990, Gay 1998).

Mule deer populations have been declining in this bioregion for several decades (Salwasser 1979, Clements and Young 1997) There are many causal factors that contribute to mule deer habitat loss: (1) increased frequencies of wildfires in sagebrush steppe that comprise winter ranges, (2) suppression of fire in the lower- and mid-montane zone that are key transition and summer ranges have facilitated western juniper expansion (Clements and Young 1997), and increased Jeffrey and ponderosa pine stand density and canopy development limiting light levels for regeneration (Peek et al. 2001), and (3) in the lower elevation sites, the introduction of highly competitive exotic annual grasses that reduce antelope bitterbrush regeneration success and also facilitate frequent fires that are extremely flammable when dry (Clements and Young 1997). In south central Oregon mule deer preferred canopy closures ≤40% in their core home ranges (Gay 1998). Thinning forest stand density to <40% canopy cover will provide antelope bitterbrush with the higher light levels that are optimal for establishment and growth (Gillespie and Loik 2004). The paradox for land managers is how to maintain acceptable fuel loads, which includes pine needle drape on antelope bitterbrush (*pine needle drape* acts as accelerant that increases flame heights 2 to 5 fold), while growing bitterbrush for habitat and dietary needs of mule deer. See Young and Clements (2002) for a thorough and practical approach to understanding the biology and management of antelope bitterbrush.

FIGURE 11.1. Sagebrush steppe zone. This mountain big sagebrush site had recently burned. Rabbitbrush has resprouted and bitterbrush appears to be growing in the background. (Photo by Rick Miller.)

TABLE 11.2
Fire regime characteristics for sagebrush types in shrub steppe ecological zones in northeastern California

Vegetation type	Low sage	Basin big sage	Wyoming sage	Mountain big sage
Temporal				
Seasonality	Summer–early fall	Summer–early fall	Summer–early fall	Summer–early fall
Fire-return interval	Truncated medium	Truncated medium	Truncated medium	Truncated medium
Spatial				
Size	Medium	Small	Large	Large
Complexity	High	High	Multiple	Multiple
Magnitude				
Intensity	Moderate	Moderate	Moderate	Moderate
Severity	Very high	Very high	Very high	High
Fire type	Surface	Multiple	Multiple	Multiple

fire-tolerant and fire-intolerant shrubs prior to the burn influences shrub composition during the early years following fire.

Sagebrush Species of sagebrush found in the northeast bioregion are easily killed by fire with the exception of silver sagebrush *(Artemisia cana)* and snowfield sagebrush *(Artemisia speciformis)*, both of which resprout (Winward 1985) (Table 11.4). Reestablishment of the big sagebrush subspecies and the two forms of low sagebrush *(Artemisia arbuscula)* is entirely dependent on unburned seed. The potential for large inputs of sagebrush seed following a fire is limited and dependent on the amount of unburned edge and the amount and distribution of unburned sagebrush shrubs in the burn. Sagebrush seed is mainly distributed by wind, with no evidence of seed caching by animals. Seed movement from adjacent unburned areas has been reported to be slow, taking many years to move into the interior of the burn (Mueggler 1956, Johnson and Payne 1968, Ziegenhagen 2003, EOARC data*). The majority of sagebrush seed (more than 90%) is disseminated within

*EOARC is the Eastern Oregon Agricultural Research Center, jointly operated by Oregon State University and USDA Agricultural Research Service, Burns, OR.

TABLE 11.3

Fire regime characteristics for non-sagebrush types in shrub steppe ecological zones in the Northeastern Plateaus bioregion

Vegetation type	Bitterbrush	Mountain mahogany	Juniper	Quaking aspen	Cheat grass
Temporal					
Seasonality	Summer–early fall	Summer–early fall	Late summer–early fall	Late summer	Summer–early fall
Fire-return interval	Truncated medium	Truncated medium	Truncated long	Medium	Short
Spatial					
Size	Medium	Small	Small–large	Small	Large
Complexity	Multiple	Multiple	Multiple	Low	Low
Magnitude					
Intensity	Moderate	High	High	Multiple	Low
Severity	Multiple	Very high	Very high	High	Low
Fire type	Multiple	Multiple	Multiple	Surface	Surface

TABLE 11.4

Relative response of common shrubs in the sagebrush biome and salt deserts to fire

Tolerant	*Moderately Tolerant*	*Intolerant*
Yellow rabbitbrush (s)	Rubber rabbitbrush (s)	Low sagebrush
Wax currant (s)		Big sagebrush
Desert gooseberry (s)		Fourwing saltbrush
Wood rose (s)		Curl-leaf mountain-mahogany
Greasewood (s)		Hop-sage
Mountain snowberry (s)		Antelope bitterbrush (ws)
Horsebrush (s)		

NOTE: s = sprouter, ws = weak sprouter. Derived from Blaisdell 1953, Wright et al. 1979.

9 m (30 ft) of the parent plant and nearly 100% within 30 m (98 ft) (Harniss and McDonough 1974, Johnson and Payne 1968). This suggests the rate of recovery is highly dependent on soil seed pools immediately following the fire, particularly on large fires with few unburned islands and unburned plants in the burn perimeter.

Sagebrush seed matures and disperses in the autumn and winter (Young and Evans 1989, Meyer and Monsen 1991). The majority of wild and prescribed fires occur during the summer and early fall, prior to maturity of the current year's seed crop. So soil seed pool reserves are determined by the previous year(s) seed crop(s).

Although seed crop production is highly variable from year to year, seed production is usually not a limiting factor for big sagebrush establishment (Harniss and McDonough 1974). Goodwin (1956) reported that sagebrush produced 350,000 seeds m^{-2} of canopy annually. In the first year after fire, sagebrush seedling density can be high (Blaisdell 1953, Mueggler 1956). However, with little seed input, seed pool reserves will decline through germination, loss of viability, and predation within the first two years following fire. Hassan and West (1986) reported Wyoming big sagebrush seed densities following a summer fire were 0.7 viable seeds m^{-2} compared to 3.7 viable seeds m^{-2} on an adjacent unburned site. In the second fall after the fire, viable seed densities declined to 0.3 m^{-2}.

In northwestern Nevada, a 4,000-ha (9,900-ac) wildfire occurred in 1994 prior to seed maturation and left few surviving sagebrush plants. In 1993, an extremely wet year, large seed crops were produced. During the first two growing seasons following the fire, rapid establishment of mountain big sagebrush and antelope bitterbrush occurred (Ziegenhagen 2003) (Fig. 11.2). During this period, 0.3 and 0.9 shrubs m^{-2} were successfully established. In mountain big sagebrush

TABLE 11.5
Relative response of common perennial forbs in the sagebrush biome to fire

None to Slight	*Moderate to Severe*
Yarrow *(Achillea millefolium)*	Pussy-toes *(Antennaria* spp.*)*
Agoseris *(Agoseris* spp.*)*	Sandwort *(Arenaria* spp.*)*
Onion *(Allium* spp.*)*	Matted buckwheat *(Eriogonum caespitosum)*
Aster *(Aster* spp.*)*	Parsnipflower buckwheat *(Eriogonum heracleoides)*
Milkvetch *(Astragalus* spp.*)*	Douglas' buckwheat *(Eriogonum douglasii)*
Woollypod milkvetch *(Astragalus purshii)*	Slender buckwheat *(Eriogonum microthecum)*
Balsam-root *(Balsamorhiza* spp.*)*	Sulfur flower *(Eriogonum umbellatum)*
Indian paintbrush *(Castilleja* spp.*)*	Spiny phlox *(Phlox hoodii)*
Hawksbeard *(Crepis* spp.*)*	
Fleabane daisy *(Erigeron* spp.*)*	
Geranium *(Geranium* spp.*)*	
Avens *(Geum macrophyllum)*	
Prickly lettuce *(Lactuca serriola)*	
Lomatium *(Lomatium* spp.*)*	
Lupine *(Lupinus* spp.*)*	
Bluebells *(Mertensia* spp.*)*	
Slender phlox *(Phlox gracilis)*	
Penstemon (*Penstemon* spp.*)*	
Longleaf phlox *(Phlox stansburyi)*	
Cinquefoil *(Potentilla* spp.*)*	
Lambstongue ragwort *(Senecio integerrimus)*	
Goldenrod *(Solidago* spp.*)*	
Dandelion *(Taraxacum* spp.*)*	
Goat's beard *(Tragopogon* spp.*)*	
Largehead clover *(Trifolium macrocephalum)*	
Foothill death camas *(Zigadenus paniculatus)*	

NOTE: Derived from Blaisdell 1953, Pechanec et al. 1954, Lyon and Stickney 1976, Klebenow and Beall 1977, Wright et al. 1979, Volland and Dell 1981, Bradley et al. 1992.

associations, shrub densities in fully established stands ranged between 0.8 and 1.2 shrubs m^{-2} (Ziegenhagen 2003, EOARC data). The decline in seedling establishment in 1997 and 1998, both wetter-than-average years, was attributed to depleted seed pools due to germination of seed, loss of seed viability, granivory, and limited input from newly established plants. In several other fires, seedling establishment began to increase 8 to 10 years after the fire as earlier established plants matured and began producing adequate seed.

Rate of big sagebrush recovery following fire is highly variable but tends to be slower in the more arid plant associations (Bunting 1984a). Within 15 to 25 years following the fire, mountain big sagebrush canopy cover can be 15% to 25% (Bunting et al. 1987, Ziegenhagen 2003). Rate of recovery is highly dependent on soil seed pools and moisture availability in growing seasons immediately following the fire.

Antelope Bitterbrush Antelope bitterbrush is a weak sprouter in the Northeastern Plateaus bioregion, and its response to fire is highly variable (Tables 11.4 and 11.5). Blaisdell and Mueggler (1956) reported survival from resprouting in eastern Idaho was 49% in a light burn, 43% in a moderate burn, and 19% in a hot burn. They also reported greater survival of resprouting in plants less than 15 years old. Driscoll (1963) concluded survival of antelope bitterbrush resprouts was related more to soil surface texture than fire intensity in central Oregon. In California, 5% to 25% of the antelope bitterbrush successfully resprouted in 5 of 13 fires (Nord 1965). Survival from resprouting was also greater in spring burns than fall burns (Clark et al. 1982).

In northwestern Nevada, less than 1% of the antelope bitterbrush survived by resprouting (Ziegenhagen 2003). The majority of shrub establishment occurred from soil seed reserves. However, unlike sagebrush, an important vector of antelope bitterbrush seed dispersal is small mammal seed caches. When antelope bitterbrush successfully resprouts following fire, the species can recover to nearly fully stocked stands within 9 to 10 years after fire (Wright et al. 1979). However, on sagebrush-associated sites evaluated

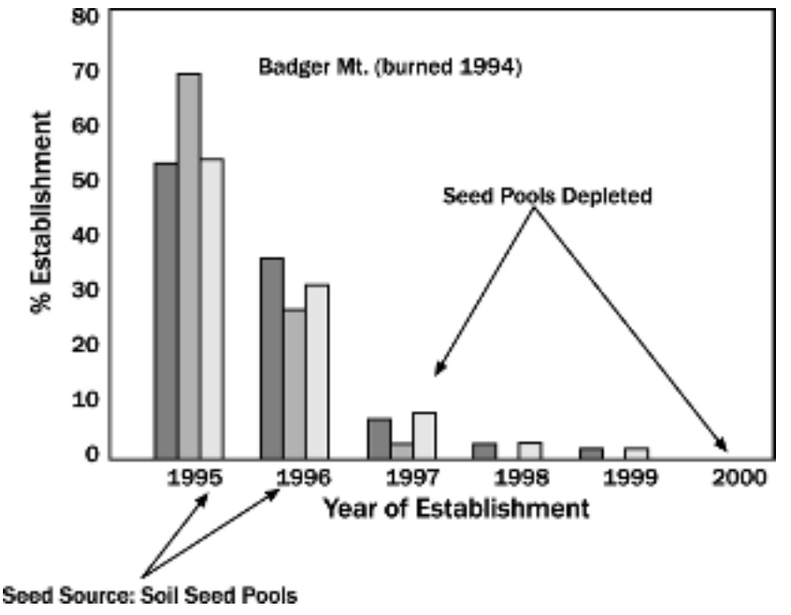

FIGURE 11.2. Mountain big sagebrush seedlings became established from soil seed pools only in the first two years post-fire. The seed pool decline was precipitous in the first year and by the third year the pools were exhausted, emphasizing the short-lived seeds of sagebrush. Beyond the second year, recruitment largely occurs from new seed that is wind dispersed on the site (Ziegenhagen 2003).

several years after fire in northern California, southeastern Oregon, and northwestern Nevada, reestablishment was primarily from seed and occurred at the same rate as sagebrush on the same sites. On a heavily browsed deer winter range in south central Oregon, antelope bitterbrush did not recover to preburn levels 40 years after the fire (EOARC data). Antelope bitterbrush seedlings also compete poorly with cheat grass, which can severely limit reestablishment (Holmgren 1956).

Rabbitbrush and Horsebrush Rubber and yellow rabbitbrush and horsebrush are capable of sprouting and more rapidly recovering immediately following fire than big sagebrush (Tables 11.4 and 11.5). However, rubber rabbitbrush is more sensitive to fire than yellow rabbitbrush (Wright et al. 1979). In some areas, establishment is from both seeds and shoots (Young and Evans 1978). In eastern Idaho and northeastern California, establishment was primarily from crown and root shoots with little from seed (Bunting et al. 1987, EOARC data). Although percent composition is usually higher for these sprouting species during the early years following fire, percent cover of these sprouters often does not exceed preburn levels on good condition sites (Bunting et al. 1987, EOARC data). The abundance of these sprouting species also usually declines over time as sagebrush abundance increases and the intervals between disturbance increase (Young and Evans 1978, Whisenant 1990). However, density and cover of these species can exceed preburn levels, especially on degraded sites (Chadwick and Dalke 1965, Young and Evans 1978). Abundance of horsebrush (*Tetradymia* spp.) remained higher than preburn levels 30 years after the fire (Harniss and Murray 1973). Heavy grazing following fire can also increase the abundance of rabbitbrush and horsebrush as these taxa are not palatable to livestock.

Curl-Leaf Mountain-Mahogany Curl-leaf mountain-mahogany, a weak sprouter, is highly susceptible to fire (Wright et al. 1979) (Tables 11.4 and 11.5). Presettlement-aged plants are usually found on rocky ridges, which are fire protected (Dealy 1975, Gruell et al. 1984, Davis and Brotherson 1991). Postsettlement stands are commonly found in communities where presettlement fire return intervals were less than 20 years, but have significantly increased since the late 1800s (Dealy 1975, Gruell et al. 1984, Miller and Rose 1999). Age analysis indicates a large increase of curl-leaf mountain-mahogany since the late 1800s.

Reestablishment following fire is largely dependent on seedling establishment (Wright et al. 1979); thus a nearby seed source is important. Curl-leaf mountain-mahogany has regenerated in several large burns in northeastern California (Tule Mountain 1959 and Three Peaks 1957) where mature plants, often located on rocky micro sites, survived the fire. However, little recruitment has occurred in a large curl-leaf mountain-mahogany stand following the 1973 Ninemile fire where few plants survived as a result of high fire severity.

Herbs and Grasses Ground cover of native herbaceous vegetation can quickly recover and exceed preburn levels in some plant associations and decrease or equal preburn levels in others (Blaisdell 1953). Rate of recovery and composition following a fire is largely determined by moisture regime, ecological site plant composition prior to the burn, soil seed reserves, fire tolerance of species on the site, fire intensity, weather conditions, and management following the fire. In general, ground cover of native grasses and forbs growing in

the more mesic sagebrush communities—characterized by mountain big sagebrush, Columbia needlegrass *(Achnatherum lemmonii)*, Idaho fescue *(Festuca idahoensis)*, and bluebunch wheatgrass *(Pseudoroegnaria spicata)*—recovers rapidly and often exceeds preburn levels within 1 to 3 years after fire. However, in more arid plant associations and plant associations with fire-sensitive grasses and forbs, recovery is slower and cover will often not exceed preburn levels for many years. If density of native species is less than 2 plants m^{-2} (Bates et al. 2000), and cheat grass is abundant in the understory, burning will likely convert the site to introduced annual grassland.

In general, broad-leaf grasses such as squirreltail *(Elymus elymoides)*, bluebunch wheatgrass, and Columbia needlegrass are relatively resistant to fire, recovering quickly and often producing greater amounts of biomass following a fire (Blaisdell 1953, Wright 1971, Bunting et al. 1987, EOARC data). In contrast, fine-leaf grasses such as Idaho fescue and Thurber needlegrass *(Achnatherum thurberianum)* are more sensitive to fire, suffering greater crown mortality and slower recovery rates than broad-leaf grasses (Blaisdell 1953, Wright 1971). Fine-leaf grasses usually accumulate more dead material in the crown, which causes the plant to burn more slowly, transferring more heat to the growing points (Wright 1971).

Conrad and Poulton (1966) reported grazed Idaho fescue and bluebunch wheatgrass plants were less damaged than ungrazed plants. In a Wyoming big sagebrush plant association in south central Oregon, bluebunch wheatgrass was severely damaged in an ungrazed exclosure where considerable dead leaf material had accumulated in the crowns (EOARC data). Outside of the exclosure on the grazed portion of the study area, bluebunch wheatgrass biomass and ground cover was near preburn levels in the first year following fire.

Although the majority of literature reports Idaho fescue is fire sensitive and declines in the first year following fire (Blaisdell 1953, Conrad and Poulton 1966, Countryman and Cornelius 1957), this species usually recovers, and biomass and cover can exceed preburn levels within 3 to 5 years after the fire (EOARC data). In Lassen County, Idaho fescue crown area decreased by one third in the first year following fire but was 35% greater than preburn levels in the third growing season.

Forb species, which resprout below ground from a caudex, corm, bulb, rhizome, or rootstock, usually exhibit rapid recovery following fire. The majority of these forbs are dormant at the time of the fire and their growing points are protected from heat. However, forbs that are *suffrutescent* or mat forming, such as sandwort *(Arenaria)* and wild buckwheat *(Eriogonum)* species, have their growing points above ground and can be severely damaged by fire, resulting in crown area reduction or mortality. Perennial forb production usually increases two- to threefold following fire in the more mesic sagebrush plant associations (Blaisdell 1953, EOARC data). However, perennial forb response is usually less in the drier plant associations (Blaisdell 1953, Bunting et al. 1987, Fischer et al. 1996, EOARC data).

In relatively good condition sites, the largest increases in vegetation during the first several years following fire are often the native annuals, if sufficient moisture is available. Most species have completed their life cycle by early summer, prior to most fire events. During the first growing season following fire, annuals are able to take advantage of increased nutrient availability and decreased competition from perennials. In several fires in northeastern California and northwestern Nevada, native annuals increase three- to fivefold in the first and second year following fire (EOARC data). However, annual response typically lasts only two to five growing seasons following a fire event. Their response can also be greatly limited by dry conditions in the spring. In heavily disturbed or warmer sites (e.g., Wyoming big sagebrush series), the native annual response is replaced by exotic annuals and biennials, which dominate the site.

FIRE REGIME–PLANT COMMUNITY INTERACTIONS

Proxy information must be used to develop fire regimes for much of the sagebrush steppe zone since little direct information is available. The lack of large presettlement wood or trees that repeatedly scar, such as ponderosa pine, across most of the sagebrush biome, limits our ability to date presettlement fires and determine mean fire return intervals. Presettlement fire-return intervals for some of the wetter mountain big sagebrush associations adjacent to forested communities have been described (Houston 1973, Miller and Rose 1999, Heyerdahl et al. 2006). But descriptions of fire regimes for the majority of plant associations in the sagebrush steppe are lacking. Here we use available literature and ecological–fire–climate relationships to describe presettlement fire regimes for shrub steppe, desert shrub, and woodland associations. This includes: (1) fire scars and charred wood, (2) fine fuel load potentials, and (3) juniper age structure to describe presettlement fire regimes in several mountain big sagebrush plant associations. Tables 11.2 and 11.3 show the fire regimes for the shrub steppe plant communities.

Juniper trees less than 50 years old are easily killed by fire (Burkhardt and Tisdale 1969, 1976; Bunting 1984b; Miller and Rose 1999). Taking into consideration the variability of fire-return intervals, mean fire-return intervals were probably less than 50 years to inhibit woodland encroachment. As fire-return intervals increase in length (more than 50 years) the potential for some trees to survive fire increases. In mountain big sagebrush it is not uncommon to see scattered presettlement trees on less-productive south slopes. However, on the opposing north aspect, evidence of presettlement trees is typically absent unless present on fuel-limited micro sites. In these arid land plant communities, the potential for shorter fire-return intervals decreases along an increasing moisture gradient resulting in more abundant fuels and greater fuel continuity.

Frost (1998) estimated a very general presettlement fire-return interval of 13 to 25 years for the sagebrush region in the Intermountain West. However, this was based on fire-return intervals reported for the more mesic mountain big sagebrush communities in Yellowstone, Wyoming (Houston 1973). Brown (2000) estimated the presettlement fire-return interval for this same region as varying between 35 and 100

years, which probably better characterizes the more arid sagebrush associations that occupy a larger portion of this zone. The combined range of both authors, 13 to 100 years, probably best captures the range of complexity of fire-return intervals that occurred across much of the sagebrush steppe, with return intervals getting increasingly longer going from the mesic to arid big sagebrush communities. In the fuel-limited low sagebrush and black sagebrush communities, fire-return intervals were more commonly greater than 150 years (Young and Evans 1981, Miller and Rose 1999).

Pre- and postsettlement fire regimes among and within desert shrub, salt desert shrub, shrub steppe, and juniper woodlands of northeastern California are spatially and temporally complex. Presettlement fire regimes varied from low-intensity fires occurring at intervals of less than 20 years in some mountain big sagebrush associations (that were more likely grasslands or shrub grassland mosaics) to events rarely occurring in desert salt shrub associations. Even with the mountain big sagebrush series presettlement fire-return intervals on the Lava Beds National Monument range from less than 20 years to greater than 200 years among different plant associations (Miller et al. 2003). Since the late 1800s, fire regimes have significantly changed among many of the plant associations in northeastern California. The reduction of fine fuels through grazing, introduction of exotic weeds, fire suppression, and the removal of anthropogenic burning have significantly altered fire interval, intensity, severity, and season (Burkhardt and Tisdale 1969, 1976; Whisenant 1990; Miller and Rose 1999, Miller and Tausch 2001).

Amount and continuity of fuels are often limiting among many plant associations in the sagebrush biome (Bunting et al. 1987) and associated plant communities. Fire models that describe rate of spread and behavior are usually inadequate for most of these plant associations due to the patchiness of fuels and presence of bare ground (Brown 1982). Minimum levels of fuels required to carry fire in sagebrush steppe communities are 20% shrub cover, 200 to 300 lb ac^{-1} of herbaceous fuel, 8 to15 mph winds, 15% to 20% RH, and 21°C (70°F) to 27°C (80°F) temperatures (Britton et al. 1981). Brown (1982) reported shrub canopies began influencing fire spread at 30% to 40% cover. For fire to carry through woodland under relatively dry conditions, Klebenow and Bruner (1976) reported a minimum of 40% to 60% pinyon, juniper, and shrub cover were required. Wright et al. (1979) concluded that 600 to 700 lbs of fine fuel were necessary to carry a fire through pinyon-juniper woodland. However, wildfires that burn under severe weather conditions are capable of burning across plant communities having very limited fuels.

Areas in this zone with short fire-return regimes were probably grass grasslands or sagebrush-grassland mosaics. As MFRI (median fire-return interval) increase, the physiognomy shifts from grassland to shrub steppe. Sparse stands of western juniper occupied sites with MFRI greater than 50 years. Since the late 1800s, fire events have generally decreased across the mountain big sagebrush series, especially those where presettlement MFRI were less than 100 years. In northern California, the result has been an increase in shrub cover and density and decline in the herb layer, and often a shift from shrub steppe communities to western juniper woodlands (see Sidebar 11.2).

More than 90% of the presettlement fires dated in this region occurred between mid-summer and early fall (Miller and Rose 1999, EOARC data). The likely ignition source was lightning, which usually occurs in dry thunderstorms during July and August. During a two-week period in August of 2001, more than 100,000 strikes occurred in Lassen and Modoc Counties in California and adjacent Lake and Harney Counties in Oregon (BLM, personal communication) resulting in numerous fire starts. Although there is evidence of Native Americans burning in this region, there is no consensus as to magnitude of human-caused fire starts compared with those caused by lightning.

The northwesternmost populations of pinyon pine are found on the edge of the Northeastern Plateaus bioregion and the eastern slopes of the Sierra Nevada; Long Valley Creek in the foothills of the Diamond Mountains, which drains into the Honey Lake 40 km (25 mi) to the east, and is 30 km (19 mi) north of the Loyalton population in Sierra County (David Charlet and Robin Tausch, personal communication). Both stands consist of only a few scattered trees and have been spared by several wildfires that have burned in the vicinity since 1989. Much more expansive stands of pinyon pine occur in the northern Mono section of the bioregion. Their fire ecology is discussed in Chapter 16.

Lower-Montane Zone

The lower-montane zone is wetter and cooler than the sagebrush steppe, and generally occurs in this bioregion where precipitation levels range from approximately 40 cm (16 in) to about 64 cm (25 in) (Figs. 11.3 and 11.4). Vegetation within the lower-montane zone is characterized by ponderosa and Jeffrey pine forests growing on sites with moderately deep to deep well-drained soils. The lower-montane zone in this bioregion occurs on the lower slopes of the Warner and Big Valley Mountains, at the base of the Diamond Mountains, and in large patches on the flat, expansive Devil's Garden area. Slopes are generally less than 30%. At the north end of the bioregion, ponderosa pine is the sole native yellow pine, although Jeffrey pine can be found in older tree plantations such as Sugar Hill (reforested after the 1929 fire) in the north Warner Mountains. South of a latitude line corresponding with Alturas, California, Jeffrey pine mixes with ponderosa pine. Common tree and shrub species are ponderosa pine, Jeffrey pine, western juniper, incense-cedar *(Calocedrus decurrens)*, California black oak *(Quercus kelloggii)*, antelope bitterbrush, mountain big sagebrush, low sagebrush, greenleaf manzanita *(Arctostaphylos patula)*, curl-leaf mountain-mahogany, Utah service-berry *(Amelanchier utahensis)*, and western choke-cherry *(Prunus virginiana)* (Smith and Davidson 2003).

This zone was heavily affected in the twentieth century by logging, grazing, fire exclusion, and possibly climate change.

SIDEBAR 11.2. WESTERN JUNIPER

The northwestern portion of the pinyon and juniper region is represented by western juniper *(Juniperus occidentalis)*. This subspecies occupies more than 3.64 million hectares (9 million acres) in northeast California, eastern Oregon, northwest Nevada, southwest Idaho, and a few outlying stands in southeast Washington (Miller et al. 2005). In California, woodlands (more than 10% tree canopy cover) occupy 520,000 ha (1,284,948 ac), and juniper shrub savannas (less than 10% tree canopy cover) occupy 323,000 ha (798,150 ac) (Bolsinger 1989). It is typically the only conifer occupying the site except where juniper woodlands integrate with ponderosa pine. Precipitation typically averages from 30 to 40 cm (12 to 16 in) across the majority of these woodlands. In its southern limit, south of Susanville, western juniper integrates with the subspecies Sierra juniper *(Juniperus occidentalis* spp. *australis)*, which occurs along the upper-elevation slopes of the Sierra Nevada Mountains.

During the Pleistocene period, the northern range of western juniper was in the Lahontan Lake basin (Thompson et al. 1986, Tausch 1999). In California, it grew in Kings Canyon during maximum glaciation (Thompson 1990). At the end of the Pleistocene (10,000 years B.P.), western juniper began migrating north, reaching northeast California and southeast Oregon during the mid-Holocene period, between 7,000 and 4,000 years B.P. (Bedwell 1973, Mehringer and Wigand 1987). At Lava Beds National Monument in northern California, western juniper seeds and twiglets were found in an ancient wood rat midden dated 5,400 years old (Mehringer and Wigand 1987). From the mid-Holocene to Euro-American settlement, the abundance of western juniper populations have increased and decreased with climate and fire throughout their current range (Mehringer and Wigand 1987, 1990; Wigand 1987). However, expansion of western juniper woodlands during the past 130 years appears to be unprecedented when compared with any other time period in the last 10,000 years (Miller and Wigand 1994).

Prior to settlement, which began around the late 1860s, juniper was primarily confined to rocky surfaces or ridges and pumice sands with sparse vegetation (West 1984, Miller and Rose 1995, Waichler et al. 2001, Johnson and Miller in press). In northeastern California, the most extensive stands of old juniper are found on shallow, heavy clay soils supporting low sagebrush and Sandberg bluegrass *(Poa secunda* ssp. *secunda)*. These stands are typically open with less than 10% canopy cover. On fuel-limited sites, such as rocky outcrops, western juniper can attain ages exceeding 1,000 years (Waichler et al. 2001). In Devil's Garden, located north of Alturas, the majority of old-growth trees growing in association with low sagebrush and Sandberg bluegrass ranged between 200 and 500 years of age (EOARC data 1940) with some exceeding 700 years (Fig. 11.2.1). The widely scattered old trees (canopy cover less than 5%) would suggest the occurrence of infrequent stand-replacing fires.

Western juniper is capable of growing in a broad range of soils. Since the late 1800s, western juniper has rapidly expanded into deeper, more productive soils (Burkhardt and Tisdale 1969, 1976; Adams 1975; Miller and Rose 1995, 1999; Gruell 1999). In northeast California, western juniper has encroached into the mountain big sagebrush and quaking aspen alliances and riparian series. Expansion has been attributed to a decline in fire occurrence, the introduction of livestock, and wetter-than-average conditions between 1870 and 1915. In northeast California and southeast Oregon, western juniper expansion coincides with the reduced occurrence of fire in the late 1800s following the introduction of livestock (Miller and Rose 1999, EOARC data). The lack of old wood or trees on the more productive plant associations suggests fire was frequent enough to limit juniper encroachment into these communities. In nearly a dozen mountain big sagebrush/Idaho fescue sites and adjacent ponderosa pine alliances located in southeast Oregon and northeast California, mean fire-return intervals between 1700 and 1870 varied between 10 and 20 years for small fires and 32 years for large fires (Miller and Rose 1999, EOARC data). Since the

FIGURE 11.2.1. Sagebrush steppe zone. Old-growth western juniper growing in low sagebrush and Sandberg bluegrass in the Devil's Garden, Modoc County. Low site productivity on heavy clay soils yields widely spaced shrubs and herbaceous species making fire ignition and spread difficult. Though the Devil's Garden has the highest number of lightning strikes within the bioregion, fire-return intervals are very long, which has allowed western juniper trees to live up to 500 years in this area. (Photo by Rick Miller.)

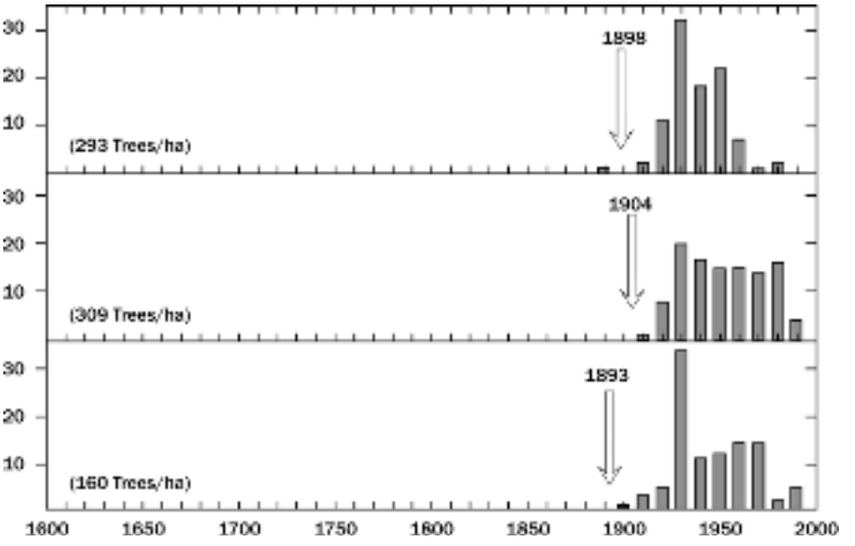

FIGURE 11.2.2. Establishment dates of western juniper trees on three different sites in the Lava Beds National Monument. Prior to settlement these communities were open ponderosa pine/Idaho fescue with scattered mountain big sagebrush and bitterbrush. Presettlement fire-return intervals were 8 to 10 years; last fire is reported in the graph (from Miller et al. 2003).

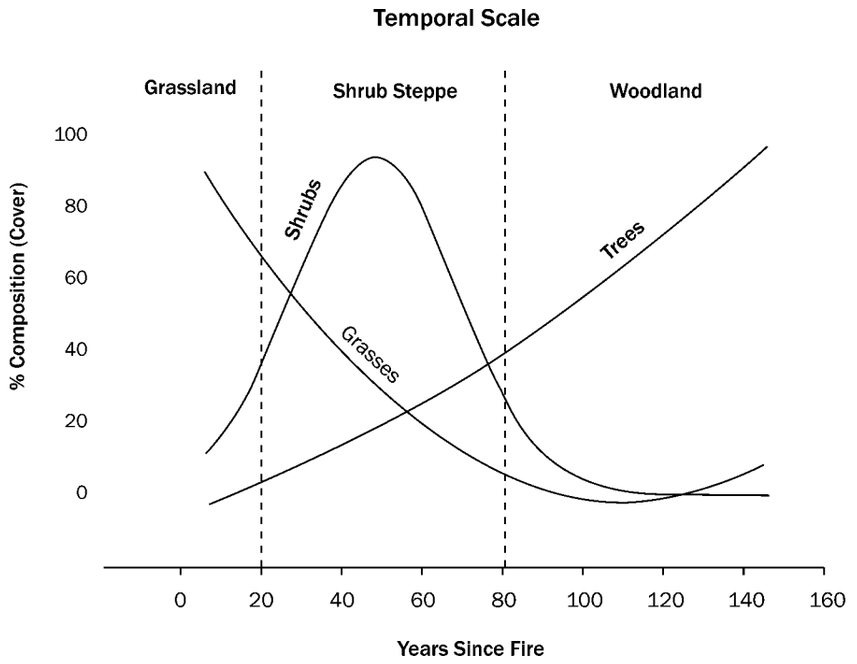

FIGURE 11.2.3. Model of the rate of post-fire succession from grassland to shrub steppe to western juniper woodland. Shrubs and trees co-dominate during the period shown in gray. Percentage composition is derived from measured cover of existing herb, shrub (excluding mountain-mahogany), and overstory (western juniper and mountain-mahogany) at Lava Beds National Monument. Moist sites are those plant associations that contain Idaho fescue while arid sites are those that contain western needlegrass (from Miller et al. 2003).

decline in fire, western juniper is invading these sites. On the Lava Beds National Monument, fire-return intervals were between 8 and 10 years during 1750 and 1904 in open ponderosa pine/Idaho fescue communities (Miller et al. 2003). Following the cessation of fire on these sites, mountain big sagebrush increased followed by the rapid establishment of western juniper and the decline of sagebrush (Figs. 11.2.2 and 11.2.3). Although presettlement fire events were likely less frequent in more arid mountain big sagebrush associations, fire-return intervals would probably have to have been less than 50 years to limit western juniper encroachment into shrubland communities (Burkhardt and Tisdale 1976, Bunting 1984a, 1984b, Miller and Rose 1999). As the length of fire-return intervals increases with more limited fuels on increasingly drier sagebrush communities, scattered trees can establish and occupy the site. A relatively dry western juniper, mountain big sagebrush, and Thurber needlegrass plant association in the Lava Beds National Monument supported infrequent moderate- to high-severity fires. Two fire events recorded on this site, one in the early 1700s and a second in 1856, resulted in scattered patches of old western juniper trees and an abundance of charred wood and stumps (Miller et al. 2003).

The majority of western juniper woodlands in Oregon and California are still in transition from shrub steppe to juniper woodland. Although western juniper has increased tenfold in the past 130 years, it currently occupies far less land than it is capable of under current climatic conditions (Miller et al. 2000). During the early stages of encroachment, juniper initially adds structural diversity to shrub steppe communities, which often increases wildlife abundance and diversity (Reinkensmeyer 2000, Miller 2001). However, structural diversity declines as woodlands become fully developed and structural complexity, productivity, and diversity of understory vegetation declines (Miller et al. 2000). In Modoc County north

FIGURE 11.2.4. Sagebrush steppe zone. Many of the lower-elevation and drier aspen stands have succumbed to juniper succession. This process has occurred due to: (1) reduction of competition from herbaceous plants primarily by livestock grazing, (2) browsing of aspen regeneration by livestock and native ungulates, (3) soil compaction primarily by livestock, (4) fire exclusion, and (5) the complex interactions of factors 1 through 4. (Photo by Rick Miller.)

of Alturas, western juniper overstory increased and live shrubs decreased in permanent transects measured from 1957 to 1998 (Schaefer et al. 2003). Sagebrush, an important ladder fuel during the early stages of woodland development decreases as juniper dominance on the site increases. The production of fine fuels can also decrease with the decline of sagebrush, essentially fire-proofing the site.

Western juniper began invading quaking aspen communities in the 1890s, with peak establishment occurring between 1900 and 1940 (Fig. 11.2.4). In nearly 100 quaking aspen stands studied across north eastern California, southeastern Oregon, and northwestern Nevada below 2,100 m (6,890 ft), western juniper was present in over 90% of the stands (Wall et al. 2001). They reported 12% of the quaking aspen stands were completely replaced by western juniper; juniper was the dominant tree species in 23% of the stands and was common to codominant in 42%. No juniper tree more than 140 years of age was measured across these stands, nor was large juniper wood found. Prior to 1900, disturbance intervals within relatively large quaking aspen groves occurred between 10 to 16 years, with total stand replacement occurring about 60 to 100 years (Romme et al. 1996, Wall et al. 2001). In southeastern Oregon, juniper trees have been rapidly encroaching into riparian areas (EOARC data). The lack of old wood or presettlement trees in both of these series suggests mature western juniper was probably absent prior to Euro-American settlement.

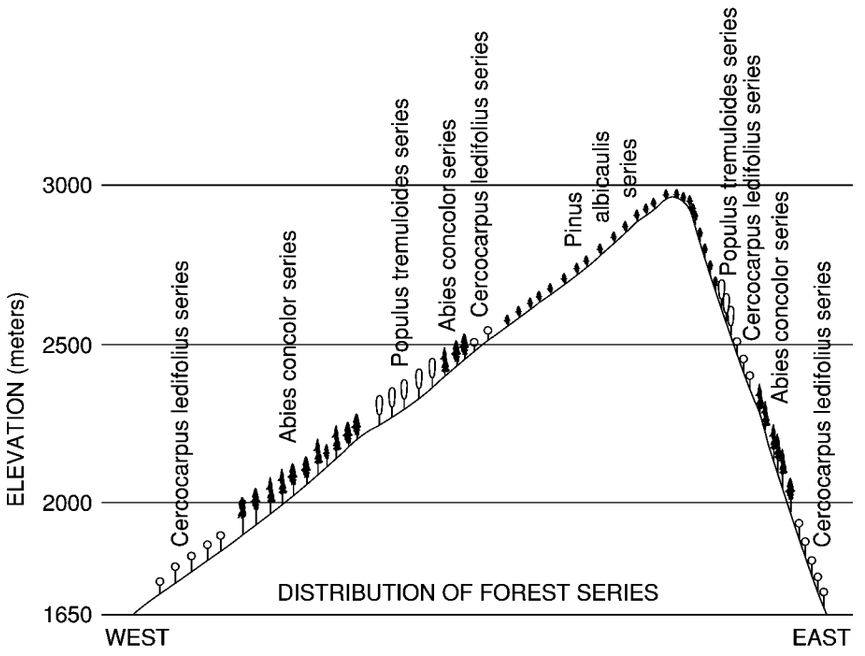

FIGURE 11.3. Distribution of climatic potential vegetation types and elevation relationships in the South Warner Mountains (Riegel et al. 1990). Fire suppression has allowed for white fir expansion into (1) Jeffrey and ponderosa pine sites in the lower-montane zone that typically occur from 1,670 to 2,043 m (5,479 to 6,703 ft), and (2) Washoe pine sites from 1,890 to 2,195 m (6,201 to 7,201 ft) in the mid-montane to upper-montane zones. The upper-montane zone spans from 1,829 to 2,390 m (6,001 to 7,841 ft) where white fir is both the historic dominant under a natural fire regime as well as the climatic potential with occasional stands of western white pine.

FIGURE 11.4. Lower-montane zone. Landscape view of ponderosa and Jeffrey pines, California black oak, western juniper, and antelope bitterbrush dispersed between large expanses of low and mountain big sagebrush on the Big Valley Ranger District, Modoc National Forest. (Photo by Sydney Smith.)

In general, stands are much denser than in the late nineteenth century, and trees are much smaller, larger ones having been removed mostly by logging. Fuel loadings are likely quite high compared to pre-twentieth century levels because of dense trees, dead trees, and accumulations of logging slash (Smith and others, personal observations).

FIRE RESPONSES OF IMPORTANT SPECIES

Before the twentieth century, frequent surface fires of low to moderate intensity removed needle and branch litter and created exposed surface mineral soil where yellow pines, both ponderosa and Jeffrey, and incense cedar seeds can germinate and grow (Table 11.6). Winged seeds of these species are wind dispersed in the late summer through fall. Depending on the time at which fire returns to a site these seeds may be eaten or cached by animals, consumed by fire, or fall on mineral surface soil.

Jeffrey pine occurs on the coldest sites, higher elevations, steeper slopes, and shallower soils relative to ponderosa pine (Smith 1994). Thus, Jeffrey pine sites are generally lower in productivity compared with ponderosa pine sites (Jenkinson 1990). Antelope bitterbrush is the most widespread understory dominant associated with Jeffrey pine forests.

TABLE 11.6
Fire response types for important species in the lower montane zone

	Type of Fire Response			
Lifeform	Sprouting	Seeding	Individual	*Species*
Conifer	None	None	Resistant/killed	Ponderosa pine, Jeffrey pine, incense cedar, western juniper
Hardwood	Fire stimulated	None	Resistant/killed	California black oak
Shrub	Fire stimulated	Fire stimulated	Top-killed	Mahala mat, greenleaf manzanita, western choke-cherry
	Fire stimulated	None	Top-killed	Utah service-berry, bitter cherry, Modoc plum
	+/−Fire stimulated	None	Killed	Antelope bitterbrush
	None	None	None	Mountain big sagebrush
Forb	Fire stimulated	None	Top-killed	Woolly mule's ears, arrowleaf balsam-root, lambs tongue ragwort, lupine, Nevada wild pea, lomatium, peony, Indian paintbrush
Graminoids	+/− Tillering	None	Top-killed	Idaho fescue
	Tillering	None	Top-killed	Squirreltail, Ross' sedge, Canby bluegrass
	None	None	Killed	Cheat grass

Utah service-berry may be slightly injured by fire, depending on moisture conditions and flame lengths but is generally considered to be fire tolerant (Crane 1982, Bradley et al. 1992). However, mortality does occur if a thick litter layer is present at the time of the fire. Utah service-berry sprouts from adventitious buds at the root crown and it is also a prolific seed producer with persistent animal-dispersed berries.

Fire often kills aboveground choke-cherry stems and foliage, but it quickly resprouts from root crowns and rhizomes (Volland and Dell 1981, Young 1983). Fire-induced sprouts can be produced within the growing year following a spring burn, or if a fall burn, by the next spring. It can double its stem density within one growing season prior to being burned. Large, animal-dispersed seeds aid in post-fire regeneration. Scarification of the seed coat by the heat from fire improves germination.

The primary response of herbaceous plants to a prescribed fire on the Fremont National Forest, in south-central Oregon, was a slight increase in diversity and a change in the relative dominance of graminoid species, Ross' sedge *(Carex rossii)*, and Idaho fescue (Busse et. al. 2000). Squirreltail and Ross' sedge both increased in biomass and cover, whereas fescue cover declined. Fire stimulates squirreltail root and shoot biomass of surviving plants and increases seedling recruitment (Young and Miller 1985, Vose and White 1991). Ross' sedge has shallow rhizomes and is capable of responding rapidly to disturbance. In comparison, fescue, a tufted bunchgrass, varies from moderate to severe susceptibility to fire, depending on the amount of detritus in the basal tufts that cause fire to linger and kill the perennating buds (Wright 1971, Johnson 1998) (see further discussion in the sagebrush steppe zone above). The most common species typically found after burning are tailcup lupine *(Lupinus caudatus)*, Nevada pea *(Lathyrus lanszwertii)*, nineleaf biscuitroot *(Lomatium triternatum)*, sticky cinquefoil *(Potentilla glandulosa)*, Brown's peony *(Paeonia brownii)*, and fireweed *(Epilobium angustifolium)*.

FIRE REGIME–PLANT COMMUNITY INTERACTIONS

Though no fire history studies have been conducted in the Jeffrey pines forests in northeastern California, Smith (1994) speculates that return intervals were short, probably ranging from 5 to 20 years (Table 11.7). In the ponderosa pine/antelope bitterbrush-tobacco brush/needlegrass plant association on Pringle Butte in Central Oregon (Deschutes National Forest), between 1362 and 1900, the MFRI ranged from 7 to 20 years where mean annual precipitation was 61 cm (24 in) (Bork 1985). However, Jeffrey pine, which reaches its eastern geographic range limit in Modoc County (Little 1971, Griffin

TABLE 11.7

Fire regime characteristics for the lower montane zone in the Northeastern Plateaus bioregion

Vegetation type	Jeffrey pine	Ponderosa pine	Jeffrey-ponderosa pine
Temporal			
Seasonality	Summer–early fall	Summer–early fall	Summer–early fall
Fire-return interval	Short–medium	Short	Short–medium
Spatial			
Size	Medium	Medium	Medium
Complexity	Medium–high	Low–medium	Low–high
Magnitude			
Intensity	Low	Low	Low–moderate
Severity	Low	Low	Low–moderate
Fire type	Surface	Surface	Surface

and Critchfield 1972), is absent from these Central Oregon sites. Thus the fire-return intervals may be shorter than would be expected for Jeffrey pine. The harsher conditions usually associated with Jeffrey pine sites, as noted above, usually leads to slower fuel accumulation, typically resulting in longer fire return intervals than found on sites dominated by ponderosa pine (Skinner and Chang 1996, Taylor 2000, Stephens 2001, Skinner 2003, Stephens et al. 2003) (see also Chapters 9 and 10, this volume).

Within the lower montane zone, fire-return intervals were probably longer at the drier and/or lower-elevation sagebrush steppe zone ecotone than in the mid and upper elevations, regardless of dominance Jeffrey pine or ponderosa pine. Bork (1985) reported an MFRI of 16 to 38 years, from 1460 to 1970, in a ponderosa pine/antelope bitterbrush-mountain big sagebrush/Idaho fescue plant association on pumice soils, where mean annual precipitation is 24 cm (9 in). Fire history was reconstructed from scattered ponderosa pine growing within mountain big and low sagebrush communities in the upper Chewaucan River basin near Paisley, Oregon, where precipitation was approximately 40 cm (16 in). MFRI ranged from 3 to 38 years between the years 1601 and 1897 (Miller and Rose 1999). Fire return intervals from a relatively small sample size were reported along an elevation and vegetation gradient in the Buck Creek watershed on the west side of the North Warner Mountains (Goheen 1998). For the period between 1650 and 1879, return intervals were four times longer (MFRI ranged from 2 to 56 years) on a lower-elevation (1,567 m [5,140 ft]) juniper and ponderosa pine site than in the middle-elevation (1,735 to 1,905 m [5,690 to 6,250 ft]) ponderosa pine sites (MFRI ranged from 1 to 11 years). Vegetation spatial pattern is more heterogeneous on these lower-elevation sites as limited soil moisture regulates plant density and biomass production. Both limited fuel quantities and lack of continuity result in longer intervals between fires. Additionally, slopes are not as steep on the lower-elevation sites and would be less likely to produce fires intense enough to scar trees as on steeper slopes.

Busse et al. (2000) demonstrated that low-intensity prescribed fire following a prolonged period of wildfire exclusion produced a significant, yet slight reduction in tree growth in thinned ponderosa pine stands. Basal area growth reduction was proportional to the level of crown scorch and "O" horizon reduction during burning. No evidence was found to suggest that prescribed fire will have a long-term impact on stand productivity. Specifically, soil resources, including nutrient content and exposed mineral soil, were unaffected by fire.

Woolly mule's ears *(Wyethia mollis)* is a common and abundant understory species of Jeffrey, ponderosa, and Washoe pine forests east of the Cascade Range from south-central Oregon and northeastern California through the eastern Sierra Nevada of California and Nevada (Hopkins 1979, Riegel 1982, Smith 1994, Sawyer and Keeler-Wolf 1995), as an associate species in the montane and subalpine sagebrush steppe throughout this range (Cronquist 1994). An herbaceous, disturbance-adapted long-lived perennial (more than 50 years) with a deep taproot and fleshy resinous foliage, woolly mule's ears was most likely an early successional plant with low frequency and cover prior to Euro-American arrival (Young and Evans 1979, Williams 1995). Woolly mule's ears is also one of the first plants to initiate growth in the spring and typically completes its flower and fruit set by early August. Under current fire regimes, its early season phenology and taproot provide a competitive advantage to exploit soil resources for expansion and dominance (Rundel et al. 1988, Agee 1993, Barbour and Minnich 2000). Heavy livestock grazing of palatable plants from 1860 to the 1930s, primarily by sheep, also resulted in increased woolly mule's ears abundance (Coville 1898, Kennedy and Doten 1901, Leiberg 1902, USDA Forest Service 1937, Olmstead 1957). Woolly mule's ears continues to dominate sites in Lassen National Park that have been excluded from livestock grazing for more than 80 years (Oswald et al. 1995). Woolly mule's ears-dominated sites may suppress succession to mid-seral

FIGURE 11.5. Mid-montane zone. The mid-montane zone provides habitat for many woody species. In the foreground, ponderosa pine is regenerating at the edge of a mountain big sagebrush and woolly mule's ears site. Sagebrush is edaphically controlled by higher soil clay content, which limits plant-available water. Quaking aspen occurs, as both small clones comprised of several trees growing within conifer stands or as larger clones that are topographically controlled where the water table is closer to the surface. Historically, discontinuous fuel types and varying biomass productivity contributed to complex burn patterns and fire intensity. With fire exclusion, many smaller quaking aspen clones are being lost to confer succession. In the background, mid-montane through subalpine zones are visible in the South Warner Wilderness Area. (Photo by Gregg Riegel.)

conifer regeneration for up to 100 years (Kennedy and Doten 1901, Evanko 1951, Parker and Yoder-Williams 1989) and can persist in the understory of mid- and late-seral Jeffrey pine overstories as an infrequent, low cover (less than 5 %) associate (Smith 1994).

Water-extractable allelochemicals from woolly mule's ears have been suggested as an additional factor in suppressing Jeffrey pine regeneration (Heisey and Delwiche 1983, Yoder-Williams and Parker 1987, Parker and Yoder-Williams 1989, Williams 1995). In a field bioassay, Yoder-Williams and Parker (1987) found woolly mule's ears litter leachates inhibited germination and reduced radicle elongation of pine seedlings. However, results of a greenhouse study did not support the field bioassays discussed above and concluded that allelochemicals or soil nutrients do not limit regeneration of Jeffrey pine (Riegel et al. 2002).

Hardwood species are relatively rare in northeastern California due to extreme cold minimum temperatures and aridity. Current populations of California black oak and Oregon ash *(Fraxinus latifolia)* are at their furthest northeast distribution (Little 1971, Griffin and Critchfield 1972) and are considered relict remnants from a time when the regional climate experienced warmer minimum temperatures, higher precipitation, and lower evapotranspiration (Major 1988). Oregon oak achieves its most interior distribution here with the exception of those found along the Columbia River (Little 1971, Griffin and Critchfield 1972). These disjunct populations are primarily found on shallow clay-textured soils within fractured basalt flows (e.g., Egg Lake and Ash Creek) and secondarily, in the Pit River Canyon. Fire-return intervals were likely less frequent because ignition must be initiated on site due to the discontinuity of vegetation in basalt flows and shallow soils matrix.

The largest population of Modoc cypress *(Cupressus bakeri)* occurs on the northwestern edge of the bioregion on the corners of Modoc, Shasta, and Siskiyou counties (Stone 1965, Griffin and Critchfield 1972). This 2,833-ha (7,000-ac) population is found on recent basalt flows near Timbered Crater at elevations from 1,067 to 1,219 m (3,500 to 4,000 ft). Fire is the only effective process that opens the serotinous cones and creates conditions for successful regeneration required for reproduction (Vogl et al. 1988). Most cones remain closed until the resinous seal between the ovuliferous scales is heated by fire, allowing seeds to shed for up to several months. Stands with limited fuel continuity probably lived longer than those growing within ponderosa pine with more frequent fire return intervals. Heat generation from low fire intensity may not be hot enough for resins to come to a boil and melt. Cones will also open when the resinous seals dry, though not as wide as with fire, following mechanical detachment such as disturbance caused during logging operations. Knobcone pine *(Pinus attenuata)*,

TABLE 11.8
Fire response types for important species in the mid-montane zone

Lifeform	*Type of Fire Response*			*Species*
	Sprouting	Seeding	Individual	
Conifer	None	None	Resistant, killed	Ponderosa pine, Jeffrey pine, white fir, incense cedar
Shrub	Fire stimulated	Fire stimulated	Top-killed	Mahala mat, greenleaf manzanita, tobacco brush
	Fire stimulated	None	Top-killed	Bitter cherry, western choke-cherry, Modoc plum, mountain snowberry, creeping barberry
	None	None	None	Mountain big sagebrush
Forb	Fire stimulated	None	Top-killed	Heartleaf arnica, tuber starwort, hawkweeds, lupines, Nevada wild pea, Sierra pea, sweet cicely
Graminoids	Tillering	None	Top-killed	Squirreltail, Ross' sedge, Wheeler's bluegrass, Canby bluegrass, needlegrasses, orcutt brome

another closed-cone fire-adapted species is discussed in Chapters 9, 10, and 14.

Mid-Montane Zone

The mid-montane zone in this bioregion is defined as the area where the climate supports the lower-elevation white fir forests (Figs. 11.3 and 11.5). The mid-montane zone occurs in the Big Valley Mountains, the northern part of the Devil's Garden, and in the Warner Mountains. This zone is colder and wetter and usually at higher elevations than the lower montane zone. Average annual precipitation in this zone ranges from about 64 cm (25 in) to 82 cm (32 in) and elevations range from about 1,430 to 1,980 m (4,700 to 6,500 ft) in the Big Valley Mountains and about 1,615 m (5,300 ft) to as high as 2,225 m (7,300 ft) in the Warner Mountains (Smith and Davidson 2003). White fir occurs mixed with ponderosa pine, Jeffrey pine, incense cedar, and as more-or-less pure stands. Common tree and shrub species in addition to those already mentioned include western juniper, quaking aspen, California black oak, antelope bitterbrush, mountain big sagebrush, low sagebrush, bitter cherry *(Prunus emarginata)*, western choke cherry, Modoc plum *(Prunus subcordata)*, Utah service-berry, roundleaf snowberry *(Symphoricarpos rotundifolius)*, greenleaf manzanita, and tobacco brush *(Ceanothus velutinus* var. *velutinus)* (Smith and Davidson 2003).

This zone presents unusual fuels management challenges because of: (1) changes in stand species composition and structure in the twentieth century, and (2) susceptibility of trees to mortality from drought, disease, and insects. Because of preferential logging of ponderosa pine, fire exclusion, and climate variation, white fir has generally increased, whereas the pines have decreased. These changes have dramatically altered the fire regime in this zone. Stand physiognomy has generally changed from less dense or relatively open stands of large trees to much denser stands of small trees. Extensive tree mortality from insects and diseases has accompanied periodic droughts. Thus, the potential for high-intensity fires increases as these mortality episodes lead to increased fuel accumulations from standing and fallen dead trees (Smith and others, personal observations).

FIRE RESPONSES OF IMPORTANT SPECIES

Jeffrey pine, ponderosa pine, incense cedar, and white fir all develop thick bark as they age which allows for their survival in characteristic frequent and low-intensity fire (Table 11.8). Jeffrey and ponderosa pine are more resistant to fire when young compared with white fir or incense cedar (van Wagtendonk 1983, Agee 1993). White fir sapling and pole-sized trees are very susceptible to fire-caused mortality due to (1) numerous resin blisters on the bark that increase potential for flammability, (2) flat, lateral branches that often extend to the ground especially in trees that are in canopy gaps and isolated "wolf trees", and (3) comparatively shallow roots that increase susceptibility to lethal temperatures when surface

fires slowly burn through high fuel loads in deep and compact litter (Agee 1993, Miller 2000).

Fire profoundly affects individual species responses which in turn greatly influence successional dynamics in the mid-montane zone. Surface fires burn litter, temporarily leaving an ashy mineral soil surface that ponderosa and Jeffrey pines seeds require for successful germination, whereas incense cedar tolerates a variety of surface substrates (Powers and Oliver 1990). White fir is the only tree species in this bioregion that can germinate on top of litter, which typifies sites with longer fire-return intervals.

Following fire, greenleaf manzanita and tobacco brush prolifically spout, whereas long-lived seed bank reserves are released from dormancy ensuring the establishment or perpetuation of seral montane shrubfields. Mountain snowberry response to fire is quite variable. Most plants survive fire and typically resprout from basal buds at the root crown. However, mountain snowberry is considered a weak sprouter, especially after severe fire (Young 1983) (Table 11.5). Low-severity fire can kill branches as well as all the above-crown portions of a plant, making it difficult to predict individual plant survival. Even following severe fires, mountain snowberry root crowns usually survive (Pechanec et al. 1954, Kuntz 1982). Sprouting can occur the first year but may be initially limited, eventually reaching preburn levels within 15 years (Pechanec et al. 1954, Kuntz 1982).

TABLE 11.9
Fire regime characteristics for the mid-montane zone in the Northeastern Plateaus bioregion

Vegetation type	White fir and incense-cedar
Temporal	
Seasonality	Summer-early fall
Fire-return interval	Medium
Spatial	
Size	Medium
Complexity	Medium
Magnitude	
Intensity	Multiple
Severity	Multiple
Fire type	Multiple

FIRE REGIME–PLANT COMMUNITY INTERACTIONS

Historic fire regimes are similar to those described for the east montane zone of the Sierra Nevada and the mid-montane eastside zone of the Southern Cascade Range. Fire-return intervals were short, and intensity and severity were low (Table 11.9). Intensive logging of ponderosa pine, coupled with fire suppression and exclusion, has shifted species composition from pine-dominated and co-dominated stands to overstories where white fir is the dominant species. Fire helped to regulate stand density and suppressed the continual recruitment of white fir. The exclusion of fire has helped promote the shift to white fir dominance. White fir seedlings, which establish best in partial shade, can survive under the canopy of montane shrubs and dense forests, but once established, grow best in full sunlight (Laacke 1990). Shade-tolerance ranking for tree regeneration (seedlings and saplings) from most to least tolerant is white fir, incense cedar, and ponderosa pine and Jeffrey pine (Minore 1979). White fir is also a prolific tree, producing up to 1.5 million seeds ha^{-1} (6,000,000 ac^{-1}) generally every 2 to 5 years, beginning at age 40 and continuing beyond 300 years (Laacke 1990).

When precipitation is greater than or equal to the annual mean for a site, white fir can be very successful with rapid growth and continual reproduction and recruitment. However, in the 64-cm (25-in) precipitation zone in northeastern California, due to recurring droughts, white fir tends to become stressed, especially where the denser stands have developed, and eventually succumbs to insects and diseases—often *before* a fire occurs. White fir's stomatal regulation, water use efficiency (WUE), and ability to control transpiration per unit of fixed carbon is far less than that of ponderosa pine (Hinckley et al. 1982). If white fir is under moisture stress (−2.0 or greater MPa) for several summers, they are more susceptible to successful fir engraver attacks than are trees under less stress (Ferrel 1978). Recurring drought is the common denominator that regulates a site's ability to produce more leaf area and stand density. Cochran (1998) advises that if prolonged droughts are forecast, removal of white fir on drier sites is recommended. Fire would have regulated white fir regeneration under a function of historic fire regime on these sites.

Despite extended drought in the southeastern Warner Mountains, Vale (1975) found that white fire invaded the mountain big sagebrush ecotone between 1915 and 1944. He concluded that heavy livestock grazing of competing vegetation was the main factor and that successful fire suppression, which did not occur until after the trees had been established, was secondary. Eventually these trees on the ecotone will be eliminated by (1) drought, (2) stress, (3) insect interactions, or (4) fire.

Outbreaks of Modoc budworm defoliating white firs occurred during the years 1959 to 1962 and 1973 to 1975 in the Warner Mountains and nearby ranges of northeastern California (Ferrel 1980). These outbreaks were associated with periods of three years or more when precipitation averaged at least 25% below the historic mean annual precipitation. A Douglas-fir Tussock Moth infestation occurred from 1959 to 1962 on 182 ha (450 ac), and from 1964 to 1965 another infestation defoliated white fir populations in second-growth pine stands in the Warner Mountains between 1,700 and 1,900 m (5,600 and 6,200 ft). Wickman (1978) interpreted from the existing number and size of stumps that the prelogged forest had been a ponderosa and Washoe pine stand with large trees, 91 to 102 cm dbh (36 to 40 in), before being logged around the early 1900s. Though the area had been

FIGURE 11.6. Upper-montane zone. Landscape perspective of the Owl Creek watershed on the east side of the Warner Mountains showing the patchy, discontinuous nature of forest in this zone. Photo was taken during an August thunderstorm. (Photo by Carl Skinner.)

logged a second time for white fir overstory trees in 1954, Wickman noted that no fires had burned in the area for at least 50 years and more likely 75 years. The effect of logging the pine and suppressing and excluding fire has been to allow white fir to become the dominant tree species in all structural and age classes. The consequence of this change is that the current forest is more susceptible to high-intensity fire because of large amounts of live and dead fuels. The ensuing high-intensity fires have produced large landscape-level stand-replacing events such as the Scarface fire (1977), the Crank fire (1987), and the Blue fire (2001).

After a forest is removed by fire or logging, the developing shrub and herbaceous understory often suppresses conifer regeneration as a result of competition for soil moisture (Conard and Radosevich 1981, 1982; McDonald 1983b; Lanini and Radosevich 1986; Shainsky and Radosevich 1986; Parker and Yoder-Williams 1989). Though there are many pathways succession can take following disturbance in mid-montane pine forests, a generalized summary includes: (1) early seral–woolly mule's ears dominance (discussed above in the lower-montane zone) with associated herbaceous species and some conifer regeneration; (2) mid-seral–greenleaf manzanita, tobacco brush, mahala mat *(Ceanothus prostratus)*, and antelope bitterbrush shrub dominance with pole-sized conifers; and (3) late seral–conifer dominance with some shade-tolerant herbs and shrubs.

The shrub species associated with the mid-seral stage compete with establishing conifers for light, water, and nutrients (Conard and Radosevich 1981, 1982; McDonald 1983a, 1983b; Roy 1983; Radosevich 1984; Shainsky and Radosevich 1986). A shrub canopy also provides a shaded microenvironment, however, that reduces evaporative demand and improves the water balance of trees, which may be less physiologically stressful than competition for soil resources on hot, dry sites (Conard and Radosevich 1982, Lanini and Radosevich 1986). Soil moisture availability on shrub-dominated sites can be significantly greater than on woolly mule's ears–dominated sites during the latter part of the growing season (Williams 1995).

Shrub dominance in the mid-seral phase should increase the soil nutrient capital with time-between-fire-and-logging disturbance intervals that temporarily reduce the cover of nitrogen-fixing shrubs, tobacco brush, mahala mat, and antelope bitterbrush (Conard et al. 1985; D. W. Johnson 1995; Busse et al. 1996; Busse 2000a, 2000b). Nitrogen fixation occurs from actinorhizal symbionts associated with tobacco brush, mahala mat (Delwiche et al. 1965; Conard et al. 1985; Busse 2000a, 2000b), and antelope bitterbrush (Webster et al. 1967; Busse 2000a, 2000b). Annual nitrogen fixation in forests east of the Sierra Nevada and Cascade Range crest varies from 5 to 15 kg ha^{-1} (5 to 15 lb ac^{-1}) for tobacco brush and 1 kg ha^{-1} (1 lb ac^{-1}) for mahala mat and antelope bitterbrush (Busse 2000a, 2000b). Busse et al. (1996) found increased soil carbon, nitrogen, and microbial biomass in the upper horizon of a ponderosa pine forest due to long-term retention of shrubs from longer fire return intervals due to fire exclusion. D. W. Johnson (1995) also found improved soil nitrogen status at pine sites with a dominance of tobacco brush. With successful fire exclusion for nearly 50 years, nitrogen-fixing shrubs that would have been periodically consumed in historic fire regimes increased radial-increment growth in naturally regenerated ponderosa pine stands in Central Oregon (Cochran and Hopkins 1991). Succession in these forests may be more influenced by competition for soil moisture and changes in microclimate.

Upper Montane Zone

The upper montane zone in this bioregion is defined as the area where the climate supports the higher-elevation white fir forests (Figs. 11.3, 11.6, and 11.7). The upper-montane

FIGURE 11.7. Upper-montane zone. Photo shows the pattern of exposed slopes dominated by shrubs and herbaceous plants with trees confined mostly to more protected sites such as the draw in the foreground and the patch of trees in the upper center on the lee side of the crest. (Photo taken by Carl Skinner near High Grade Spring north of Mt. Vida.)

zone occurs at elevations of about 1,830 to 2,225 m (6,000 to 7,300 ft) in the Big Valley Mountains, and at elevations between about 1,800 and 2,430 m (5,900 to 8,000 ft) in the Warner Mountains (Smith and Davidson 2003). This zone is cold and wet compared to the mid-montane zone. Average annual precipitation in this zone exceeds 81.5 cm (32 in). The vegetation of the zone is quite patchy responding to soil moisture, edaphic variation, and winter wind patterns. The patches are of several general types: (1) conifer-dominated, (2) shrub-dominated, (3) herbaceous-dominated, and (4) rocky, relatively barren areas primarily along exposed ridges. White fir occurs in pure stands and in mixes with western white pine, lodgepole pine, quaking aspen, occasional yellow pines (ponderosa, Jeffrey, or Washoe pines), and whitebark pine at the upper elevations of this zone (Riegel et al. 1990, Smith 1994). Western juniper at these higher elevations can still be found on exposed sites with shallow soils. Common shrubs include mountain big sagebrush, snowfield sagebrush, low sagebrush, creeping snowberry *(Symphoricarpos mollis)*, sticky currant *(Ribes viscosissimum)*, bitter cherry, tobacco brush, curl-leaf mountain-mahogany, and pinemat manzanita *(Arctostaphylos nevadensis)* (Riegel et al. 1990, Smith and Davidson 2003). We are aware of no published fire research conducted in this zone. Most of the following discussion is based on the authors' observations and discussions with local fire and resource managers.

Although this zone has been affected by twentieth-century logging, grazing, and climate change, the fire regime, as indicated by conifer and shrub species composition, has likely been affected the least of the montane zones in this bioregion. One major change, however, is the decrease in the extent and occurrence of quaking aspen. For comparison, relative changes from historic to current fire regime ranked from least changed zone to the most changed zone are upper montane, lower montane, and mid-montane.

FIRE RESPONSES OF IMPORTANT SPECIES

Scouler's willow (*Salix scouleriana*) is fire sensitive, and crown mortality can be 0% to 100% depending on fire intensity (Owens 1982) (Table 11.10). Severe fires may result in 100% aboveground mortality (Lyon and Stichney 1976). Following low- to moderate-intensity fire, Scouler's willow will quickly resprout. High-intensity fire that kills live foliage but does not kill the vascular cambium will cause vigorous epicormic sprouting from the root crown (Weixelman et al. 1998). Sprouting typically occurs within days following a fire and it is not uncommon for a plant to reach and exceed preburn frequency and cover in 5 years or less. Scouler's willow also produces windborne seeds that can travel for considerable distances off site.

Bitter cherry response to fire is variable. It can be killed if burned while actively growing or if fire intensity is high (Young 1983). Resprouting occurs from the root crown. It also establishes from large buried seed or seed dispersed from off site by animals, primarily birds.

Creeping snowberry is usually top-killed by fire; however, vegetative regrowth does occur from rhizomes (Volland and Dell 1981). Fire will kill rhizomes if surface litter layers are thick and/or organic matter constitutes a significant portion of the upper soil horizon (Neuenschwander n.d., cited in FEIS).

Sticky currant is considered only moderately resistant to fire (Volland and Dell 1981). Low- to moderate-intensity fire can stimulate rapid resprouting from basal stems. Seeds are relatively heavy and animal dispersed. Heat scarification increases germination rates.

Pinemat manzanita is killed by fire but re-establishes from seed during the first post-fire growing season. Kruckeberg

TABLE 11.10
Fire-response types for important species in the upper-montane zone

	Type of Fire Response			
Lifeform	Sprouting	Seeding	Individual	*Species*
Conifer	None	None	Resistant/killed	White fir, Washoe, ponderosa, Jeffrey, and western white pine, lodgepole pine
Hardwood	Fire stimulated	None	Top-killed	Quaking aspen
Shrub	Fire stimulated	Fire stimulated	Top-killed	Tobacco brush, bush chinquapin
	Fire stimulated	None	Top-killed	Bitter cherry, creeping snowberry, mountain snowberry, Scouler's willow, sticky currant, snowfield sagebrush
	None	Fire stimulated	Top-killed	Pinemat manzanita
	None	None	Killed	Mountain big sagebrush
Forb	Fire stimulated	None	Top-killed	Heartleaf arnica, tuber starwort, hawkweeds, lupines, slender penstemon, whiteveined wintergreen, sweet cicely
Graminoids	Tillering	None	Top-killed	Squirreltail, Ross' sedge, Wheeler bluegrass, needlegrasses, orcutt brome, Brainerd's sedge

(1977) speculated that it may be an obligate seeder, requiring fire and/or leachate from smoke or charred wood to break seed dormancy.

FIRE REGIME–PLANT COMMUNITY INTERACTIONS

Fuel-accumulation rates—not ignitions—are probably the limiting factor for fire occurrence in this zone of frequently occurring summer thunderstorms. The generally dry climate coupled with cold winters contributes to slow fuel accumulation. Although episodes of widespread lightning are common in summer, ignitions are not likely a limiting factor for fire occurrence. The predominance of short-needle conifers and the winter snow compact the fuel bed. Thus, fires started by the frequently occurring thunderstorms are often slow-spreading, patchy, low-intensity surface fires becoming more intense only where concentrations of fuels from dead trees or small thickets, usually of white fir regeneration, have developed. Accumulations of dead fuels are generally caused by (1) the stem-exclusion phase of white fir regeneration patches, (2) beetle-killed patches, or (3) deaths of individual large trees.

Some areas in this zone that were once dominated by very large yellow pines have converted to lodgepole pine and white fir (Vale 1977). Examples of this are found: (1) on the north-facing slopes near Lily Lake where large stumps of both yellow pines and incense-cedar are found in what are now dense thickets of mostly lodgepole pine, and (2) along the west side of the main crest of the Warner Mountains in the vicinity of the headwaters of the south fork of Davis Creek where large stumps of yellow pines are found in lodgepole pine/white fir stands. In both of these cases no yellow pine regeneration is evident. It is likely the original fire regimes for these specific sites have changed dramatically with this change in species. The South Warner Wilderness Area provides an outstanding example of succession from yellow pines to white fir without logging but with fire exclusion (Riegel et al. 1990).

There are three yellow or three needle pines (subgenus *Diploxylon of Pinus*)—ponderosa, Jeffrey, and Washoe—that occur in the upper montane zone in the South Warner Mountains (Haller 1961, Critchfield and Allenbaugh 1969, Griffin and Critchfield 1972, Riegel et al. 1990, Smith 1994). Washoe pine, which grows at elevations from 1,890 to 2,195 m (6,199 to 7,200 ft) has attracted considerable interest to the Warner Mountains from taxonomists and geneticists (Haller 1961, Smith 1967, 1971, 2000, Critchfield 1984, Conkle and Critchfield 1988, Niebling and Conkle 1990, Lauria 1997, Mitton et al. 1997, Rehfeldt 1999, Patten and Brunsfeld 2002). Land managers have been concerned that Washoe pine would respond differently than ponderosa or Jeffrey pines to silviculture treatments (letter from W. B. Critchfield, PSW Research Geneticist, to Dick Lund, Forest Supervisor, 20 July 1972, on file Modoc N.F.). Our observations concur with Critchfield

TABLE 11.11

Fire history data derived from crossdated fire scars at six sites of less than 10 ha each in the Horse Creek watershed of the Warner Mountains

Site	*Number of Trees*	*Number of Scars*	*Median FRI*	*Mean FRI*	*Minimum FRI*	*Maximum FRI*
Horse A	8	21	7.5	13.3	1	51
Horse B	9	17	3.5	8.4	1	43
Horse C	10	12	37.0	33.0	8	55
Horse D	6	6	40.0	35.3	1	65
Horse F	14	20	5.5	7.1	1	20
Horse H	14	19	15.5	20.6	4	56

NOTE: FRI = fire return interval.

(1984) that the ecological response of Washoe pine to various land management activities such as logging and prescribed fire differs from ponderosa and Jeffrey pines. Washoe pine has distinctive adaptive characteristics that favor its survival in more severe climate and environments at higher elevations and slow growth rates (Rehfeldt 1999). Recent genetic research of Rehfeldt (1999) supports the distinction of Washoe pine as a robust and resilient taxon in the Warner Mountains. The ancestral lineage of Washoe pine continues to be of interest (Critchfield 1984, Lauria 1997, Rehfeldt 1999, Patten and Brunsfeldt 2002), which further emphasizes the need for land mangers to work with researchers to develop an adaptive fire management program to ensure survival of this unique species. The primary threats to Washoe pine are: (1) the potential destruction by stand-replacing fires where white fir succession has altered the fire regime, and (2) loss of habitat due to climate change (Brunsfeld 1999).

The only cross-dated fire scar record in the Warner Mountains is from this zone (Table 11.11). These data are from stands dominated by white fir with lodgepole pine and western white pine as associates. The fire scar record was developed from stumps of mostly white fir, lodgepole pine, and white pine. The record of fires extends from 1746 to 1957. Median fire-return intervals ranged from 3.5 to 40 years. Intra-ring position of fires scars was 90.6% ring boundary, 7.8% latewood, 1.6% in middle-earlywood (S. Smith unpublished data). This indicates that seasonality was mostly late summer to early fall and was similar to other upper-montane forests in northern California (Taylor 2000, Skinner 2003) (see Chapters 9 and 10).

In this zone, western juniper often occupies exposed, almost barren ridgelines. These are areas where fire rarely spreads due to very limited fuels. The barren nature of these ridgelines often limits the spread of fires from one watershed to the next. Many of the larger trees in these locations appear to be quite old, although no formal aging has been done.

Curl-leaf mountain-mahogany is often found on sites exposed to both the winter winds that remove snow and more intense summer insolation. Curl-leaf mountain-mahogany plants appear to initiate in small areas protected from wind and sun by rock outcrops. These sites are often surrounded by open, sparse stands of low shrubs and herbaceous plants. They can then survive for long periods as these areas appear to not carry fire well (e.g., the Blue fire in August 2001). Larger curl-leaf mountain-mahogany plants in one location of this type north of Fandango Pass have been aged at more than 400 years (unpublished data on file, Forest Service Pacific Southwest Research Station, Redding).

With the exception of the more-extensive stands to the east of Dismal Swamp, quaking aspen is generally found in small clones associated with increased soil moisture and surrounded by one of the other types of patches described above. The trees in larger quaking aspen clones affected by the Blue fire often survived even though the surrounding conifer stands sustained considerable mortality. The fire burning intensely in adjacent conifer stands appears to have dropped to the ground when entering the larger quaking aspen groves and burned through as a low-intensity surface fire. This change in fire behavior was likely due to higher moisture in the live and dead fuels, higher humidity, and the open nature of the crowns. None of the clones with this effect had significant encroachment of conifer regeneration.

Conifer stands in this zone likely have similar responses to fire as found in other bioregions, and the reader should reference Chapters 9, 10, and 12. The character of lodgepole pine stands varies from more open stands of larger trees and scattered individuals in the south Warner Mountains to more extensive and denser stands north of Highway 299, such as around Dismal Swamp, continuing through the Warner Mountains into Oregon. It is likely the fire regimes will vary with the nature of the stands, being more like the Cascade Range in the north (Chapter 10) and the Sierra Nevada in the south (Chapter 12).

Subalpine Zone

The subalpine zone is limited in this bioregion to the Warner Mountains with the majority of the zone found in the South

FIGURE 11.8. Subalpine zone. Photo looking south from Warren Peak toward Eagle Peak along the main crest of the Warner Mountains. All of the forests (foreground, middle ground, and background) shown in this photo are nearly 100% whitebark pine. The patches of whitebark pine are interspersed with patches of shrubs and herbaceous plants dominated mostly by snowfield sagebrush. Photo was taken during an August thunderstorm. (Photo by Carl Skinner.)

TABLE 11.12
Fire response types for important species in the subalpine zone

	Type of Fire Response			
Lifeform	Sprouting	Seeding	Individual	*Species*
Conifer	None	None	Resistant/killed	Whitebark pine
Shrub	Fire stimulated	None	Top-killed	Mountain snowberry, snowfield sagebrush
	+/−Fire stimulated	Fire stimulated	Top-killed	Gooseberry currant
Forb	Fire stimulated	None	Top-killed	Slender penstemon, prickly sandwort, Davidson's penstemon, phlox
Graminoids	Tillering	None	Top-killed	Western needlegrass, Wheeler's bluegrass, Ross' sedge

Warner Wilderness Area at elevations starting at about 2,253 m (7,390 ft) (Riegel et al. 1990). Whitebark pine forms nearly pure stands in this zone (Figs. 11.3 and 11.8). White fir can be either rare or common at the lower ecotonal edge; however, lodgepole pine, western white pine, and quaking aspen are relatively rare (Vale 1977, Riegel et al. 1990, Figura 1997). Common shrubs include snowfield sagebrush, Utah snowberry (*Symphoricarpos oreophilus* var. *utahensis)*, mountain gooseberry *(Ribes montigenum)*, and rabbitbush *(Ericameria bloomeri)*. Common herbaceous species include western needlegrass *(Achnatherum occidentale)*, pussypaws *(Calyptridium umbellatum)*, Wheeler's bluegrass *(Poa wheelerii)*, slender penstemon *(Penstemon gracilentus)*, prickly sandwort *(Arenaria aculeata)*, Davidson's penstemon, and spreading phlox.

FIRE RESPONSES OF IMPORTANT SPECIES

Whitebark pine is the only subalpine tree species in northeastern California. Chapter 12 contains a discussion of the fire response of whitebark pine. Snowfield sagebrush can resprout from root crowns or lower stem bases after being top-killed by fire (Winward 1985) (Table 11.12).

FIRE REGIME–PLANT COMMUNITY INTERACTIONS

The fire regimes are similar to that described for the subalpine zone of the Sierra Nevada and Cascade Range, with infrequent and small, low-intensity fires due to the lack and discontinuity of fuels. The largest and most contiguous stands are found in South Warner Wilderness Area from Mill

Creek to the southern slopes of Eagle Peak where relatively high-elevation ridges occur on a gentle west slope. Heavy sheep grazing in the late nineteenth century greatly reduced primarily perennial grasses and forbs in the subalpine sagebrush allowing for downslope expansion (Figura 1997). These lower-elevation stands, below approximately 2,470 m (8,102 ft), are characterized as relatively young (most trees established between 1915 and 1965) with a few older trees (200 to 300 years), yielding low stem density and basal area. Figura (1997) noted the lack of standing snags, large fallen logs, burnt coniferous wood, and downed wood in the lower-elevation stands, supporting his downslope migration hypothesis. High-elevation forests are old (400 to 800 years), self-perpetuating stands. Fire-scarred trees were found in the upper-elevation stands, though all trees cored by Figura (1997) were rotten, precluding dating historic fires. Lightning strikes are very common at these elevations yet fuel continuity and cool, moist conditions limit the ability of fire to spread beyond one to several multi-stemmed trees. We are unaware of evidence that suggests large fires have burned sizeable whitebark pine stands in the Warner Mountains. Small, old wind- and snow-shaped krummholz whitebark pine grow to the summit of Eagle Peak (3,016 m [9,892 ft]), which is the highest peak in the bioregion (Riegel et al. 1990). There is no alpine zone (land above treeline) in this bioregion (Riegel et al. 1990).

Non-Zonal Vegetation

Riparian areas, including wetlands, meadows, and riparian forests, are dispersed throughout the bioregion in all ecological zones due to both topographic and edaphic controls of surface and subsurface water. To our knowledge no fire ecology research has been done in these hydric vegetation types in this bioregion. Fire regimes in riparian areas probably differ from those in adjacent uplands but are interconnected through shared disturbance events. We speculate that presettlement fire return intervals were longer and were typically more spatially complex than adjacent uplands. When these sites did burn, fire intensity was moderate to high with mixed severity due to microtopograhic and surface–water table interactions. Livestock removal of fine fuels in the form of forage has altered these fire regimes since postsettlement.

One of the most obvious changes has been tree invasion—mostly lodgepole pine—into meadows. The reasons for invasion are complex, but the combination of reduced competition from herbaceous species and fine fuel removal due to livestock grazing coupled with fire exclusion has reduced the number and size of meadows in this bioregion. Because lodgepole pine is an evergreen conifer, it continues to transpire as long as the soil within the rooting zone is not frozen (Running 1980, Running and Reid 1980, Sparks et al. 2001). This increased, nearly year-long transpiring leaf area significantly reduces water to the meadow and is a net water loss to a watershed.

Common riparian hardwood trees and shrubs, comprised mostly of black cottonwood *(Populus balsamifera* ssp. *trichocarpa)*, willows, silver sagebrush, interior rose *(Rosa woodsii* var. *ultramontana)*, and scattered populations of mountain *(Alnus incana* ssp. *tenuifolia)* and white alder *(Alnus rhombifolia)*, and dwarf *(Betula nana)* and water birch *(Betula occidentalis)*, can sprout from stumps, stems, and root crowns following low- and, sometimes, moderate-intensity fires (Dwire and Kauffman 2003).

Quaking aspen, considered a facultative or facultative upland wetland species occurs in different settings in the Northeastern Plateaus bioregion, depending on the zone and location. In the sagebrush steppe zone, quaking aspen trees are somewhat rare; if they do occur, they are confined to sites such as streams, springs, and talus rock "reservoirs" that have perennial moisture. In the lower- and mid-montane zones, quaking aspen stands are more common and are similarly associated with sites containing perennial water. The most extensive occurrences of quaking aspen in the Northeastern Plateaus bioregion are in the upper-montane zone, mainly in the Warner Mountains. In this zone, quaking aspen stands are found on sites with added moisture from streams, meadows, and springs, as well as sites with added moisture from snowdrifts. In the subalpine zone, quaking aspen occurrence declines with increasing elevation. In areas where snow accumulations can persist well into the growing season (typically June through July), nivial or snow-morphed stands occur (Riegel et al. 1990), which are very important to many wildlife species in this zone.

Quaking aspen are clonal. The clones can be extremely long lived, hundreds to perhaps thousands of years, although the aboveground stems *(ramets)* usually only live for 125 years or so. Continuous successful regeneration from root suckers is necessary for clone survival over time. Quaking aspen has very low shade tolerance. Healthy clones with a range of size and age classes are stimulated and regenerated by stand-replacing fire.

Quaking aspen stands in this bioregion occur in two broad "themes": "Meadow" quaking aspen and "Upland" quaking aspen. Meadow quaking aspen that grow on wetter sites on stream sides and meadow edges are often not susceptible to conifer succession. If the aboveground stems senesce and are not replaced by younger stems, the clones will often thin out or disappear and the sites will be occupied by forb communities, such as corn lily (*Veratrum californicum*) in the upper-montane zone, or even mountain big sagebrush if the sites are also dewatered. Upland quaking aspen grow in drier sites and are susceptible to conifer succession in the absence of fire or other disturbances that remove or kill the conifers. Quaking aspen in both environments have decreased in the last 150 years. Meadow quaking aspen have been more affected by livestock grazing than by changes in fire regimes in the twentieth century. Upland quaking aspen stands have been profoundly affected by changed fire regimes, and have largely become dominated by conifers. These stands appear to occupy but a small fraction of their former extent, judging from scattered

SIDEBAR 11.3 SAGE GROUSE

Sage grouse *(Centrocercus urophasianus)* are a sagebrush-obligate and depend on the fire-sensitive plant for both food and cover. During late summer and fall, sagebrush leaves make up the majority of their diet and constitute 100% of their diet during the winter (Crawford et al. 2004). During the early spring and throughout summer (depending on availability), forbs are an important part of their diet. Prior to the twentieth century, sage grouse were able to adapt to an ecosystem where fire was an important ecosystem process. However, fire regimes have drastically changed across much of the sagebrush biome since Euro-American settlement, resulting in large losses of habitat. Ironically, fire-return intervals have significantly lengthened in the mountain big sagebrush series and greatly shortened in the Wyoming big sagebrush series (Wright and Bailey 1982, Miller and Rose 1999, Miller and Tausch 2001). In northeast California, exotic annuals have replaced large areas of warmer, drier sagebrush communities, which are potentially important winter habitat. On cooler, more mesic sagebrush communities, western juniper woodlands are rapidly increasing, resulting in the decline of sagebrush and forbs. Both scenarios of altered fire regimes resulting in conifer and exotic weed encroachment have caused a significant loss in sage grouse habitat (Miller and Eddleman 2000, Crawford et al. 2004). A primary question related to sage grouse is "How do we manage fire"?

There is considerable debate on the pros and cons of using fire to enhance sage grouse habitat. The decision to use or suppress fire for the enhancement of sage grouse habitat must be made on a site-by-site basis. Five factors that determine the negative or positive outcomes of fire on sage grouse habitat are: (1) site potential, (2) site condition, (3) functional plant group(s) (4) what is limiting in the habitat, (5) pattern and size of the burn, and (6) plant community composition on a landscape basis. Because sage grouse have extensive home ranges, habitat limitations must be considered at large scales.

Fire can be a useful tool to enhance the availability of native perennial forbs and grasses among many of the mountain big sagebrush plant associations. This is primarily in communities where sagebrush cover exceeds 30%, good populations of native herbs are present, and exotic species are limited. Perennial forb abundance can increase two- to threefold following fire (Pyle and Crawford 1996, EOARC data). Fire also enhances nutrient quality of forbs, especially protein content (McDowell 2000), and lengthens the growing season. The affects of fire on beetles and ants (important chick food) are mixed. Pyle and Crawford (1996) reported that fire did not affect beetle populations, whereas Fischer et al. (1996) reported decreases in beetle populations following fire. Fire also has a negative impact on thatch ants (J. McIver, USFS PNW Research Station, La Grande, OR, personal communication). Burning in good condition mountain big sagebrush affected species composition but not abundance of caterpillars (EOARC data), an important food for sage grouse chicks (Mike Gregg, USDI Fish and Wildlife Service, personal communication). In Nevada, sage grouse have been reported to be attracted to burn areas during the summer (Klebenow and Beall 1977, Martin 1990). Small burns with adjacent stands of sagebrush have also been used as leks. Possibly the most important justification for the use of fire for the restoration or maintenance of sage grouse habitat is that of limiting the development of juniper woodlands across many of these shrub steppe communities.

The use of fire in arid sagebrush plant associations for improving sage grouse habitat is probably very limited. Burning in good condition Wyoming big sagebrush communities or low sagebrush/Sandberg bluegrass associations usually does not increase desirable forbs used as sage grouse food (Fischer et al. 1996, EOARC file). In addition, Fischer et al. (1996) reported a decrease in beetle populations, an important chick food (Pyle and Crawford 1996). Many Wyoming big sagebrush sites also have depleted understories and have a low resistance to invasion of exotic plant species. With little to gain and the potential to increase populations of exotic weeds, the use of fire as a tool to enhance sage grouse habitat in these drier sagebrush communities is limited and very risky.

living and dead remnants throughout the mid- and upper montane zones. Quaking aspen clones respond vigorously to stand-replacing fire by producing thousands of suckers per hectare. Often the extent of a degraded clone with a few surviving stems per hectare is not apparent until a stand-replacing fire event releases the clone and multitudes of quaking aspen suckers appear where before there was nothing but white fir or some other conifer.

Management Issues

Altered fire regimes have likely caused large-scale changes in plant composition and structure across many plant communities. These changes affect important ecological processes, including the water, nutrient, and energy cycles, in addition to wildlife habitat and forage production.

The reintroduction of fire has been proposed to restore some of these arid-land communities. However, there is considerable debate over the use of fire as a tool to restore these habitats. Much of the debate is due to a lack of research on fire ecology from this bioregion. Concerns related to the use of fire, including the rapid expansion of exotic weedy species further loss of habitat for sagebrush obligates, watershed, and liability, are fueled primarily by a lack of local information. However, the lengthening periods without fire in some plant communities in the quaking aspen and mountain big sagebrush series will likely lead to their loss.

Besides altered fire regimes, the primary issues in the Northeastern Plateaus bioregion include grazing effects, wildlife needs, and impacts of invasive plant species, especially cheat grass (see Chapter 22). However, due to the lack of investigation and interest, this region is one of the most poorly understood with regard to fire ecology. In the future, more attention and investigation need to be given to the fire ecology of the various brush and woodland communities that occur in this region.

Grazing

Both the presence and level of grazing have issues for many areas in the Northeastern Plateaus bioregion and have affected a wide variety of species, including Idaho fescue, bluebunch wheatgrass, Wyoming big sagebrush, antelope bitterbrush, rabbitbrush, white fir, and whitebark pine. In some instances, grazing maintains conditions suitable for native vegetation by keeping areas more open and preventing excessive accumulation of dead material. On the other hand, direct impacts to plants occur from foraging and trampling, and indirect impacts occur when both native and invasive plant species are promoted by livestock grazing, reducing competition for resources, primarily soil moisture.

Because grazing affects the vegetation composition and condition, the regime of fire is influenced by grazing regimes (Miller et al. 1994). Thus, subsequent fire regimes that affect native vegetation communities are affected by how grazing has been conducted. Prescribed burning is used to improve range conditions and such burning can have dramatic effects on sagebrush and grass species, as well as on junipers. Grazing timing and regimes, as well as prescribed burning conducted for range value, need to integrate the unique fire ecology of the various northeast plateau plant communities to minimize both impacts to native vegetation and alterations to future fire regimes.

Wildlife

Many key plant species in the shrub and woodland communities of the northeast plateaus are greatly affected by fire. These species, such as antelope bitterbrush, rabbitbrush, and junipers, are also significant habitat and food for wildlife species. Mule deer, pronghorn *(Antilocapra americana)*, hares and rabbits, and sage grouse and other ground-nesting birds are dependent on the cover, shelter, and food that the scattered native grass, brush, and woodland vegetation provide.

Greatly altered fire regimes and the spread of exotic plant species have affected this bioregion's wildlife. As in other areas of the state, the use of prescribed fire has become a topic of interest among land managers and agencies responsible for the conservation of wildlife. One example of this debate is the sage grouse, a species that in recent years has shown a decline in numbers sufficient to raise concerns about its health and viability. Whether prescribed fire has a conservation role for the sage grouse or any other wildlife species is still unknown (see Sidebar 11.3).

Summary

This bioregion has a rich history of fire throughout the vast arid basins, uplands, and forested mountain ranges interspersed with fresh and alkaline wetlands. Both lightning and Native American burning patterns abruptly changed when fuel continuity was disrupted, and non-native invasive species were introduced, with the arrival of large numbers of domestic livestock in the last two decades of the 1800s. Fire suppression and agriculture have further altered historic fire regimes. Changes to fire regimes have caused changes in plant community composition and structure, and wildlife habitat in many plant communities. Although fire once limited woody plants, juniper now takes advantage of longer fire return intervals; its expanding range into the sagebrush steppe has also affected fire regimes. Fire has also become less frequent throughout the montane zones with the mid-montane more changed than the lower montane, and upper montane the least changed of the three. Fire continues to be an important ecological process throughout the Northeastern Plateaus bioregion but its role has greatly changed.

References

Adam, D. P., A. M. Sarna-Wojcicki, H. J. Rieck, J. P. Bradbury, W. E. Dean, and R. M. Forester. 1989. Tulelake California: the last 3 million years. Palaeogeography, Palaeoclimatology, Palaeoecology 72:89–103.

Adams, A. W. "Bud." 1975. A brief history of juniper and shrub populations in southern Oregon. Oregon State Wildlife Commission, Wildlife Research Report No. 6, Corvallis, OR.

Agee, J. K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Washington, D.C.

Alderfer, J. M. 1977. A taxonomic study of bitterbrush [*Purshia tridentata* (Pursh). DC.] in Oregon. M. S. Thesis. Oregon State University, Corvallis

Antevs, E. 1938. Rainfall and tree growth in the Great Basin. Carnegie Institution of Washington, Publication 469, American Geographers Society, Special. Publication 21. New York, N.Y.

Arno, S. F. 1976. The historical role of fire on the Bitterroot National Forest. USDA Forest Service, Intermountain Forest and Range Experiment Station, General Technical Report, INT-187, Ogden, UT.

Barbour, M. G., and R. A. Minnich. 2000. California upland forests and woodlands. P. 161–202 in M. G. Barbour and W. D. Billings (eds.), North America terrestrial vegetation, 2nd. ed. Cambridge University Press, New York.

Bates, J., R. F. Miller, and T. S. Svejcar. 2000. Understory dynamics in cut and uncut western jumper woodlands. Journal of Range Management 53:119–126.

Bedwell, S. F. 1973. Fort Rock Basin prehistory and environment. University of Oregon Books, Eugene, OR.

Billings, W. D. 1952. The environmental complex in relation to plant growth and distribution. Quarterly Review of Biology 27:251–265.

Blaisdell, J. P. 1950. Effects of controlled burning on bitterbrush on the Upper Snake River Plains. USDA Forest Service, Intermountain Forest and Range Experiment Station, INT Research Paper 20, Ogden, UT.

Blaisdell, J. P. 1953. Ecological effects of planned burning of sagebrush-grass range on the upper Snake River Plains. U.S. Department of Agriculture, Technical Bulletin No. 1075.

Blaisdell, J. P., and W. F. Mueggler. 1956. Sprouting of bitterbrush *(Purshia tridentata)* following burning or top removal. Ecology 37:365–370.

Bolsinger, C. L. 1989. California's western juniper and pinyon-juniper woodlands: area, stand characteristics, wood volume, and fenceposts. USDA Forest Service Pacific Northwest Research Station, Bulletin PNW-RB-166, Berkeley, CA.

Bork, J. L. 1985. Fire history in three vegetation types on the east side of the Oregon Cascades. Ph.D. Thesis, Oregon State University, Corvallis, OR.

Bradley, A. F., N. V. Noste, and W. C. Fischer. 1992. Fire ecology of forests and woodlands in Utah. USDA Forest Service Research, Intermountain Research Station, General Technical Report INT-287, Ogden, UT.

Britton, C. M., R. G. Clark, and F. A. Sneva. 1981. Will your sagebrush range burn? Rangelands 3:207–208.

Brown, J. K. 1982. Fuel and fire behavior predication in big sagebrush. USDA Forest Service Research, Intermountain Forest and Range Experiment Station, Research Paper INT-290, Ogden UT.

Brown, J. K. 2000. Introduction and fire regimes. P. 1–7 in J. K. Brown and J. K. Smith (eds.), Wildland fire in ecosystems: effects of fire on flora. USDA Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-42-Volume 2, Fort Collins, CO.

Brown, V. A., and W. S. Brown. 1991. The Sugar Hill fire and reforestation, and lookout tower. Journal of the Modoc County Historical Society (Warner Mountains Issue) 13:173–183.

Bunting, S. C. 1984a. Fire in sagebrush grass ecosystems: successional changes. In K. Sanders and J. Durham (eds.), Rangeland fire effects. USDI Bureau of Land Management Idaho State Office, Boise.

Bunting, S. 1984b. Prescribed burning of live standing western juniper and post-burning succession. P. 69–73 in T. E. Bedell (comp.), Oregon State University Extension Service Proceedings, Western juniper short course. October 15–16, Bend, OR.

Bunting, S. C., B. M. Kilgore, and C. L. Busbey. 1987. Guidelines for prescribed burning sagebrush-grass rangelands in the northern Great Basin. USDA Forest Service General Technical Report INT-231.

Burkhardt, J. W., and E. W. Tisdale. 1969. Nature and successional status of western juniper vegetation in Idaho. Journal of Range Management 22:264–270.

Burkhardt, J. W., and E. W. Tisdale. 1976. Causes of juniper invasion in southwestern Idaho. Ecology 76:472–484.

Busse, M. D. 2000a. Ecological significance of nitrogen fixation by actinorhizal shrubs in interior forests of California and Oregon. P. 23–41 in R. F. Powers, D. L. Hauxwell, and G. M. Nakamura (tech. coords.), Proceedings of the California Forest Soils Council conference on forest soil biology and forest management. USDA Forest Service General Technical Report GTR-PSW-178, Albany, CA.

Busse, M. D. 2000b. Suitability and use of the ^{15}N-istope dilution method to estimate nitrogen fixation by actinorhizal shrubs. Forest Ecology and Management 136:85–95.

Busse, M. D., P. H. Cochran, and J. W. Barrett. 1996. Changes in ponderosa pine site productivity following removal of understory vegetation. Soil Science Society of America Journal 60:1614–1621.

Busse, M. D., S. A. Simon, and G. M. Riegel. 2000. Tree-growth and understory responses to low-severity prescribed burning in thinned *Pinus ponderosa* forests of Central Oregon. Forest Science 46:258–268.

Chadwick, H. W., and P. D. Dalke. 1965. Plant succession on dune sands in Fremont County, Idaho. Ecology 46:765–780.

Clark, R. G., C. M. Britton, and R. A. Sneva. 1982. Mortality of bitterbrush after burning and clipping in eastern Oregon. Journal of Range Management 35:711–714.

Clements C. D., and J. A. Young. 1997. A viewpoint: Rangeland health and mule deer habitat. Journal of Range Management 50:129–138.

Clements, C. D., and J. A. Young. 2001. Antelope bitterbrush seed production and stand age. Journal of Range Management 54:269–273.

Cochran, P. H. 1998. Examples of mortality and reduced annual increments of white fir induced by drought, insects, and diseases at different stand densities. USDA Forest Service, Pacific Northwest Research Station, Research Note PNW-RN-525, Portland, OR.

Cochran, P. H., and W. E. Hopkins. 1991. Does fire exclusion increase productivity of ponderosa pine? P. 224–228 in A. E. Harvey and L. F. Neuenschwander (comps.), Proceedings: management and productivity of western montane forest soils. USDA Forest Service Research, Intermountain Research Station, General Technical Report GTR-INT-280, Ogden, UT.

Conard, S. G., and S. R. Radosevich. 1981. Photosynthesis, xylem pressure potential, and leaf conductance of three montane chaparral species in California. Forest Science 27:627–639.

Conard, S. G., and S. R. Radosevich. 1982. Growth responses of white fir to decreased shading and root competition by montane chaparral shrubs. Forest Science 28:309–320.

Conard, S. G., A. E. Jaramillo, K. Cromack, Jr., and S. Rose. 1985. The role of *Ceanothus* in western forest ecosystems. USDA Forest Service, Pacific Northwest Forest and Range Experiment Station, General Technical Report PNW-182, Portland, OR.

Conkle, M. T., and W. B. Critchfield. 1988. Genetic variation and hybridization of Ponderosa Pine. P. 27–43 in D. M. Baumgartner and J. E. Lotan. Ponderosa pine: the species and its management, symposium proceedings, Washington State University, Pullman.

Conrad, C. E., and C. E. Poulton. 1966. Effect of a wildfire on Idaho fescue and bluebunch wheatgrass. Journal of Range Management 19:148–141.

Cook, J. G., T. J. Hershey, and L. L. Irwin. 1994. Vegetative response to burning on Wyoming, mountain-shrub big game ranges. Journal of Range. Management 47:296–302.

Countryman, C. M., and D. R. Cornelius. 1957. Some effects of fire on perennial range type. Journal of Range Management 10:39–41.

Coville, F. 1898. Forest growth and sheep grazing in the Cascade Mountains of Oregon. U.S. Department of Agriculture, Division of Forestry Bulletin 15, Washington, DC.

Crane, M. F. 1982. Fire ecology of Rocky Mountain Region forest habitat types. Final Report Contract No. 43-83X9-1-884. Missoula, MT: USDA Forest Service, Region 1. 272 p. On file with: USDA Forest Service, Intermountain Research Station, Fire Sciences Laboratory, Missoula, MT.

Crawford, J. C., R. A. Olson, N. E. West, J. C. Mosley, M. A. Schroeder, T. D. Whitson, R. F. Miller, M. A. Gregg, and C. S. Boyd. 2004. Ecology and management of sage-grouse and sage-grouse habitat. Journal of Range Management 57:2–19.

Critchfield, W. B. 1984. Crossability and relationships of Washoe pine. Madroño 31:144–170.

Critchfield, W. B., and G. L. Allenbaugh. 1969. The distribution of the Pinaceae in and near northern Nevada. Madroño 19:12–26.

Cronquist, A. 1994. Asterales. P. 17 in A. Cronquist, A. H. Holmgren, N. H. Holmgren, J. L. Reveal, and P. K. Holmgren. Intermountain Flora: Vascular Plants of the Intermountain West, U.S.A, Volume 5, New York Botanical Gardens, Bronx, NY.

Daly, C., R. P. Neilson, and D. L. Phillips. 1994. A statistical-topographic model for mapping climatological precipitation over mountainous terrain. Journal of Applied Meteorology 33:140–158.

Davis, J. N., and J. D. Brotherson. 1991. Ecological characteristics of curl-leaf mountain-mahogany (*Cercocarpus ledifolius* Nutt.) communities in Utah and implications for management. Great Basin Naturalist 51:153–166.

Dealy, J. E. 1975. Ecology of curl-leaf mountain-mahogany *(Cercocarpus ledifolius)* in eastern Oregon and adjacent area. Ph.D. Thesis, Oregon State University, Corvallis.

Delwiche, C. C., P. J. Zinke, and C. M. Johnson. 1965. Nitrogen fixation by *Ceanothus*. Plant Physiology 40:1045–1047.

Dobkin, D. S., and J. D. Sauder. 2004. Shrubsteppe landscapes in jeopardy. Distributions, abundances, and thee uncertain future of birds and small mammals in the Intermountain West. High Desert Ecological Research Institute, Bend, OR.

Driscoll, R. S. 1963. Sprouting bitterbrush in central Oregon. Ecology 44:820–821.

Dwire, K. A., and J. B. Kauffman. 2003. Fire and riparian ecosystems in landscapes of the western USA. Forest Ecology and Management 178:61–74.

Evanko, A. B. 1951. Response of *Wyethia* to 2,4-D. USDA Forest Service North. Rocky Mt. Forest Range Experiment Station Research Note No. 98, Fort Collins, CO.

Ferrel, G. T. 1978. Moisture stress threshold of susceptibility to fir engraver beetles in pole-sized white firs. Forest Science 24:85–92.

Ferrel, G. T. 1980. Growth of white firs defoliated by Modoc budworm in northeastern California. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Research Paper PSW-154, Berkeley, CA.

Figura, P. J. 1997. Structure and dynamics of whitebark pine forests in the South Warner Wilderness Area. M. S. Thesis, Humboldt State University, Arcata, CA.

Fischer, R. A., K. P. Reese, and J. W. Connelly. 1996. An investigation on fire effects within xeric sage grouse brood habitat. Journal of Range Management 49:194–198.

Frost, C. C. 1998. Presettlement fire frequency regimes of the United States: A first approximation. P. 70–81 in T. L. Pruden and L. A. Brennan (eds.), Fire in ecosystem management: shifting the paradigm from suppression to prescription. Tall Timbers Fire Ecology Conference Proceedings, No. 20 Tall Timbers Research Station, Tallahassee, FL.

Gay, D. 1998. A test of the southcentral Oregon mule deer habitat suitability index. M. S. Thesis, University of Idaho, Moscow.

Gillespie, I. G., and M. E. Loik. 2004. Pulse events in Great Basin Desert shrublands: Responses of *Artemisia tridentata* and *Purshia tridentata* seedlings to summer rainfall pulses. Journal of Arid Environments 59:41–57.

Goheen, A. 1998. Buck Creek Drainage fire analysis: the ecological consequences of aggressive fire suppression. Technical Fire Management 11, Washington Institute, Colorado State University, Fort Collins.

Goodwin, D. L. 1956. Autecological studies of *Artemisia tridentata* Nutt. Ph.D. Dissertation, State College of Washington, Pullman.

Graumlich, L. J. 1987. Precipitation variation in the Pacific Northwest (1675–1975) as reconstructed from tree rings. Annals of the Association of American Geographers. 77:19–29.

Griffin, J. R., and W. B. Critchfield. 1972. The distribution of forest trees in California. Research Paper, PSW-82, USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, CA.

Gruell, G. E., S. Bunting, and L. Neuenschwander. 1984. Influence of fire on curl-leaf mountain-mahogany in the Intermountain West. P. 58–72 in J. K. Brown and J. Lotan (eds.), Proceedings: fire's effects on wildlife habitat. USDA Forest Service Research, Intermountain Research Station, General Technical Report GTR-INT-186, Ogden, UT.

Gruell, G. E., 1999. Historical and modern roles of fire in pinyon-juniper. P. 24–28 in S. B. Monsen and R. Stevens (comps.), Proceedings: ecology and management of pinyon-juniper communities within the interior West, Sept. 15–18, 1997, Provo, UT. USDA Forest Service, Rocky Mountain Research Station Proceedings RMRS-P-9, Fort Collins.

Guenther, G. E., C. L. Wamboldt, and M. R. Frisina. 1993. Characteristics of bitterbrush habitats that influence canopy cover and mule deer browsing. Journal of Environmental Management 36:175–181.

Hakala, K. J., and D. P. Adam. 2004. Late Pleistocene vegetation and climate in the southern Cascade Range and the Modoc Plateau region. Journal of Paleolimnology 31:189–215.

Haller, J. R. 1961. Some recent observations on ponderosa, Jeffrey and Washoe pines in northeastern California. Madroño 16: 126–132.

Harniss, R. O., and W. T. McDonough. 1974. Yearly variation in germination in the subspecies of big sagebrush. Journal of Range Management 29:167–168.

Harniss, R. O., and R. B. Murray. 1973. 30 years of vegetal change following burning of sagebrush-grass range. Journal of Range Management 29:322–325.

Hassan, M. A., and N. E. West. 1986. Dynamics of soil seed pools in burned and unburned sagebrush semi-deserts. Ecology 67:269–272.

Heisey, R. M., and C. C. Delwiche. 1983. A survey of California plants for water-extractable and volatile inhibitors. Botanical Gazette 144:382–390.

Heyerdahl, E. K., R. F. Miller, and R. A. Parsons. (2006). History of fire and Douglas-fir establishment in a savanna and sagebrush-grassland mosaic, southwestern Montana, USA. Forest Ecology and Management 230:107–118.

Hinckley, T. M., R. O. Teskey, R. H. Waring, and Y. Morikawa. 1982. The water relations of true firs. P. 85–94 in C. D. Oliver and R. M. Kenady (eds.), Proceedings of the biology and management of true fir in the Pacific Northwest symposium. University of Washington, College of Forest Resources, Contribution, No. 45, Seattle, WA.

Holmgren, R. C. 1956. Competition between annuals and young bitterbrush *(Purshia tridentata)* in Idaho. Ecology 37:370–377.

Hopkins, W. E. 1979. Plant associations of the Fremont National Forest. U.S. Department of Agriculture Forest Service, Pacific Northwest Region, R6-ECOL-79-004, Portland, OR.

Hormay, A. L. 1943. Bitterbrush in California. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Research Note 34, Berkeley, CA.

Houston, D. B. 1973. Wildfires in northern Yellowstone National Park. Ecology 54:1109–1117.

Hurley, J., and T. Ratcliff. 1986. Resprouting success in antelope bitterbrush following a prescribed burn. Unpublished report on file, USDA Forest Service, Plumas National Forest, Quincy, CA.

Jabbes, M. 2000. Hybridization and its evolutionary consequences in *Purshia Purshia* and *Cowania*. Ph.D. dissertation. University of Idaho, Moscow.

Jenkinson, J. L. 1990. *Pinus jeffreyi* Grev. & Balf. Jeffrey Pine. P. 359–369 in R. M. Burns and B. H. Honkala (tech. coords.), Silvics of North America: Volume I. Conifers. USDA Forest Service, Agricultural Handbook 654. Washington, DC.

Johnson, C. G., Jr. 1998. Vegetation responses after wildfire in national forests of northeastern Oregon. USDA Forest Service, Pacific Northwest Region, R6-NR-ECOL-TP-06-98.

Johnson, D. D., and R. F. Miller. (in press). Structure and development of expanding western juniper as influenced by two topographic variables. Forest Ecology and Management. (in press).

Johnson, D. W. 1995. Soil properties beneath *Ceanothus* and pine stands in the eastern Sierra Nevada. Soil Science Society of America Journal 59:918–924.

Johnson, J. R., and G. F. Payne. 1968. Sagebrush reinvasion as affected by some environmental influences. Journal of Range Management 21:209–213.

Kennedy, P. B., and S. B. Doten. 1901. A preliminary report on the summer ranges of western Nevada sheep. Nevada State University, Agricultural Experiment Station, Bulletin, No. 51, Reno.

Kelly, I. T. 1932. Ethnography of the Surprise Valley Paiute. University of California, Publications in American Archaeology and Ethnology 31:67–209.

Klebenow, D. A., and R. C. Beall. 1977. Fire impacts on birds and mammals on Great Basin rangelands. In Proceedings, Joint Intermountain Rocky Mountain Fire Research Council, Casper, WY.

Klebenow, D. A., and A. Bruner 1976. Determining factors necessary for prescribed burning. P. 69–47 in Use of prescribed burning in western woodland and range ecosystems. Utah State University, Logan, UT.

Kruckeberg, A. R. 1977. Manzanita *(Arctostaphylos)* hybrids in the Pacific Northwest: effects of human and natural disturbance. Systematic Botany 2: 233–250.

Kuntz, D. E. 1982. Plant response following spring burning in an *Artemisia tridentata* subsp. *vaseyana*/*Festuca idahoensis* habitat type, M.S. Thesis, University of Idaho, Moscow.

Laacke, R. J. 1990. *Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr. White fir. P. 36–46 in R. M. Burns and B. H. Honkala (tech. coords.), Silvics of North America: Volume I. Conifers. USDA Forest Service, Agricultural Handbook 654. Washington, DC.

Lanini, W. T., and S. R. Radosevich. 1986. Response of three conifer species to site preparation and shrub control. Forest Science 32:61–77.

Laudenslayer, W. F., Jr., H. H. Darr, and S. Smith. 1989. Historical effects of forest management practices on eastside pine communities in northeastern California. P. 26–34 in A. Tecle, W. W. Covington, R. H. Hamre (tech. coords.), Multiresource management of ponderosa pine forests: Proceedings of the symposium, November 14–16, 1989, Flagstaff, AZ. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report, GTR-RM-185, Fort Collins, CO.

Lauria, F. 1997. The taxonomic status of *Pinus washoensis* H.Mason & Stockw. (Pinaceae). Annalen des Naturhistorischen Museums in Wien 99:655–671.

Leiberg, J. B. 1902. Forest conditions in the northern Sierra Nevada, California. U.S.Geological Survey Professional Paper No. 8, U.S. Government Printing Office, Washington DC.

Little, E. L., Jr. 1971. Conifers and important hardwoods. Vol. 1. Atlas of the United States trees. USDA Forest Service, Miscellaneous Publication No. 1146, Washington, DC.

Lyon, L. J., and P. F. Stickney. 1976. Early vegetal succession following large northern Rocky mountain wildfires. In Proceedings Montana Tall Timbers Fire Ecology Conference and Fire and Land Management Symposium. No. 14:355–375. Tall Timbers Research Station, Tallahassee, FL.

MacDonald, G. A., and T. E. Gay, Jr. 1966. Geology of the southern Cascade Range, Modoc Plateau, and Great Basin areas in northeastern California. In Mineral resources of California. California Division of Mines and Geology Bulletin 191:43–48.

Major, J. 1988. California climate in relation to vegetation. P. 11–74 in M. G. Barbour and J. Major (eds.), Terrestrial vegetation of California. California Native Plant Society Special Publication Number 9, new expanded edition.

Martin, R. C. 1990. Sage grouse responses to wildfire in spring and summer habitats. M.S. Thesis. University of Idaho, Moscow.

Martin, R. E. 1982. Fire history and its role in succession. P. 92–99. In Means, J.E. (ed.), Forest succession and stand development

in the Northwest, Forest Research Lab, Oregon State University, Corvallis.

Martin, R. E. 1983. Antelope bitterbrush seeding establishment following prescribed fire in the pumice zone of the southern Cascade Mountains. P. 92–99. In A. R. Tiedemann, and K. L. Johnson (comps.), Proceedings-research and management of bitterbrush and cliffrose in western North America, USDA Forest Service, Intermountain Forest and Range Experiment Station, General Technical Report, INT-152, Ogden, UT.

Martin, R. E., and C. H. Driver. 1983. Factors affecting antelope bitterbrush reestablishment following fire. P. 266–279. In A.R. Tiedemann, and K. L. Johnson (eds.), Proceedings-research and management of bitterbrush and cliffrose in western North America, USDA Forest Service, Intermountain Forest and Range Experiment Station, General Technical Report INT-GTR-152, Ogden, UT.

Martin, R. E., and A. H. Johnson. 1979. Fire management of Lava Beds National Monument. P. 1209–1217 in R. E. Linn (ed.), Proceedings of the First Conference of Science and Research in the National Parks. USDI National Parks Service Transactions Proceedings Series No. 5.

McArthur, E. D., H. C. Stutz, and S. C. Sanderson. 1983. Taxonomy, distribution, and cytogenetics of *Purshia, Cowania,* and *Fallugia* (Rosoideae, Rosaceae). P. 4–24. In A.R. Tiedemann, and K.L. Johnson (eds.), Proceedings-research and management of bitterbrush and cliffrose in western North America, USDA Forest Service, Intermountain Forest and Range Experiment Station, General Technical Report INT-GTR-152, Ogden, UT.

McDonald, P. M. 1983a. Climate, history, and vegetation of the eastside pine type in California. P. 1–16 in T. F. Robson and R. B. Standiford (eds.), Management of the eastside pine type northeastern California: proceedings of a symposium. Northern California Society of American Foresters, S.A.F. 83-06, Arcata, CA.

McDonald, P. M. 1983b. Weeds in conifer plantations of northeastern California . . . management implications. P. 70–78 in T. F. Robson and R. B. Standiford (eds.), Management of the eastside pine type northeastern California: proceedings of a symposium. Northern California Society of American Foresters, S.A.F. 83-06, Arcata, CA.

McDowell, M.K.D. 2000. The effects of burning in mountain big sagebrush on key sage grouse habitat characteristics in southeastern Oregon. M.S. Thesis, Oregon State University, Corvallis.

Mehringer, P. J., Jr., and P. E. Wigand. 1987. Western juniper in the Holocene. P. 13–16 in R. L. Everett (ed.), Proceedings: Pinyon Juniper Conference. USDA Forest Service Research, Intermountain Research Station, General Technical Report GTR-INT-215.

Mehringer, P. J., and P. E. Wigand. 1990. Comparison of late Holocene environments from woodrat middens and pollen. P. 294–325 in J. L. Betancourt, T. R. Van Devender, and P. S. Martin (eds.), Packrat middens: the last 40,000 years of biotic change. University of Arizona Press, Tucson.

Meyer, S. E., and S. B. Monsen. 1991. Habitat-correlated variation in mountain big sagebrush *(Artemisia tridentata* ssp. *vaseyana)* seed germination patterns. Ecology 72:739–732.

Miller, M. 2000. Fire autecology. In J. K. Brown and J. K. Smith (eds.), Wildland fire in ecosystems: effects of fire on flora. General Technical Report RMRS-GTR-42-vol-2, USDA Forest Service, Rocky Mountain Research Station, Ogden, UT.

Miller, R. F. 2001. Managing western juniper of wildlife. Woodland Fish and Wildlife. Pub. No. MISC0286.

Miller, R. F., J. D. Bates, T. J. Svejcar, F. B. Pierson, and L. E. Eddleman. 2005. Biology, ecology, and management of western juniper *(Juniperus occidentalis)*. Oregon State University Agricultural Experiment Station Technical Bulletin 152.

Miller, R. F., P. Doescher, and T. Purrington. 1991. Dry-wet cycles and sagebrush in the Great Basin. The Grazier 30:5–10, Oregon State University, Department of Rangeland Resources, Corvallis.

Miller, R. F., and L. E. Eddleman. 2000. Spatial and temporal changes of sage grouse habitat in the sagebrush biome. Oregon State University Agricultural Experiment Station, Technical Bulletin 151, Corvallis.

Miller, R. F., E. K. Heyerdahl, and K. Hopkins. 2003. Fire regimes, pre- and post-settlement vegetation, and the modern expansion of western juniper at Lava Beds National Monument, California. Final Report to the USDI Lava Beds National Monument.

Miller, R. F., and J. A. Rose. 1995. Historic expansion of *Juniperus occidentalis* (western juniper) in southeastern Oregon. Great Basin Naturalist 55:37–45.

Miller, R. F., and J. A. Rose. 1999. Fire history and western juniper encroachment in sagebrush steppe. Journal of Range Management 52:520–559.

Miller, R. F., T. Svejcar, and J. A. Rose. 2000. Western juniper succession in shrub steppe: impacts on community composition and structure. Journal of Range Management 53:574–585.

Miller, R. F., T. Svejcar, and N. E. West. 1994. Implications of livestock grazing in the intermountain sagebrush region: plant composition. P. 101–146 in M. Vavra, W. A. Laycock, and R. D. Pieper (eds.), Ecological implications of livestock herbivory in the West. Western Society for Range Management. Denver, CO.

Miller, R. F., and R. J. Tausch. 2001. The role of fire in pinyon and juniper woodlands: a descriptive analysis. Tall Timbers Research Station. Miscellaneous Publication 11:15–30, Tallahassee, FL.

Miller, R. F., and P. Wigand. 1994. Holocene changes in semiarid pinyon-juniper woodlands: response to climate, fire and human activities in the U.S. Great Basin. BioScience 44:465–474.

Minckley, T. A. 2003. Holocene environmental history of the northwestern Great Basin and the analysis of modern pollen analogues in western North America. Ph.D. Dissertation, University of Oregon, Eugene.

Minore, D. 1979. Comparative autecological characteristics of northwestern tree species—a literature review. USDA Forest Service, General Technical Report PNW-87, Pacific Northwest Forest and Range Experiment Station, Portland, OR.

Mitton, J. B., R. G. Latta, and G. E. Rehfeldt. 1997. The pattern of inbreeding in Washoe pine and survival of inbred progeny under optimal environmental conditions. Silvae Genetica 46:215–219.

Mueggler, W. F. 1956. Is sagebrush seed residual in the soil of burns or is it wind borne? USDA Forest Service INT-RN-35.

Niebling, C. R., and M. T. Conkle. 1990. Diversity of Washoe pine and comparisons with allozymes of ponderosa pine races. Canadian Journal of Forest Research 20: 298–308.

Nord, E. C. 1959. Bitterbrush Ecology—Some recent findings. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, PSW Res. Note 148, Berkeley, CA.

Nord, E. C. 1965. Autecology of bitterbrush in California. Ecological Monographs 35:307–334.

Noste, N. V., and C. L. Bushey. 1987. Fire response of shrubs of dry forest habitat types in Montana and Idaho. USDA Forest Service, Intermountain Forest and Range Experiment Station, General Technical Report INT-GTR-239, Ogden, UT.

Olmstead, P. 1957. The Nevada-California-Oregon border triangle: a case study in sectional history. M.A. Thesis, University of Nevada, Reno.

Osborn, G., and K. Bevis. 2001. Glaciation in the Great Basin of the western United States. Quaternary Science Reviews 20:1377–1410.

Oswald, V. H., D. W. Showers, and M. A. Showers. 1995. A flora of Lassen National Park, California. California Native Plant Society, Sacramento, CA.

Owens, T. E. 1982. Postburn regrowth of shrubs related to canopy mortality. Northwest Science 56:34–40.

Parker, V. T., and M. P. Yoder-Williams. 1989. Reduction of survival and growth of young *Pinus jeffreyi* by an herbaceous perennial, *Wyethia mollis*. American Midland Naturalist 121:105–111.

Patten, A. M., and S. J. Brunsfeld. 2002. Evidence of a novel lineage within the *Ponderosae*. Madroño 49:189–192.

Pease, R. W. 1965. Modoc County: a geographic time continuum on the California volcanic tableland. University of California Publications in Geography Vol. 17. University of California Press, Berkeley.

Pechanec, J. F., G. Steward, and J. P. Blaisdell. 1954. Sagebrush burning—good and bad. U.S. Department of Agriculture, Farmer's Bulletin 1948.

Peek, J. M., F. D. Johnson, and N. N. Pence. 1978. Successional trends in a ponderosa pine/bitterbrush community related to grazing by livestock, wildlife, and to fire. Journal of Range Management 31:49–53.

Peek J. M., J. J. Korol, D. Gay, and T. Hershey. 2001. Overstory-understory biomass changes over a 35-year period in south-central Oregon Forest Ecology and Management 150:267–277.

Powers, R. F., and W. W. Oliver. 1990. *Libocedrus decurrens* Torr. Incense-cedar. P. 173–180 in R. M. Burns and B. H. Honkala (tech. coords.), Silvics of North America: Volume I. Conifers. USDA Forest Service, Agricultural Handbook 654. Washington, DC.

Pyle, W. H., and J. A. Crawford. 1996. Availability of foods of sage-grouse chicks following prescribed fire in sagebrush-bitterbrush. Journal of Range Management 49:320–324.

Radosevich, S. R. 1984. Interference between greenleaf manzanita *(Arctostaphylos patula)* and ponderosa pine *(Pinus ponderosa)*. P. 259–270 in M. L. Dureya and G. N. Brown (eds.), Seedling physiology and reforestation success. Martinus Nijhoff/Dr. W. Junk Publishers, Dordrecht, The Netherlands.

Rehfeldt, G. E. 1999. Systematics and genetic structure of Washoe pine: applications in conservation genetics. Silvae Genetica 48:167–173.

Reinkensmeyer, D. P. 2000. Habitat associations of bird communities in shrub-steppe and western juniper woodlands. M.S. Thesis, Oregon State University, Corvallis.

Riegel, G. M. 1982. Forest habitat types of the South Warner Mountains, Modoc County, northeastern California. M.S. Thesis, Humboldt State University, Arcata, CA.

Riegel, G. M., T. J. Svejcar, and M. D. Busse. 2002. Does the presence of *Wyethia mollis* limit growth of affect growth of *Pinus jeffreyi* seedlings? Western North American Naturalist 62: 141–150.

Riegel, G. M., D. A. Thornburgh, and J. O Sawyer. 1990. Forest habitat types of the South Warner Mountains, Modoc County, California. Madroño 37:88–112.

Romme, W. H., M. L. Floyd-Hanna, D. Hanna, and E. J. Bartlett. 1996. Landscape analysis for the South Central Highlands Section, southwestern Colorado and northwestern New Mexico, Chapter 5: Aspen forests. Unpublished draft; final report of the San Juan National Forest, Durango, CO.

Rosentreter, R. 2005. Sagebrush Identification, Ecology, and Palatability Relative to Sage-Grouse. P. 3–16 in N.L. Shaw, M. Pellant, and S.B. Monsen (comps.), Sagegrouse habitat restoration symposium proceedings; 2001 June 4–7; Boise, ID, USDA Forest Service, Rocky Mountain Research Station, Proceedings RMRS-P-38, Fort Collins, CO.

Rorig, M. L., and S. A. Ferguson. 1999. Characteristics of lightning and wildland fire ignition in the Pacific Northwest. Journal of Applied Meteorology 38:1565–1575.

Rorig, M. L., and S. A. Ferguson. 2002. The 2000 fire season: lightning-caused fires. Journal of Applied Meteorology 41: 786–791.

Roy, D. F. 1983. Natural regeneration. P. 87–102 in T. F. Robson and R. B. Standiford (eds.), Management of the eastside pine type in northeastern California, proceedings of a symposium. Northern California Society of American Foresters, S.A.F. 83-06, Arcata, CA.

Rundel, P. W., D. J. Parsons, and D. T. Gordon. 1988. Montane and subalpine vegetation of the Sierra Nevada and the Cascade Ranges. P. 559–599 in M. G. Barbour and J. Major (eds.), Terrestrial vegetation of California. California Native Plant Society Special Publication Number 9, new expanded edition.

Running, S. W. 1980. Environmental and physiological control of water flux through *Pinus contorta*. Canadian Journal of Forest Research 10: 82–91.

Running, S. W., and C. P. Reid. 1980. Soil temperature influences on root resistance of *Pinus contorta*. Plant Physiology 65:635–640.

Salwasser, H. 1979. The ecology and management of the Devil's Garden interstate deer herd and its range. University of California, Berkeley, Ph.D. dissertation.

Sawyer, J. O., and T. Keeler-Wolf. 1995. A manual of California vegetation. California Native Plant Society, Sacramento, CA.

Schaefer, R. J., D. J. Thayer, and T. S. Burton. 2003. Forty-one years of vegetation change on permanent transects in northeastern California: implications for wildlife. California Fish and Game 89:55–71.

Shainsky, L. J., and S. R. Radosevich. 1986. Growth and water relations of *Pinus ponderosa* seedlings in competitive regimes with *Arctostaphylos patula* seedlings. Journal of Applied Ecology 23:957–966.

Sherman R. L., and W. W. Chilcote. 1972. Spatial and chronological patterns of *Purshia tridentata* as influenced by *Pinus Ponderosa*. Ecology 53:294–298.

Shinn, D. A. 1980. Historical perspectives on range burning in the Inland Pacific Northwest. Journal of Range Management 33:415–422.

Simon, S. A. 1990. Fire effects from prescribed underburning in central Oregon ponderosa pine plant communities: first and second growing season after burning. Unpublished report on file, USDA Forest Service, Fremont National Forest, Lakeview, Oregon.

Skinner, C. N. 2003. Fire regimes of upper montane and subalpine glacial basins in the Klamath Mountains of northern California. Tall Timbers Research Station Miscellaneous Publication 13:145–151, Tallahassee, FL.

Skinner, C. N., and C. Chang. 1996. Fire regimes, past and present. P. 1041–1069 in Sierra Nevada Ecosystem Project: final report to Congress. Volume II: Assessments and scientific basis

for management options. Centers for Water and Wildland Resources, University of California, Davis, Water Resources Center Report No. 37.

Smith, R. H. 1967. Variations in the monoterpene composition of the wood resin of Jeffrey, Washoe, Coulter, and lodgepole pines. Forest Science 13:246–252.

Smith, R. H. 1971. Xylem monoterpenes of *Pinus ponderosa, P. washonesis,* and *P. Jeffreyi* in the the Warner Mountains of California. Madroño 21:26–32.

Smith, R. H. 2000. Xylem monoterpenes of pines: distribution, variation, genetics, function. USDA Forest Service, Pacific Southwest Research Station, General Technical Report PSW-GTR-177, Albany, CA.

Smith, S. 1994. Ecological guide to eastside pine plant associations, northeastern California: Modoc, Lassen, Klamath, Shasta-Trinity, Plumas, and Tahoe National Forests. USDA Forest Service, Pacific Southwest Region R5-ECOL-TP-004, San Francisco, CA.

Smith, S., and B. Davidson. 2003. User's manual: Terrestrial Ecological Unit Inventory (TEUI), land type associations, Modoc National Forest. USDA Forest Service, Pacific Southwest Region, R5-TP-015, Version 1.0.

Smith, T. G., and Maguire. 2004. Small-mammal relationships with down wood and antelope bitterbrush in ponderosa pine forests of Central Oregon. Forest Science 50:711–728.

Sparks, J. P., G. S. Campbell, and R. A. Black. 2001. Water content, hydraulic conductivity, and ice formation in winter stems of *Pinus contorta*: a TDR case study. Oecolgia 27:469–475.

Stephens, S. L. 2001. Fire history differences in adjacent Jeffrey pine and upper montane forests in the Sierra Nevada. International Journal of Wildland Fire 10:161–167.

Stephens, S. L., C. N. Skinner, and S. J. Gill. 2003. A dendrochonology based fire history of Jeffrey pine-mixed conifer forests in the Sierra San Pedro Martir, Mexico. Canadian Journal of Forest Research 33:1090–1101.

Stone, C. G. 1965. Modoc cypress, *Cupressus bakeri,* does occur in Modoc County. Aliso 6:77–87.

Tausch, R. J. 1999. Historic woodland development. P. 12–19. In R. Stevens, R. J. Tausch, R. Miller, and S. Goodrich (eds.), Ecology and management of pinyon and juniper communities with the interior west. USDA Forest Service, Rocky Mountain Research Station Proceedings RMRS-P-9, Fort Collins, CO.

Taylor, A. H. 2000. Fire regimes and forest changes along a montane forest gradient, Lassen Volcanic National Park, southern Cascade Mountains, USA. Journal of Biogeography 27:87–104.

Thompson, R. S. 1990. Late quaternary vegetation and climate in the Great Basin. P. 200–239 in J. L. Betancourt, T. VanDevender, and P. S. Martin (eds.), Packrat middens: the last 40,000 years of biotic change. University of Arizona Press, Tucson.

Thompson, R. S., L. Benson, and E. M. Hattori. 1986. A revised chronology for the last Pleistocene Lake cycle in the central Lahontan Basin. Quaternary Research 25:1–10.

USDA Forest Service. 1937. Range Plant Handbook. U.S. Government Printing Office, Washington, DC.

Vale, T. R. 1975. Invasion of big sagebrush *(Artemisia tridentata)* by white fir *(Abies concolor)* on the southeastern slopes of the Warner Mountains, California. Great Basin Naturalist 35:319–324.

Vale, T. R. 1977. Forest changes in the Warner Mountains, California. Annals of the Association of American Geographers 67:28–45.

van Wagtendonk, J. K. 1983. Prescribed fire effects on understory mortality. Fire and Forest Meteorology Conference 7:136–138.

van Wagtendonk, J. W., and D. Cayan. 2007. Temporal and spatial distribution of lightning strikes in California in relationship to large-scale weather patterns. Fire Ecology (in press).

Vander Wall, S. B. 1994. Seed fate pathways of antelope bitterbrush: dispersed by seed-caching yellow pine chipmunks. Ecology 75:1911–1926.

Vasek, F. C., and R. F. Thorne. 1988. Transmontane coniferous vegetation. P. 797–832 in M. G. Barbour and J. Major (eds.), Terrestrial vegetation of California. California Native Plant Society Special Publication Number 9, new expanded edition.

Vogl, R. J., W. P. Armstrong, D. L White, and K. L. Cole. 1988. The closed-cone pines and cypress. P. 295–358 in M. G. Barbour and J. Major (eds.), Terrestrial vegetation of California. California Native Plant Society Special Publication Number 9, new expanded edition.

Volland, L. A., and J. D. Dell. 1981. Fire effects on Pacific Northwest forest and range vegetation. USDA Forest Service, Pacific Northwest Region. Range Management and Aviation and Fire Management, R6 Rm 067 1981, Portland OR.

Vose, J. M., and A. S. White. 1991. Biomass response mechanisms of understory species the first year after prescribed burning in an Arizona ponderosa-pine community. Forest Ecology and Management 40:175–187.

Wagstaff, F. J. 1980. Impact of the 1975 Wallsburg Fire on antelope bitterbrush (*Purshia tridentata*). The Great Basin Naturalist 40:299–302.

Waichler, W. S., R. F. Miller, and P. S. Doescher. 2001. Community characteristics of old-growth western juniper woodlands. Journal of Range Management 54:518–527.

Wall, T. G., R. F. Miller, and T. Svejcar. 2001. Juniper encroachment into aspen in the Northwest Great Basin. Journal of Range Management 54:691–698.

Weaver, H. 1943. Fire as an ecological and silvicultural factor in the ponderosa pine region of the pacific slope. Journal of Forestry 41:7–14.

Webster, S. R., C. T. Youngberg, and A. G. Wollum. 1967. Fixation of nitrogen by bitterbrush (*Purshia tridentata* [Pursh] D.C.). Nature 216:392–393.

Weixelman, D. A., T. R. Bowyer, and V. Van Ballenberghe. 1998. Diet selection by Alaskan moose during winter: effects of fire and forest succession. In W. B. Ballard and A.R.J. Rodgers (eds.), Proceedings, 33rd North American moose conference and workshop; 4th international moose symposium, May 17–23, 1997, Fairbanks, AK. Alces 34:213–238.

West, N. E. 1968. Rodent-influenced establishment of ponderosa pine and bitterbrush seedlings in central Oregon. Ecology 49:1009–1011.

West, N. E. 1983. Western intermountain sagebrush steppe. P. 351–374 in N. E. West (ed.), Ecosystems of the world: temperate deserts and semi-deserts. Elsevier Scientific Publishing Company, New York.

West, N. E. 1984. Successional patterns and productivity potentials of pinyon-juniper ecosystems. P. 1301–1332 in Developing strategies for rangeland management: a report prepared by the committee on developing strategies for rangeland management. Westview Press, Boulder, CO.

Whisenant, S. G. 1990. Changing fire frequencies on Idaho's Snake River plains: ecological and management implications. P. 4–10 in Proceedings: symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management. USDA Forest Service General, Technical Report GTR-INT-276, Ogden, UT.

Whitlock, C., C. N. Skinner, P. J. Bartlein, T. Minckley, and J. A. Mohr. 2004. Comparison of charcoal and tree-ring records of recent fires in the eastern Klamath Mountains. Canadian Journal of Forest Research 34:2110–2121.

Wickman, B. E. 1978. A case study of a Douglas-fir tussock moth outbreak and stand conditions 10 years later. USDA Forest Service, Research Paper, PNW-244, Pacific Northwest Forest and Range Experiment Station, Portland, OR.

Wigand, P. E. 1987. Diamond Pond, Harney County, Oregon: vegetation history and water table in the eastern Oregon desert. Great Basin Naturalist 47:427–458.

Wigand, P. E., M. L. Hemphill, S. Sharpe, and S. Patra [Manna]. 1995. Eagle Lake Basin, northern California, paleoecological study: semi-arid woodland and montane forest dynamics during the late quaternary in the northern Great Basin and adjacent Sierras. University and Community College System of Nevada, Reno.

Williams, M. P. 1995. Inhibition of conifer regeneration by an herbaceous perennial, *Wyethia mollis,* in the eastern Sierra Nevada, California. Ph.D. Dissertation, University of Washington, Seattle.

Winward, A. H. 1985. Fire in the sagebrush-grass ecosystem—the ecological setting. P. 2–6 in K. Sanders and J. Durham (eds.), Rangeland fire effects: a symposium. USDI Bureau of Land Management, Idaho State Office, Boise.

Winward, A. H., and J. Alderfer-Finely. 1983. Taxonomic variations of bitterbrush (*Purshia tridentata*) in Oregon. P. 25–30. in A. R. Tiedemann, and K. L. Johnson (eds), Proceedings-research and management of bitterbrush and cliffrose in western North America, USDA Forest Service, Intermountain Forest and Range Experiment Station, General Technical Report INT-GTR-152, Ogden, UT.

Wright, H. A. 1971. Why squirreltail is more tolerant to burning than needle and thread. Journal of Range Management 24:277–284.

Wright, H.A. 1972. Shrub response to fire. P. 204–217. In Wildland shrubs- their biology and utilization, An International Symposium, Utah State University Logan, USDA Forest Service Intermountain Forest and Range Experiment Station, General Technical Report INT-1, Ogden, UT.

Wright, H. A., and A. W. Bailey. 1982. Fire ecology: United States and southern Canada. Wiley, New York.

Wright, H. A., and J. O. Klemmedson. 1965. Effects of fire on bunchgrasses of the sagebrush-grass region in southern Idaho. Ecology 46:680–688.

Wright, H. A., L. F. Neuenschwander, and C. M. Britton. 1979. The role and use of fire in sagebrush-grass and pinyon-juniper plant communities: a state of the art review. USDA Forest Service General Technical Report GTR-INT-58, Ogden, UT.

Yoder-Williams, M. P., and V. T. Parker. 1987. Allelopathic interference in the seedbed of *Pinus jeffreyi* in the Sierra Nevada, California. Canadian Journal of Forest Research 17:991–994.

Young, J. A., and C. D. Clements. 2002. Purshia: The Wild and Bitter Roses. University of Nevada Press, Reno.

Young, J. A., and R. A. Evans. 1978. Population dynamics after wildfires in sagebrush grasslands. Journal of Range Management 31:283–289.

Young, J. A., and R. A. Evans. 1979. Arrowleaf balsamroot and mules ears seed germination. Journal of Range Management 32:71–74.

Young, J. A., and R. A. Evans. 1981. Demography and fire history of a western juniper stand. Journal of Range Management 34:501–505.

Young, J. A., and R. A. Evans. 1989. Dispersal and germination of big sagebrush *(Artemisia tridentata)* seeds. Weed Science 37:201–206.

Young, J. A., R. A. Evans, and J. Major. 1988. Sagebrush steppe. P. 763–796 in M. G. Barbour and J. Major (eds.), Terrestrial vegetation of California. California Native Plant Society Special Publication Number 9, new expanded edition.

Young, R. P. 1983. Fire as a vegetation management tool in rangelands of the intermountain region. P. 18–31 in S. B. Monsen and N. Shaw (comps.), Managing intermountain rangelands—improvement of range and wildlife habitats: proceedings, September 15–17, 1981, Twin Falls, ID; June 22–24, 1982, Elko, NV. USDA Forest Service, Intermountain Forest and Range Experiment Station, General Technical Report.GTR-INT-157. Ogden, UT.

Young, R. P., and R. F. Miller. 1985. Response of *Sitanion hystrix* (Nutt.) J.G. to prescribed burning. American Midland Naturalist 113:182–187.

Ziegenhagen, L. 2003. Shrub reestablishment following fire in the mountain big sagebrush (*Artemisia tridentata* Natt. ssp. *vaseyana* [Rydb.] Beetle) alliance. Oregon State University, M. S. Thesis, Corvallis.

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