Section 2. Forestland

Chapter 2.1. Understanding and Managing the Dry Conifer Forests of Northeastern California

W. Keith Moser[1]

Introduction

The volcanic soils of the Southern Cascades and the Modoc Plateau support a diverse assemblage of conifer species. On the western slopes of the Southern Cascades, Douglas-fir (*Pseudotsuga menziesii*), ponderosa (*Pinus ponderosa*) and lodgepole (*Pinus contorta*) pines, and white (*Abies concolor*) and Shasta red firs (*Abies magnifica*) are common, whereas ponderosa and Jeffrey (*Pinus jeffreyi*) pines inhabit the drier east slopes. Eastern slopes and valleys commonly contain big sagebrush (*Artemisia* species) and western juniper (*Juniperus occidentalis*) habitats. The Great Basin, with its high desert plant assemblages of sagebrush- and shrub-steppe and piñon-juniper woodlands extends onto the Modoc Plateau in Northeastern California, where the piñon pine (as *Pinus monophylla*) component is rarely represented. These communities are shaped by patterns of fire frequency and intensity, successional dynamics and community structure, and soil type and quality (Gonzales and Hoshi 2015). As discussed in Chapter 1.1 (Dumroese, this synthesis, *The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope*), forest staffs and the public commented that portions of the forest and woodland ecosystems of the Lassen and Modoc National Forests (hereafter, the Lassen and Modoc) required attention beyond what was provided by two other syntheses that included these forests: the *Science Synthesis to Support Socioecological Resilience in the Sierra Nevada and Southern Cascade Range* (hereafter, Sierra Nevada Science Synthesis) published by Long et al. (2014) and the *Synthesis of Science to Inform Land Management Within the Northwest Forest Plan Area* (hereafter, Northwest Forest Plan Science Synthesis) published by Spies et al. (2018). These two valuable syntheses serve both to geographically frame the region that will be discussed in this *Northeastern California Plateaus Bioregion Science Synthesis* (hereafter, Plateaus Science Synthesis) and provide an excellent template for this author to adapt to the needs of the Lassen and Modoc.

National forests provide many different benefits to the Nation, including biodiversity, recreation (fig. 2.1.1.) and economic values. Often these benefits can be achieved through compatible management strategies; in other instances, priorities must be identified. This chapter does not seek to prioritize any benefits or actions as that process is more properly dealt with in the forest plan. However, no benefits will be provided in the future if forested ecosystems are no longer vigorous and productive. Given the diverse land use histories and diverse expectations about future climatic scenarios, this chapter will focus on what is known about the science of restoration in ecosystems in the Great Basin side of the Lassen and Modoc National Forests (see textbox 2.1.1 and see the discussion on restoration found later in this chapter). This discussion is often framed using examples, where appropriate, from similar ecosystems elsewhere in the

Figure 2.1.1—Fall colors in the Granger Canyon, Modoc National Forest. Given the importance of recreation on our national forests, managing scenic diversity is an important component of resource decision making (photo by Ken Sandusky, Forest Service).

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Great Basin with the goal of enhancing these ecosystems to be resistant to those disturbances they can resist and be resilient in the face of disturbances they cannot.

Based on the input during the public scoping process and from staffs of the Lassen and the Modoc, this chapter focuses on three issues in the context of dry-site ponderosa/Jeffrey pine and juniper woodlands:

- Pre-European settlement conditions and how they might guide current management,
- Fire history and its impact on current and future structure and composition, and
- In the face of changing climate, wildfire patterns, and local needs, how to increase resilience and resistance (see *Resistance and Resilience* below for definitions).

Textbox 2.1.1

- **Restoration** is the "resetting the ecological clock"– the returning of an ecosystem to a composition and structure that predates the disturbance.
- **Rehabilitation** repairs damage but does not focus on creating the former state.
- **Reallocation** may mend damage but pushes the ecosystem to an alternative land use or system. Millar et al. (2007) call this last action a "response," the transitioning of an ecosystem from the present undesirable or unsustainable state to new conditions.
- All three of these processes can involve **Repair**, which covers a variety of remediative actions, whether focused on the species, landscape, or ecosystem level.
- Hobbs and Cramer (2008), North et al. (2009b), and others presented a more detailed elaboration of restoration processes.

Topics Covered Under the Previous Syntheses

The Northwest Forest Plan Science Synthesis and the Sierra Nevada Science Synthesis have extensive coverage of important forest types that occur on the Lassen and the Modoc (fig. 2.1.2). Specifically, the forest types defined by Level IV Ecoregions 4e (High Southern Cascades Montane Forest), 4f (Low South Cascades Mixed-Conifer Forest), 4g. (California Cascades Eastside Conifer Forest), and 4h (Southern Cascades Foothills) were covered under the Northwest and Sierra Nevada science syntheses (see Appendix 1.1 in Chapter 1.1, Dumroese, this synthesis, *The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope* for full ecoregion descriptions). In addition, the Sierra Nevada Science Synthesis discussed several other forested ecoregions, especially those of Level III Ecoregion 4 (Sierra Nevada). Thus, the focus of this synthesis is largely on the Level IV Ecoregions that the public and forest staffs believed needed additional attention: eastside mixed-conifer forests (4g, 9q; mainly mixtures of ponderosa and Jeffrey pines) and juniper woodlands (9j) (fig. 2.1.3).

The Effect of Drought on Structure and Composition in Dry Forests

Any discussion of Great Basin ecosystems must acknowledge the limited annual precipitation coupled with hot summer temperatures that results in high rates of evapotranspiration and subsequent plant stress (Houghton et al. 1975). Water availability is one limit to the growing space of a tree or stand (Oliver and Larson 1996) and a combination of individual tree, species, and stand characteristics determines the landscape response to drought events. Fluctuations in climatic variables, including precipitation, place demands upon natural ecosystems; ecosystems are subject to even greater demands if they have been influenced by past management actions, including livestock grazing and fire suppression. Weather variability can create conditions that set up forests for a fall. For example, the climate of the Sierra Nevada since 1850 has been warmer and wetter than between 1650 and 1850 (Stine 1996, cited in Raumann and Cablk 2008), favoring conifer establishment and expansion. Even short-term fluctuations can result in severe disturbance events. For example, in Florida the above-average precipitation in winter 1997-98 resulted in a flush of ground vegetation growth that was all the more susceptible to the subsequent spring-summer drought, resulting in the most severe wildfire season in the State's history (Butry et al. 2001).

Temperature exacerbates any decrease in precipitation. A key element of any discussion of drought-influenced mortality in the Western United States is the expectation of higher temperatures during this century compared to the climate normal. Drought stress is a function of the high drying power of air (vapor pressure deficit or VPD) as well as lower available soil moisture. Vapor pressure deficit increases with increasing temperature and reduced relative humidity (Bradford and Bell 2017; Pallardy 2008). Average temperatures in the region are projected to increase by 3.2 to 4.3 °F (1.7 to 2.2 °C) by 2070 compared

Figure 2.1.2—Mount Shasta from Happy Camp on the Modoc National Forest. Northeastern California is situated at the intersection of the Southern Cascades, the northernmost Sierras, and the western arm of the Great Basin (photo by Chris Bielecki, Forest Service).

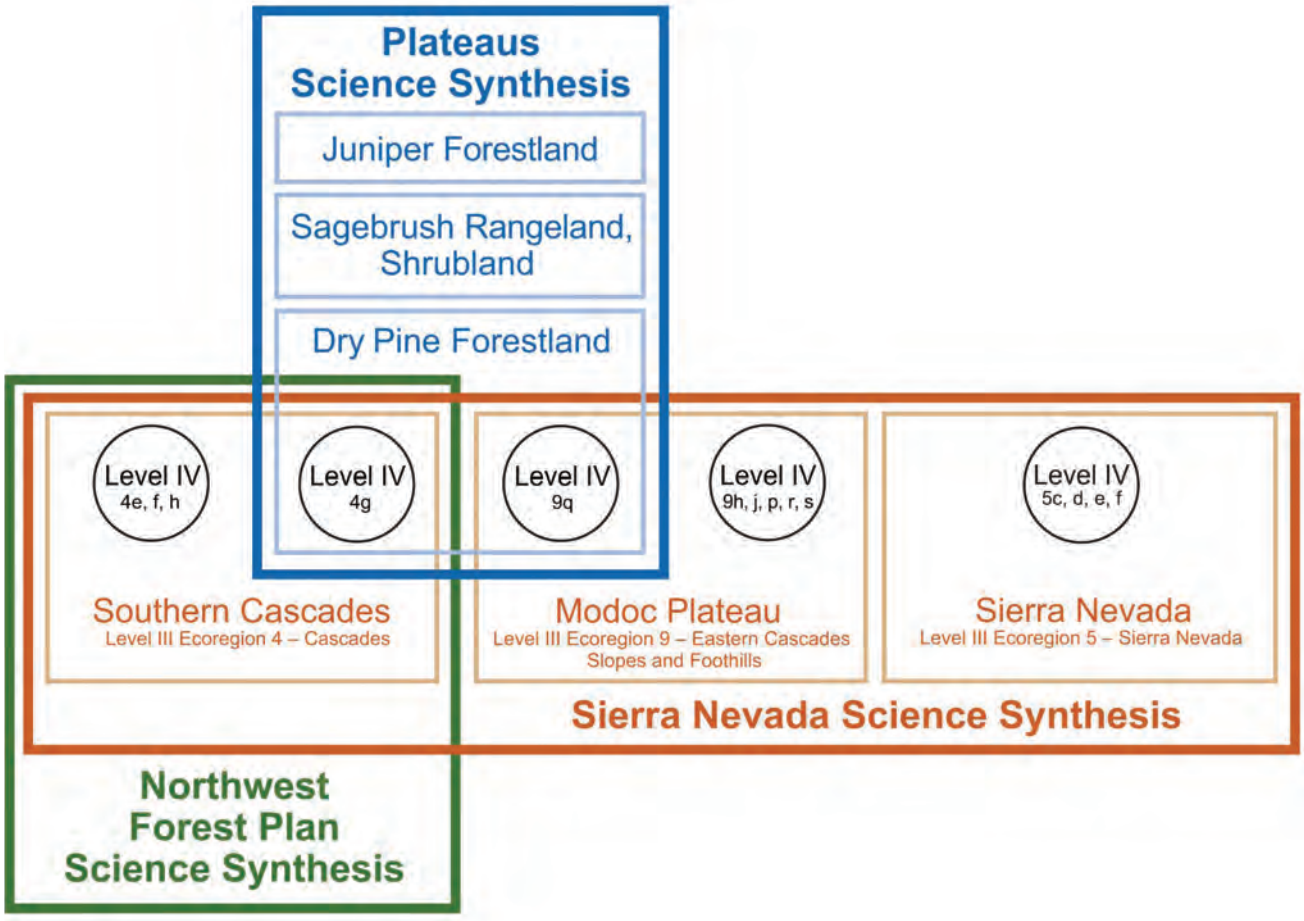

Figure 2.1.3—Ecosystem delineations used in the Northwest Forest Plan, Sierra Nevada, and Plateaus science syntheses. Because portions of the Northwest Forest Plan and Sierra Nevada Science Synthesis adequately covered the Lassen and Modoc National Forests, the Plateaus Science Synthesis focuses mainly on Dry Pine Forestland and Juniper Forestland, Sagebrush Rangeland, and Shrubland.

to the historical baseline of 1979–2009, and average precipitation is projected to decrease by 2 inches (5 cm) by 2100 ((Gonzales and Hoshi 2015; PRBO Conservation Science 2011).

Species Shift

A study of large trees and seedlings on U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis (FIA) plots found that climatic influences resulted in contractions and shifts of species, more so for montane species than subalpine (Bell et al. 2014). Twenty-nine percent of the contractions were movements away from warmer climates, perhaps reflecting the influence of increased winter precipitation. Although greater winter precipitation can aid the growth of large trees, it generally does not help with seedling establishment and growth, which are much more dependent on summer season precipitation due to the reduction of winter soil moisture before the growing season temperatures arrive (Pearson 1920).

Projections of future changing climate suggest that many variables will be altered, as has happened during climate events of the past. Some of these will be changes in absolute and seasonal precipitation and temperature, variability, and extremes. Depending upon the temperature variations, some of the changes in vegetation may be quite dramatic. Disturbances such as severe fire may accelerate these changes that might otherwise lag the temperature differences (Nolan et al. 2018). Projected temperatures may particularly affect vegetation composition and structure. The resulting vegetation communities may be quite novel in their makeup. Given the rapidity of projected change, some of the vegetation states may be quite temporary, with some projecting ecosystem stability being reached only in the 22nd century (Nolan et al. 2018).

Despite the potential for long-term increased temperature and drought to eliminate a species or even an ecosystem, particularly at the edge of its range (Bradley 2010; Miller and Wigand 1994), a more probable outcome is a modification of the composition and structure of landscapes currently forested (Clark et al. 2016). Drought-influenced mortality occurs more rapidly under higher temperatures (Vankat 2013). But projected environmental conditions, especially drought, are difficult to model because of the novelty of future climate scenarios and the complexity of interactions among climate, land use history, and disturbance.

In the Western United States, widespread drought coincides with a decline in tree growth (Clark et al. 2016), although not all drought events necessarily result in mortality (Van Gunst et al. 2016). Despite ponderosa pine's superior stomatal control and resistance to cavitation (the formation of embolisms within the plant's water-conducting tissues that cause hydraulic failure) (Maherali and DeLucia 2000), dry forest ecosystems can be susceptible to further droughty conditions (Bradford and Bell 2017), either through cavitation or carbon starvation, as the frequently closed stomata do not permit enough carbon uptake to maintain the system (North et al. 2009a). High temperatures during the growing season combined with reduced precipitation in the winter and early spring can explain much of the reduced growth rates of conifers in the Southwestern United States (Williams et al. 2013).

Competition for soil moisture can intensify the nominal effects of a drought. A comparative study of ponderosa pine in Arizona and South Dakota and red pine (*Pinus resinosa*) in Minnesota found that trees in low-density treatments displayed higher resistance and resilience to drought than trees in higher density treatments (Bottero et al. 2016). That study reported a negative relationship between growth and resistance and resilience to drought and tree population. Thinnings that reduce density can increase resistance and/or resilience to drought, at least for younger or smaller trees (D'Amato et al. 2013). Bradford and Bell (2017) found that tree mortality was positively related to temperature extremes and negatively related to winter and spring precipitation. Without the influence of other climatic factors, most of the species being studied displayed a strong relationship between mortality and density. Actions that target timber yields or fuels reduction, such as density reduction or prescribed fires, can also result in reduced drought vulnerability as fewer surviving trees are competing for a fixed amount of soil moisture. Multi-aged management can also create diverse size and age classes and therefore distribute risk (Clark et al. 2016; O'Hara 2002). But density reduction and similar measures may produce undesirable results, such as increased transpiration from larger trees with larger leaf/sapwood ratios (Clark et al. 2016) and temporary higher respiration loads of the remaining trees in newly thinned stands (Pallardy 2008) that can cause tree stress. In most piñon-juniper ecosystems, severe drought can result in the removal (by death) of water-demanding piñon. Thinning systems can certainly emulate this process and seek to favor more drought-tolerant species such as western juniper (Clark et al. 2016).

Researchers have a substantial understanding of the effects of drought on individual trees but less of an understanding about drought effects on communities or landscapes. Higher levels of carbon dioxide can increase water-use efficiency in drier areas, spurring further increases in the extent of woody encroachment into grasslands (Bradley 2010). Individuals in the canopy that might respond positively to favorable growing conditions are affected as well by neighbors that might also respond to those growing conditions (Clark et al. 2016). Any of several paradoxical outcomes may ensue. Faster growing trees might reach resource limitations more quickly, resulting in competition-induced mortality (Assmann 1970; Clark et al. 2016). Conversely, drought can slow growth and therefore reduce competition-induced mortality. The reduced vigor, however, can increase vulnerability to forest health issues, such as declines or even direct insect or disease attack (Manion 1991), as many pathogens can tolerate water stress better than the host trees (Clark et al. 2016).

Eastside Ponderosa Pine Forests[1]

Pre-European Settlement Conditions and How They Might Guide Current Management

Ponderosa pine occurs throughout the Western United States, with its greatest extent in Northeastern California (5 million acres, or 2.1 million ha) and the Northwestern Inland West (Graham and Jain 2005). Two varieties exist: the Rocky Mountain variety (*Pinus ponderosa* var. *scopulorum*), and the Pacific variety (*Pinus ponderosa* var. *ponderosa*), which ranges from British Columbia to California and Northwest Nevada (USDA NRCS 2017). It is generally believed that prior to Euro-American settlement (hereafter, simply "presettlement") ponderosa pine forests throughout the species' range were composed of large trees randomly distributed in an open, "park-like" stand of clumps at a large scale with few seedlings and saplings below the canopy (Sudworth 1900). The stands were believed to be uneven-aged, with occasional large individual trees 400 to 600 years old, which made them popular as witness trees in the region (Youngblood et al. 2004). Small gaps of even-aged cohorts were common (Crotteau and Ritchie 2014; Safford and Stevens 2017) although the frequent low- and moderate-severity events that historically occurred left more of a fine-grained pattern on the landscape that is difficult to assess years later (Safford and Stevens 2017).

The trees were large. One traveler in the area of the Lassen National Forest observed a stand of pines that had trees more than 10 feet (3 m) in diameter and up to 200 feet (61 m) tall (Reed and Gaines 1949; Safford and Stevens 2017). Van Hooser and Keegan (1988) found ponderosa pines in California exceeding 6 feet (183 cm) in diameter. While many might picture a balanced uneven-aged forest to have a reverse-J-shaped diameter distribution (de Liocourt 1898), Safford and Stevens (2017) point out that this is the distribution for an undisturbed forest. Historical studies found a flat or hump-shaped diameter distribution in old-growth forests with frequent low-severity fires (Lydersen and North 2012; North et al. 2007; Oliver 2001) and others as cited in Safford and Stevens (2017). Oliver (2001) demonstrated prescriptions for converting a J-shaped diameter distribution into a hump-shaped diameter distribution using fire. Taylor (2010) observed similar results from fire on studies on Beaver Creek and Devil's Pinery on the Lassen National Forest.

Ponderosa pine is an intermittent seeder (Bailey and Covington 2002; Savage and Swetnam 1990; White 1985). The study of old-growth stands by Youngblood et al. (2004) found that seed crops occurred about every 4 to 5 years with at least one seed crop every decade since 1850. Taylor (2010) found adequate seed crops occurring every 2 to 3 years. Due to above-normal precipitation from the middle of the 19th century through the second decade of the 20th century (Graumlich 1987; Taylor 2010), a higher proportion than normal of the seedfall successfully germinated (Garfin and Hughes 1996; Youngblood et al. 2004). As ponderosa pines often establish in gaps created by the death of one or more trees in the overstory, ponderosa pine forests start out as clumped. As they age, within-species competition thins them to a wider and more random pattern across the stand (Youngblood et al. 2004).

The species has evolved to survive in a fire-maintained ecosystem. Trees as small as 2 inches (5 cm) in diameter breast height (d.b.h.) can withstand the heat from most surface fires thanks to thick, insulating bark (Graham and Jain 2005). Bailey and Covington (2002) found 10-foot-tall (3-m) saplings with 4 inches (10 cm) of root collar diameter are fire resistant.

Although historical ponderosa pine forests may have been multi-aged, they did not necessarily have a multi-layered

[1]A comprehensive summary of ponderosa pine forests in Northern California can be found in Safford, H.D.; Stevens, J.T. 2017. Natural range of variation for yellow pine and mixed-conifer forests in the Sierra Nevada, southern Cascades, and Modoc and Inyo National Forests, California, USA. Gen. Tech. Rep. PSW-GTR-256. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 229 p

canopy. O'Hara et al. (1996) referred to such a structure as "old forest single stratum" (Youngblood et al. 2004). Munger (1917) observed ponderosa pine stands containing 12 to 40 trees of 12-inch (30.5 cm) diameter or larger per acre (30 to 99 trees per hectare [tph]) with very few seedlings or saplings in the understory (Fitzgerald 2005). On a protected old-growth site on the Blacks Mountain Experimental Forest in the Lassen, Youngblood et al. (2004) measured an overall density of 20 ± 1.4 trees per acre (50 ± 3.5 tph) with a mean diameter of about 24 inches (about 60 cm), with the distribution range from 10 to 49 inches (25 to 125 cm) d.b.h. These results were comparable to a study of an old-growth stand in Northern Arizona of 26 ± 2.8 trees per acre (65 ± 7 tph) (Covington et al. 1997), where fire has historically been the key disturbance (Moore et al. 1999).

Ponderosa Pine and Spacing

The mosaic of open space and grouping that Youngblood et al. (2004) observed most likely affords some resistance to fire by breaking up the canopy and reducing the vulnerability to crown fires (Fitzgerald 2005). Youngblood et al. (2004) found that no more than 3 to 15 percent of presettlement old-growth ponderosa pine forests in the Eastside Cascades of Oregon and California still existed. Their study of three old-growth forests in eastern Oregon and Northeastern California found that on two of the forests, at small scales, overstory trees were randomly distributed, and at larger scales, the trees were clumped, with the maximum radius of the clump being 70 to 80 feet (22 to 24 m). Trees in the upper canopy class that were aged were almost without exception larger than 14 inches (36 cm) diameter and older than 100 years (Youngblood et al. 2004).

Vegetative Complexes of Ponderosa Pine

Eastside Ponderosa Pine and Associates

Associates of ponderosa pine in California include western juniper, quaking aspen (*Populus tremuloides*), Jeffrey pine, lodgepole pine, Douglas-fir, incense cedar (*Calocedrus decurrens*), California black oak (*Quercus kelloggii*), and white fir (Oliver and Ryker 1990). In Northeastern California, ponderosa pine and Jeffrey pine forests grow on deep, well-drained soils. On dry sites south of Alturas, CA, ponderosa and Jeffrey pine are found together and the two species are often managed in the same manner. Ponderosa pine is the only native yellow pine north of Alturas (fig. 2.1.4). Jeffrey pine is generally found on lower-productivity sites than ponderosa pine. The lower productivity in Jeffrey pine stands probably resulted in longer fire-return intervals than on sites with ponderosa pine (Taylor 2000) and others as cited in Riegel et al. (2006a). On slightly moister sites, such as the west slope of the Sierras and the east side of the Cascades, ponderosa pine becomes more of a mid-seral species with Douglas-fir becoming a late-seral species. On sites where white fir is the potential vegetation type, disturbance, particularly fire, will keep the forest cover in ponderosa pine (Graham and Jain 2005).

Figure 2.1.4—Ponderosa pine is the dominant tree species on the dry forested areas of much of the Lassen and Modoc National Forests (photo by Ken Sandusky, Forest Service).

Stand Structure of Ponderosa Pine

Non-lethal fire caused by humans and lightning were instrumental in maintaining ponderosa pine forests in the Western United States (Graham and Jain 2005). Frequent dry-lightning days coincided with the season when the fine fuels were the driest. Frequent low fires killed small stems by cambial scorch, thus performing a thinning from below (Youngblood et al. 2004). Although these fires

Figure 2.1.5—The clumpy nature of ponderosa pine regeneration is apparent on the Lassen National Forest (photo by Bonnie Lou Millar, Forest Service).

that maintained ponderosa pine forests usually caused little mortality, fire intensity varied across the landscape. Intensive burning of patches followed by ponderosa pine regeneration would create a mosaic of even-aged patches. Variation of fire intensity in a patch would create uneven-aged patches (Taylor 2010), albeit often resulting in a fine-grained landscape (Safford and Stevens 2017). The clumpy nature of ponderosa pine regeneration (fig. 2.1.5) resulted from these heterogeneous effects of wildfire (Moore et al. 1999; Pearson 1950; Sánchez Meador et al. 2009; Youngblood et al. 2004). Regeneration processes were not the only process or function influenced by spatial patterns; others include tree mortality, snow accumulation and melt, wind patterns, and fire behavior (Larson et al. 2012). Even in even-aged groups, differentiation might occur due to phenotypic variation (Taylor 2010).

Influence of Large Ponderosa Pine on Forest Structure

In the dry forest ecosystems of the Pacific Northwest, large ponderosa pines are the principal drivers of forest structure and function (Franklin and Johnson 2012). Even putatively mixed-conifer forests were dominated by ponderosa pine at low densities. The total large tree (greater than 21 inches or 53 cm) basal area (of all species) has declined by more than 50 percent and the proportion of these large trees compared to the total number of trees is only 20 percent of the historical numbers (Hagmann et al. 2013), and are at much greater risk of catastrophic fire now than before. Fire suppression has dramatically increased lodgepole pine density in Central Oregon since the 19th century. Besides the obvious implications for the susceptibility to fire-induced and density-dependent mortality, another impact on stand dynamics is being noted, as ponderosa and lodgepole pines in these dense stands are taking longer to grow to a given height (Hagmann et al. 2013; Shuffield 2010).

Fire History and Its Impact on Current and Future Structure and Composition

Presettlement fire frequency throughout the ponderosa pine range was less than 40 years (Agee 1993). In the southern edge of its range, presettlement fires were a mixture of low- and high-intensity fires (Fulé et al. 1997). Historical frequency of low-intensity fire ranged from 4 to 11 years in the Eastside Cascades (Youngblood et al. 2004 and citations therein), a median interval of 7 years (mean of 11 years) on dry ponderosa pine sites in California (Van de Water and Safford 2011), 5 to 29 years on the west side of the Northern Sierras and 3 to 8 years on east side (Raumann and Cablk 2008; Taylor 2004). Fire frequency

of around 12 years in old-growth ponderosa pine forests in the Ishii Wilderness in Northeastern California was comparable to ponderosa pine forests on the east side of the Southern Cascades and to mixed-conifer (but predominantly ponderosa pine) stands in the Klamath and Sierra mountain ranges (Taylor 2010). Although declines in fire frequency are likely due to the cessation of Native American burning, widespread grazing on public lands that reduced fine fuels, and reduced ignitions after the California Gold Rush, the main driver of reduced fire effects across the landscape was the State and Federal mandates to suppress all fires (Taylor 2010). Subsequently, ponderosa pine sites on the South Lake Tahoe Basin indicated no evidence of fires for 62 years (Raumann and Cablk 2008) and in Northern Arizona the post-exclusion interval was 120 years, much longer than the historical fire-return interval of 2 to 8 years (up to 15 years for larger fires) (Moore et al. 1999).

Historically, lightning was the main source of ignitions in ponderosa pine forests and continues to play a significant role (Fitzgerald 2005; Hagmann et al. 2013), whereas the role of Native Americans in burning was probably low in these regions (Youngblood et al. 2004). A belt from Northern California through Oregon and Idaho and into Northern Montana contains high numbers of lightning strikes and the resulting fires (Schmidt et al. 2002). Although it is difficult to determine the extent of fires in presettlement times, the prevailing opinion is that burn areas were small. In Northern California, for example, the average area of burn was 850 acres (350 ha), although 16 fires between 1627 and 1992 were believed to be larger than 1,200 acres (500 ha) (Taylor and Skinner 1998). Another analysis determined that some fires ranged up to thousands of hectares in size in Northeastern California (Norman and Taylor 2003).

Fire frequency in ponderosa pine forests throughout the West depended upon elevation and their associated vegetation. At drier, lower elevations, frequent surface fires occurred in these forests. At higher elevations on moist sites, where ponderosa pine was frequently associated with species such as Douglas-fir, white fir, or lodgepole pine, the fire-return interval was longer and the fire intensity ranged from surface fires to stand-replacing fires (Fitzgerald 2005).

Seasonality influences fire behavior and fire effects. In Northern California, 93 percent of all fires occurred between the dry midsummer and early fall (Taylor and Skinner 2003). In contrast, 40 percent of presettlement fires in southwestern ponderosa pine in Arizona occurred in the spring and the remainder were summer fires (Fitzgerald 2005; Fulé et al. 1997).

Mean presettlement fire-return intervals in ponderosa pine forests ranged from 2 to 50 years (table 2.1.1). The Northern California study by Taylor and Skinner (1998) also observed that the postsettlement fire-return intervals (22 years) were longer than during the settlement (12 years) or presettlement (14 years) eras. Furthermore, they found that the interval was shorter on south- and west-facing slopes, a pattern also observed by some (Heyerdahl et al. 2001) and not by others (Everett et al. 2000).

The ability to survive a fire may be either species- or individual tree-based. Species-based attributes include fire-induced flowering, seed dispersal from serotinous cones, such as those on lodgepole pine, or persistent seed buried in the soil, such as those from chokecherry (*Prunus virginiana*). Individual tree-based attributes include thick bark, such as that on ponderosa pine and California black oak, sprouting from the stump, such as shortleaf pine (*Pinus echinata*) and various oak (*Quercus*) species, or

Table 2.1.1—Presettlement fire-return intervals in ponderosa pine ecosystems.

Area	Interval (years)	Source
West wide	2–47	Fitzgerald (2005)
Northern California	21	Norman and Taylor (2003)
Northern California, Lassen National Forest	11	Norman and Taylor (2005)
Central Oregon	30–50	Volland (1963)
Eastern Washington	7	Everett et al. (2000)
Northern Arizona	2–15 (median of 4)	Fulé et al. (1997)
Southern Cascades	12	Taylor (2010)

Figure 2.1.6—To reduce fuels on the Modoc National Forest, woody residues after harvesting are piled, dried, and frequently burned during late fall or winter, when surrounding snow cover reduces the chance of the fire escaping (photo by Ken Sandusky, Forest Service).

having grass-stage needles that protect the apical bud, such as longleaf pine (*Pinus palustris*). These attributes are necessary for trees to persist in a fire-mediated environment (Grace and Platt 1995; Kauffman 1990). Miller (2000) concluded that ponderosa pine is one of the most fire-resistant species in the Western United States and this resistance was believed to increase as the tree matured. Multiple characteristics contribute to improving the chances for the individual pine trees to survive a fire: thick bark that sloughs off when on fire to minimize heat transfer; a deep rooting habit; and an open crown structure that allows for dissipation of heat and reduces the potential for crown scorch (Brown and Wu 2005; Cooper 1961; Fitzgerald 2005; Moore et al. 1999).

Fire, Density, and Tree Size

In late-seral interior ponderosa pine stands, large 200- to 500-year-old trees are an important constituent for their contribution to forest structure and habitat (Ritchie et al. 2008). Despite their prominence in the stand, however, dense understory vegetation can negatively influence their growth and survival. Ritchie et al. (2008) evaluated a study (Oliver 2000) on Blacks Mountain Experimental Forest that compared an unburned, unharvested, 65-year-old stand with neighboring treated stands. On the experimental forest in the 1990s, plots first measured in 1934 were treated to create low-diversity (removing all large overstory trees) and high-diversity (thinning from below) structures that were then subjected to burn and no-burn treatments (Ritchie et al. 2008). Untreated stands displayed a dramatic increase in density of trees less than 12 inches (30 cm) diameter between 1934 and 1999. The proportion of the total basal area in pine declined by 12 to 20 percent in that time, mainly replaced by white fir and incense cedar. Thinned and burned stands (similar to those in fig. 2.1.6) adjacent to the untreated stand had fewer large trees at risk and lower rates of mortality (Ritchie et al. 2008). Such treatments are, however, not without risk. Careful thinning can accelerate growth of old pine trees (Kolb et al. 2007), but prescribed burning on a site with a long history of fire exclusion may

result in secondary mortality of the older trees (Crotteau and Ritchie 2014; Ritchie et al. 2008). Delayed effects upon the entire forest ecosystem can occur, too. Not only does fire remove snags, but snags from fire-killed trees deteriorate faster than trees killed by other causes, such as insects (Laudenslayer 2002).

Land Use Practices and Their Impact on Fire

Livestock grazing and timber harvesting were widely practiced in ponderosa pine forests by the end of the 19th century (Graham and Jain 2005; Madany and West 1983; Rummell 1951). In 1880 in Northeastern California, 45,000 grazing animals were reported; this number increased threefold by 1909. This intensive grazing pattern reduced fine fuels and native grasses and palatable shrubs; consequently, annual grasses and unpalatable trees and shrubs (e.g., sagebrush and western juniper) became established (Laudenslayer et al. 1989; Riegel et al. 2006). In comparison, grazing in the Southwestern United States reduced grass competition on ponderosa pine sites and the cessation of grazing, coupled with favorable seed crops and climate, resulted in dense stands of ponderosa pine regeneration (Moore et al. 1999; Pearson 1950; Savage and Swetnam 1990). Postsettlement, heavy grazing and fire suppression changed the fire regime in ponderosa pine forests, allowing regeneration to proceed without barrier and fuels to accumulate to much-greater-than-historical levels. Furthermore, the high-grading of large ponderosa pine, Douglas-fir, and western larch (*Larix occidentalis*) throughout the range of the pine resulted in greater densities of thin-barked, shade-tolerant species in the understory, creating ladder fuels and forest stands more prone to damage from insects and drought (Fitzgerald 2005).

Fire Suppression and Forest Types

Fire suppression resulted in more pronounced effects in forests subject to frequent low-intensity fires, such as pine forests, than in more moist forests. For example, in a study of the South Tahoe Lake Basin, Raumann and Cablk (2008) found stands subject to more than 6 decades of fire suppression to be much denser than those under natural fire regimes. Although forest stands up to 9,800 feet (3,000 m) displayed increased density, about three-fourths of the densification occurred below 7,500 feet (2,300 m), which was the upper limit of Jeffrey pine-white fir forests (Raumann and Cablk 2008). The authors found an increase in forest extent from 1940 through 1969 due to regeneration after harvest. After 1969, they determined that most forest expansion in undisturbed areas was succession.

Climate, Elevation, and Forest Types

Relationships exist among biophysical site characteristics, vegetation associations, and resulting disturbance frequencies and responses. Following the Life Zone Concept (Merriam 1898), which defined ranges of vegetation types in the Southwestern United States, Stephens et al. (2015) found that the most important variables for defining stands in the Southern Sierras were actual evapotranspiration, elevation, and aspect. Each ecosystem has a unique response to natural and human-caused disturbance. In a study of a mixed-conifer forest in the Lake Tahoe Basin, Van Gunst et al. (2016) determined that density-dependent mortality occurred more on lower-elevation forests with drier climates than on forests at higher elevations. Further, they found that density-dependent mortality declined as density increased in these mid- to upper-elevation forests. Separating density from drought, they surmised that lower-elevation forests may experience less drought-related mortality as the pine-dominants (Jeffrey and sugar pine [*Pinus lambertiana*]) are more drought-tolerant than other species that may occur on the site. They attribute this drought resistance to greater stomatal control and resistance to cavitation, as was observed in ponderosa pine by Maherali and DeLucia (2000). They did find greater mortality on north-facing slopes, regardless of density, suggesting that these communities did not typically face droughty situations and were thus more susceptible to drought events when they did occur.

Changes in Ponderosa Pine Forests

Grazing, Fire Suppression, and Climate Change

Where fire, grazing, and timber harvesting have stopped on moist sites (where the potential vegetation is white fir), ponderosa pine is being succeeded by Douglas-fir and white fir (Graham and Jain 2005) (see fig. 2.1.7). In some places this succession is occurring so rapidly that the normal progression from ponderosa pine to Douglas-fir of 400 years is being reduced to 50 years because of the lack of non-lethal disturbance (Graham and Jain 2005). Several factors influenced the change in pine regeneration, the most notable being the change in grazing practices (from sheep to cattle), suppression of natural, heretofore low-intensity fires by the Forest Service in 1905, and climate (Norman and Taylor 2005), although the climate influences are not as definitive as one might expect, as Norman and Taylor (2005) referred to evidence that cooler, moister springs and drier summers aided successful regeneration

Figure 2.1.7—Young ponderosa pine in the Pine Creek Valley of the Lassen National Forest (photo by Ken Sandusky, Forest Service). Fire, grazing, and available soil moisture are some of the influences that minimize pine regeneration in such open parks.

of Jeffrey and ponderosa pines. (See Chapter 6.1, Wright, this synthesis, *Ecological Disturbance in the Context of a Changing Climate: Implications for Land Management in Northeastern California* for more discussion about climate-induced changes to fire.) Paradoxically, the change in disturbance patterns created an environment where the more infrequent but more severe disturbances also negatively impacted forest regeneration. Severe wildfires occurring after decades of fire suppression can result in the lack of pine trees as a seed source and what little regeneration that does occur is mainly shade-tolerant, fire-intolerant species (Welch et al. 2016).

Youngblood et al. (2004) attribute the decline of Eastside Cascades old-growth ponderosa pine to fire suppression, livestock grazing, selective logging of old trees, and road building. Managers have responded by trying to restore disturbance frequency and stability by thinning, underburning, and other fuels-reduction treatments (Fiedler et al. 1996; Youngblood et al. 2004).

Juniper Woodlands

Piñon-Juniper Ecosystems

In the Western United States, piñon-juniper ecosystems cover 100 million acres (40 million ha). Several pine and juniper species grow together, or separately, across the piñon-juniper complex. In the Great Basin, the principal species are *Pinus monophylla/Juniperus osteosperma* (singleleaf piñon/Utah juniper) but continuing westward into Northwestern Nevada and neighboring California, the principal species is western juniper (Romme et al. 2009) with low abundance of piñon. Juniper woodland has greatly increased in both spatial distribution and density (Miller et al. 2008); however, historical evidence indicates that piñon-juniper increased and decreased in extent over the previous millennia (Miller and Wigand 1994).

Western juniper is the only representative of the Great Basin piñon-juniper complex in Northeastern California. It is typically the only conifer on the site, except along an ecotone with ponderosa pine. Western juniper began migrating northward from the Lake Lahontan basin 10,000 years before present (BP) (Tausch 1999) and reached its present-day range in Northeastern California and Southeastern Oregon between 7,000 and 4,000 years BP (Mehringer and Wigand 1987, as cited in Riegel et al. 2006a). Before Euro-American settlement in the mid-19th century, juniper occurred mainly on rocky surfaces or sandy sites with little vegetation (Miller and Rose 1995). In this region, old juniper stands are primarily found on shallow soils with heavy clays, and are generally open (Riegel et al. 2006). On sites with low fuels, western juniper can reach 1,000 years of age (Waichler et al. 2001). Sites near Devil's Garden in the Modoc contain old-growth juniper (200 to 500 years old) with some individuals as old as 700 years. The wide spacing of the trees implies that there were few stand-replacing fires. Disturbance limited western juniper to low-productivity sites before Euro-American settlement. But the species can grow in almost any type of soil. Since settlement, western juniper has established on more productive sites (Adams 1975; Burkhardt and Tisdale 1969, 1976; Miller and Rose 1995, 1999). This expansion is a result of an increase in fire-return interval, grazing, and higher-than-average annual precipitation as paleoecological evidence suggests that the presettlement fire-return interval kept the species off these sites.

In California in 1989, the total area of western juniper and piñon-juniper woodlands stands with at least 10 percent cover in tree crowns was estimated at 2.4 million acres (1 million ha) (Bolsinger 1989), with 1.1 million acres (0.4 million ha) being on national forests. This total did not include woodlands with less than 10 percent cover or located on other public land or private land, or scattered trees. Before the arrival of Euro-Americans, western

juniper was primarily restricted to areas with poor soils, which reduced the amount of fine fuels able to carry fires (Burkhardt and Tisdale 1969; Miller and Rose 1995). Favored by Euro-American settlers for grazing of cows and sheep due to the proximity to inhabited areas, the land cleared of trees to improve forage has been offset over the years by the afforestation of rangeland due to fire suppression; thus, the overall quantity of land in these woodland types has changed little since 1945 (Bolsinger 1989). Recent trends suggest that afforestation is decreasing, as reflected in the increasingly large proportion of western juniper and piñon-juniper stands greater than 100 years old (73 percent and 59 percent, respectively). Bolsinger (1989) observed that stands less than 50 years old make up only 12 percent of western juniper and 9 percent of piñon-juniper stands. Singleleaf piñon and western juniper together account for 93 percent of total wood volume in such woodlands. Grass cover is higher in western juniper stands than in piñon-juniper stands (Bolsinger 1989).

While in Northeastern California the western juniper woodlands (fig. 2.1.8) are a continuation of the extensive piñon-juniper woodlands of the Great Basin (Young and Evans 1981), piñon becomes scarce. Of the 60,000 acres (24,280 ha) of juniper habitat on the Lassen, less than 1 percent is piñon-juniper (Bolsinger 1989) and the author found no piñon-juniper on the 470,000 acres of (190,200 ha) juniper habitat on the Modoc. Juniper woodlands are largely found in Klamath Juniper Woodland/Devil's Garden (Level IV Ecoregion 9j), Modoc/Lassen Juniper Scrub (9p), Adin/Dixie Low Hills (9r), and Modoc Lava Flows and Buttes (9s) ecotypes (see Appendix 1.1 in Chapter 1.1, Dumroese, this synthesis, *The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope* for ecoregion descriptions). On the Lassen, western juniper occurs in big and low sagebrush communities (table 2.1.2).

Presettlement Conditions and How They Might Guide Current Management

Except for western juniper, common species of juniper are all associated with piñon (Miller and Wigand 1994). Not surprisingly given their wide distribution across the Western United States, most research has focused on piñon-juniper systems. Across these different juniper-dominated ecosystems, some patterns concerning juniper emerge:

- Mammals and birds spread seed more than 100 meters.
- Climate and disturbances affect the rate of infill between early-established trees.
- Understory floral diversity progressively decreases as juniper overstory increases.

When available, this chapter draws on research from pure juniper woodlands but also notes work within piñon-juniper woodlands because the similar stand dynamics, disturbance patterns, and climate responses are relevant to management questions on the Lassen and Modoc. Furthermore, some climate projections propose that the future weather will be hotter and drier (Clark et al. 2016), suggesting that future Northeastern California woodlands might behave in a manner more like those of the Central Great Basin and the constituent juniper species.

Figure 2.1.8—Western juniper, like most *Juniperus* species, can form closed-canopy, woodland, and savanna landscapes, depending upon site productivity, precipitation and disturbance (fire) interval. The Warner Mountains viewed from the Devil's Garden, Modoc National Forest (photo by John Cichoski, Forest Service).

Table 2.1.2—Summary of western juniper/sagebrush vegetation communities in Lassen County, CA (Young and Evans 1981).

	Western juniper/big sagebrush	Western juniper/low sagebrush
Soils	Deep, 10–32 inches (25–80 cm), heavy clay loam	Biscuit and swale, mounds 4–8 inches (10–20 cm), wet in spring and baked hard in summer
Trees per acre (trees per hectare; tph)	60 (150 tph) 10 (25 tph) after burns 40 (98 tph) since 1900	11 (28 tph) 2 (5 tph) since 1900
Canopy cover	40 to 60 percent	–
Age (year established)	84 percent established between 1880 and 1920; oldest 1855	Oldest 1600

Western Juniper

In the late Pleistocene, the more drought-tolerant juniper species were 310 to 400 miles (500 to 640 km) further north and 3,280 to 4,920 feet (1,000 to 1,500 m) lower in elevation than they are now. Western juniper (probably var. *australis*) grew in Kings Canyon, CA, at the glacial maximum. It was in the area of Lake Lahontan, NV, about 12,000 years BP and arrived in Northeastern California and Southeastern Oregon 5,000 to 8,000 years later (Miller and Wigand 1994).

The range of western juniper has greatly increased since the 1880s (Eddleman 1986). In Northeastern California, the peak years of establishment range from the late 1800s to the early 1900s (Young and Evans 1981), and in Southern Oregon, the early 1900s (Adams 1975). In the sagebrush steppe, presettlement juniper was primarily in shrub form on shallow, rocky soils, where there was little fine fuel. Most of the juniper woodlands on the landscape today became established as annual precipitation increased and fire frequency decreased after the late 1800s (Riegel et al. 2006). Little older western juniper was left on the land because mortality of juniper trees due to fire is much higher for trees less than 50 years old than for the older cohort (Riegel et al. 2006). Because of management and altered disturbance histories on the sites, western juniper stands can become quite dense. In Central Oregon, Eddleman (1986) found juniper densities averaging 412 trees per acre (1,018 tph), with a range of 335 to 450 trees per acre (830 to 1,120 tph), although he stated that his results were higher than other contemporaneous studies. Even 50 years ago, Burkhardt and Tisdale (1969) reported that some seral stands of western juniper had densities of more than 800 trees per acre (2,000 tph).

Western juniper is less dependent on precipitation than juniper communities to the west and south, although prolonged wet or dry spells can influence tree cover and growth. For example, relatively high winter precipitation in Southeastern Oregon in the late 1800s and early 1900s resulted in extensive recruitment of western juniper (Romme et al. 2009). Paradoxically, the western juniper that benefited from increased precipitation does not, in turn, improve water retention or soil moisture potential on the land. Tree canopy can intercept precipitation. In California, western juniper woodland cover of 40 percent intercepted 15 to 20 percent of precipitation (Evans 1988). Miller and Wigand (1994) found that heavy juniper cover results in reduced ground flora cover, which in turn results in damaged hydrologic process due to less infiltration and more runoff and erosion.

Western juniper primarily invades big sagebrush communities and to some extent, low sagebrush communities (Burkhardt and Tisdale 1969, 1976; Young and Evans 1981). In Central Oregon, Eddleman (1986) determined that woody plants, whether sagebrush or juniper, act as nurse plants for western juniper seedlings, similar to what Phillips (1909) observed for piñon pine (*Pinus edulis*), very likely due to reduced soil surface temperature and increased shading, and perhaps higher availability of moisture and nutrients.

In a study in Southeastern Oregon, Adams (1975) documented the interactions between western juniper and associated shrub species. Western juniper increased from less than 59 percent of crown area in 1929 to 73 percent in 1966. Shrub crown cover declined 36 percent during the same period even though total plant density declined by only 2 percent. The author attributed the disparity to the replacement of large-crowned bitterbrush (*Purshia tridentata*) and big sagebrush (86 percent of shrub density in 1920s, only 33 percent in the late 1960s) by narrow-crowned green rabbitbrush (*Ericameria teretifolia*). In Northeastern California, Loft (1998) estimated that in a

40-year period (1957–1997) the overstory of pine and juniper increased by more than 400 percent on bitterbrush range. Adams (1975) suggested that a certain synchronicity existed between juniper and the associated vegetation, with few shrub plants being established in years where juniper had low growth ring indices.

Young and Evans (1981) intended to examine the cause of juniper establishment (i.e., wildfire suppression, grazing, or climate change), but no ungrazed areas were available to test a grazing hypothesis, the relatively short period of time of invasion and establishment, and the dense stand's individual tree growth rate would make it difficult to explore climate influences. They pointed out that trees less than 50 years old were very susceptible to fire damage, suggesting that perhaps fire suppression played a role. Both locations were "fireproof"—the western juniper/big sagebrush site because it was so dense that no ground vegetation grew under the canopy (Bruner and Klebenow 1979) and the western juniper/low sagebrush site because of heavy grazing of the native grasses (Young and Evans 1981). They observed that only about 10 percent of the trees in the western juniper/big sagebrush stand bore fruit, with the result that the seed source in western juniper/big sagebrush stand usually came from trees in the western juniper/low sagebrush stand.

Juniper Overstory vs. Forage Understory

There is an inverse relationship between overstory cover and understory plant cover (Tausch and Tueller 1990). The strong interaction between overstory tree canopy density and understory forage productivity is often exacerbated by human activities. Drier sites on shallow soils on south-facing slopes show greatest western juniper impact on plant community. Miller and Wigand (1994) noted that desirable browse in some bitterbrush ranges was shaded out by a 400 percent increase in overstory pines and junipers. Fire suppression and grazing created the conditions that allowed juniper woodlands to move from their low-productivity sites onto rangeland (Burkhardt and Tisdale 1969; Miller and Rose 1995; Miller and Wigand 1994).

Clearing sites of western juniper to restore or create rangeland for grazing is a common practice in the Interior West (fig. 2.1.9). To test a site's ability to respond to efforts to improve ecosystem resilience, in 1991 sites in a western juniper woodland in Southeastern Oregon were harvested to evaluate the understory response (Bates et al. 2000). The site was at full occupancy and most of the co-occurring sagebrush shrubs were dead. Juniper canopy cover was 24 percent of the area and most of the remainder was bare ground, subject to erosion. Dry conditions region-wide limited treatment response the following year. In 1993, however, an almost order-of-magnitude difference in understory biomass was found on the cut plots compared to the wooded plots. Canopy cover of herbaceous species increased by a factor of 3 in the interspaces. Besides the obvious competition for water through their extensive root systems, Bates et al. (2000) also concluded that juniper interfered with nitrogen uptake of other plants as an additional competitive advantage. The authors noted that the postdisturbance ground vegetation community was connected with pre-disturbance composition (fig. 2.1.10). The expansion of herbaceous cover in the open areas was also a significant benefit in reducing erosion.

Juniper Woodlands in the West

Throughout juniper's range, three types of juniper woodlands exist: persistent woodland, wooded shrubland, and savanna (Romme et al. 2009; Vankat 2013). In persistent woodland, soils, climate, and disturbance patterns favor juniper; the fire regime consists of infrequent but high-severity crown fires. Persistent juniper woodlands were minimally affected by fire-suppression efforts as 20th-century fire management practices did not usually target such ecosystems (Vankat 2013). The dynamics of persistent woodlands are driven more by climate, disease, and insects than by fire. Less is known about wooded shrubland and savanna (Romme et al. 2009).

In juniper savannas, soils favor juniper, but fire is more frequent. In wooded shrubland, which have a shrub layer beneath an open tree canopy (Vankat 2013), soils support woody plants. But species vary between deep soils and thin rocky sites, particularly where winter precipitation constitutes a higher proportion of the total annual precipitation. Periodic wildfire may result in oscillating woodland conditions (West and Van Pelt 1986). Figure 2.1.11 displays how periodic wildfires might cause the ecosystem to alternate between woodlands and savannas.

In moist periods, the number of trees will increase, whereas in dry periods, the number of trees will decrease (Romme et al. 2009; Tausch et al. 1981). Savannas have grass and forb understory under an open tree canopy, generally on sites with moderately deep soils. In the Southwestern United States, juniper is found at low elevations adjacent to grassland ecosystems, but can also be found at higher elevations (Vankat 2013).

Figure 2.1.9—The varied nature of structure in western juniper stands provides varied habitat for native and introduced species, including these wild horses up Pencil Road Trail on the Modoc National Forest (photo by Ken Sandusky, Forest Service).

Patterns of Expansion

Cycles of high and low precipitation cause corresponding responses of piñon-juniper establishment and mortality (Chambers et al. 2013; Swetnam and Betancourt 1998). Juniper and associated woodlands expands across the landscape in one of three ways: (1) infill—in existing woodlands, the tree density within the stand expands by growth of the existing trees and by establishment of new trees; (2) expansion—trees become established in formerly treeless lands, such as sagebrush or other shrubland; and (3) succession—the successional pathway following stand-replacing fire progresses from annual herbaceous species to forbs and perennial grasses and then, after several decades, to shrub cover (fig. 2.1.11). Tree seedlings then follow, and woodland forms in 100 to 300 years (Vankat 2013). Huffman et al. (2012) also found a relationship between juniper presence and time since fire. Seedling survival in woodlands requires an enhancement of the site to reduce the environmental stresses and successfully establish. This addition may take the form of physical cover or nurse plants, even other juniper. Juniper species will establish in the interspaces, and then will expand if drought kills the competing components (Vankat 2013).

Figure 2.1.10—The influence of juniper on forage and the potential for restoration depends upon the density and length of tenure of juniper, such as this open-grown western juniper on the Lassen National Forest, and the mix of ground flora that existed prior to afforestation (photo by KC Pasero, Forest Service).

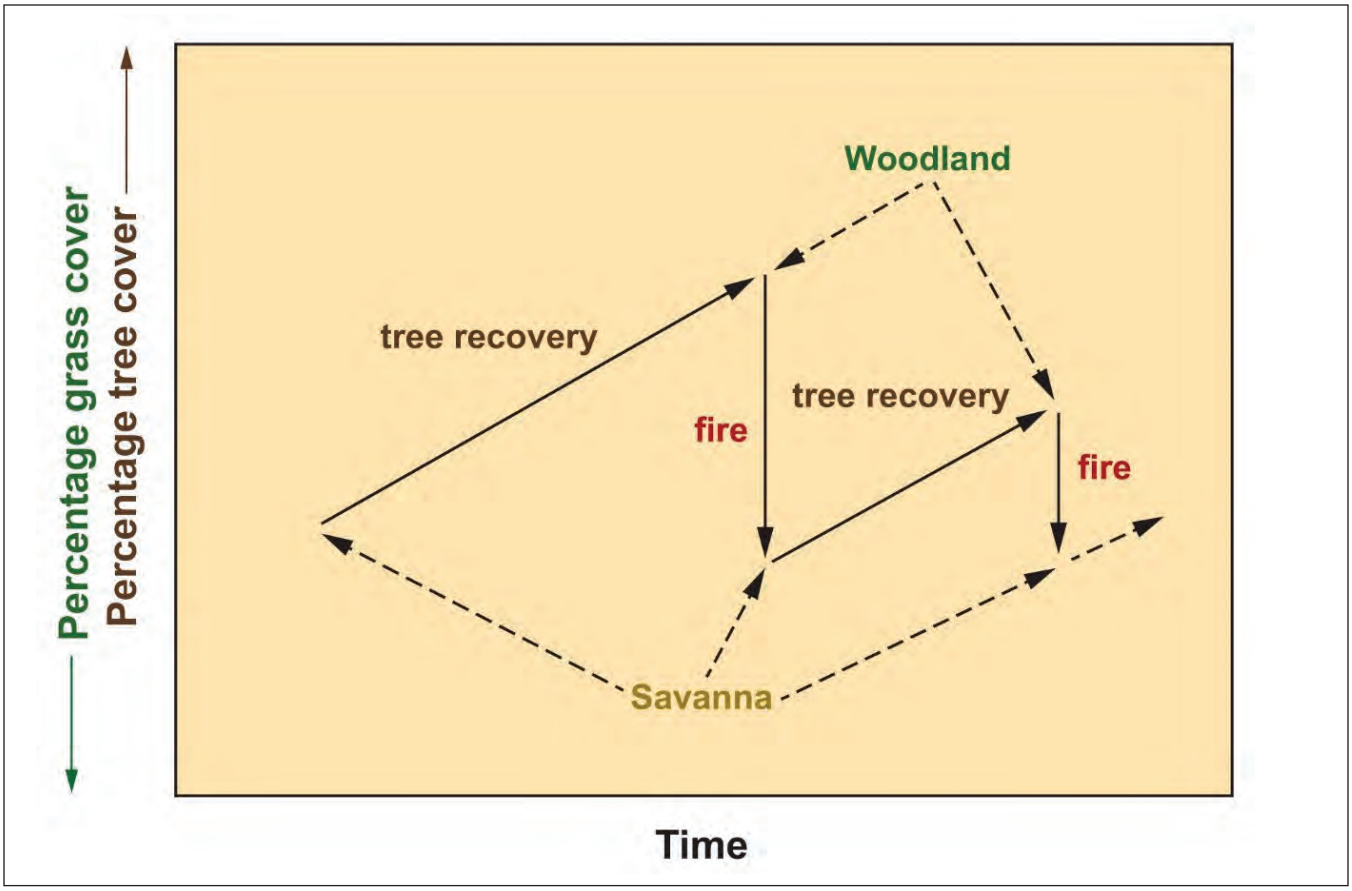

Figure 2.1.11—One model of periodic wildfires causing intercommunity cycles between juniper woodland and savanna (adapted from West and Van Pelt 1986).

The number of trees increased rapidly in the Great Basin during the 19th century (Tausch et al. 1981). A study of various juniper ecosystems in Oregon, Idaho, Nevada, and Utah (which included western and Utah juniper) found that prior to settlement, woodlands had slow rates of establishment and low densities. Piñon and juniper trees established at a higher rate in the late 1800s than they had in the several centuries before, beginning after 1860 in Utah and Nevada and after 1850 in Idaho. These species increased gradually later in the 19th century, then more rapidly in the early 20th century. The rate of establishment then declined later in the 20th century (Miller et al. 2008).

The historical increase in tree density occurred at roughly the same time across the Great Basin. Scientists have suggested several possible causes, including warmer temperatures from 1850 through 1960, wetter weather between 1850 and 1900, fire suppression, and increased cattle grazing and timber harvesting for mining (Chambers et al. 2013; Tausch and Hood 2007). Climatic conditions were milder and wetter in the latter part of the 19th century and the first half of the 20th century than they were in the latter half of the 20th century (Antevs and Wright 1938; Graumlich 1987; Miller et al. 2008). Apparently, this increase in tree number was successful, as the authors go on to suggest that the decline in recent establishment may be due to the lack of suitable sites unoccupied by trees.

Tree density in juniper woodlands of the Great Basin has also increased during the last 150 years (Romme et al. 2009), with the current density 4 to 10 times greater than the presettlement conditions (Tausch and Hood 2007). Infill has occurred locally in persistent woodlands, but infill in this type was not so widespread across the Great Basin region. Infill has occurred much more in wooded shrubland and savannas, where elimination of fine fuels due to timbering and grazing resulted in changes in the fire regime (Romme et al. 2009; Tausch et al. 1981). Grazing increased tree density by reducing fine fuels and competing vegetation in mixed-conifer forests, but had less of an effect on the density of piñon-juniper and juniper woodlands. Forty percent of the trees in the Great Basin were established less than 150 years ago (Tausch et al. 1981).

Grazing Impact in Piñon-Juniper Woodlands

Grazing had little influence on tree cover in persistent woodlands. In wooded shrubland and savanna, grazing resulted in increased tree cover and different ground vegetation species as previously dominant herbaceous species were eliminated, for example with western juniper in Oregon (Miller and Rose 1995). Subsequently, considerable attempts have been made to improve grazing on wooded shrublands and savannas by using mechanical treatments such as chaining or cutting (Vankat 2013). Elimination of shrub and grass species was not inevitable, however. Cottam and Stewart (1940) found that sites in Southern Utah that had deep soils, grasses, and sagebrush could withstand Utah juniper invasion, but on poorer sites in the foothills, junipers replaced these species. The piñon-juniper communities are very susceptible to severe wildfires. Old-growth woodlands on presettlement landscapes were usually located on sites with limited fuel loads, often with thin soils or steep slopes, or both, where disturbance was not frequent (Tausch and Hood 2007; Tausch et al. 1981; West et al. 1998). In contrast, very productive sites on deeper soils generate greater fine-fuel loads and thus experience more frequent fires. Old-growth woodlands could also be protected from severe fire by the frequently burned sagebrush communities that surrounded them.

Drought

Juniper is more resistant to drought and is affected by fewer biotic agents than piñon (Vankat 2013). Juniper exhibits low natural mortality in the Great Basin (Adams 1975). According to Romme et al. (2009), while historical fires did not thin piñon-juniper woodlands from below, stand-replacing fire was the main source of juniper mortality. Recovery from such severe fires could take hundreds of years (Romme et al. 2009), creating a multi-century fire-return interval.

Spread of Piñon-Juniper Into Sagebrush and Grasslands and Fire Effects

Disturbance characteristics, especially the frequency and intensity of fire, influence spatial arrangement of woodlands in a landscape of sagebrush and the distribution of tree sizes within these woodlands (Tausch and Hood 2007). Throughout the Great Basin, juniper woodlands are expanding into adjacent sagebrush ecosystems and increasing in density (Bradley 2010; Burkhardt and Tisdale 1976; Miller and Rose 1999; Tausch et al. 1981). For example, one study in Central Nevada determined that 20 percent of the ecotonal area between sagebrush ecosystems and piñon-juniper woodlands showed juniper woodland expansion during a 20-year (1986–2005) period (Bradley and Fleishman 2008b). Although sagebrush shrublands are adapted to many different temperature conditions (Bradley 2010), piñon-juniper woodlands can compete successfully for water at greater depths (Leffler and Caldwell 2005 as cited in Bradley 2010). The expansion of piñon-juniper woodlands is largely the result of cattle and sheep grazing that resulted in the removal of most ground flora that might compete with piñon-juniper (Bradley and Fleishman 2008a; Miller and Rose 1995; but see Vankat 2013), concurrent suppression of wildfires (Bradley and Fleishman 2008a), and late 19th century/early 20th century wet conditions that greatly facilitated establishment and growth of trees (Antevs and Wright 1938; Tausch and Hood 2007).

The pattern of fuel loading and fire behavior changed from fine-fuel dominated open woodlands to one where the tree species crowded out the grasses. Fuel continuity across a landscape increases as the tree biomass increases, thus increasing the possibility of severe, stand-replacing fires. Postfire composition and structure depend upon fire severity, the prefire composition, and the amount of tree dominance, but severe crown fires increase the probability that post-burn sites will be devoid of woody vegetation (Miller and Tausch 2001).

Fires were less likely to reset the ecological succession of any preexisting ground flora on the site, however (Tausch and West 1988). In a comparison of burned and unburned piñon–juniper woodlands in Nevada and California, Koniak (1985) concluded that species that became dominant in mid-successional and later-successional stages were present at early stages; tree species that were eliminated by burning were the only exception. Early-successional species, which bear seeds that survive fire or have buds at root crowns, take advantage of increased availability of water, light, and nutrients (Tausch and West 1988). Annual and perennial forbs tend to dominate early-successional sites, shrubs and annual grasses the mid-successional sites, and shrubs, annual grasses, and trees the late-successional sites (Koniak 1985). Root-sprouting shrubs and forbs were found more on northerly and easterly aspects, annual forbs on southerly and westerly aspects, and seed-germinated shrubs on northerly, easterly, and westerly aspects (Tausch and West 1988). As the tree species begin to dominate the site, fewer resources are

available to understory species and their percentage cover declines (Koniak 1985; Tausch and Hood 2007).

Approaches to Forest Management

Ecological restoration focuses on reestablishing "the structure, function and integrity of indigenous ecosystems" qualified by "to the extent possible" (SER 1993) (fig. 2.1.12). Higgs (1997) refers to ecological fidelity as an endpoint, where restored plant structure and composition provide functional success and durability. He also refers to "mutually beneficial human-wildland interactions," suggesting that true sustainability accepts that human beings are part of the landscape. Given that the ability to adapt to disturbance is an indicator of a healthy ecosystem, successful ecological restoration should seek to restore resilience and adaptive capacity (North et al. 2009b).

Overview

Ecological Forest Management

Silviculture is the art and science of manipulating forest composition and structure to achieve goals (Helms 1998). Silviculture is not an end, but rather a tool or series of tools intended to meet varied objectives, including timber production, wildlife habitat, or even "naturalness" (O'Hara 2016). O'Hara identifies current forest management trends that advertise being "natural" by purporting to mimic natural processes or minimize disturbance effects, often incorporating labels such as "nature" or "ecological" to broadcast their bona fides. Yet, forestry has historically focused on managing stands to attain objectives that could not otherwise be achieved by natural processes (O'Hara 2002, 2016). Proponents of "nature-emulating silviculture" fail to recognize that the cumulative effect of changing climate, historical management and mismanagement, and invasive plants, insects, and diseases are creating novel ecosystems that require novel management practices (O'Hara 2016). For example, in their study of historic (1911) and current conditions of a series of forests in Southcentral Oregon, Hagmann et al. (2013) observed that current and projected climates are hotter and drier than the period when the inventoried trees established and grew. While ecosystem responses to past disturbances and management activities can inform managers (Moore et al. 1999; Youngblood et al. 2004), managing to recreate an environment that will never occur again is likely to

Figure 2.1.12—Restoring forest and woodland ecosystems, such as these on the top of the Warner Mountains in the Modoc National Forest, requires an understanding of "the structure, function, and integrity of indigenous ecosystems" (SER 1993), but also some expectation of potential future disturbances, both long- and short-term (photo by Chris Bielecki, Forest Service).

be unsuccessful (Millar et al. 2007) and such historical responses would not necessarily predict what happens after future management actions (O'Hara 2016).

Restoration

Planning for future forests is challenging for today's managers as the future combination of environment, structure, climate conditions, and other influences will produce novel environments (Clark et al. 2016). The stands today are often no longer at their optimal condition due to years of fire suppression and inconsistent management goals and activities (Graham and Jain 2005). Nonetheless, we can obtain guidance from current and past conditions (Heyerdahl et al. 2001) and management practices that achieve composition and structure goals now can serve future managers with similar goals (Helms and Tappeiner 1996; O'Hara 2016).

North et al. (2009b) cite a definition of ecological restoration from the Forest Service Manual (2020.5): "The process of assisting the recovery of resilience and adaptive capacity of ecosystems that have been degraded, damaged, or destroyed. Restoration focuses on establishing the composition, structure, pattern, and ecological processes necessary to make terrestrial and aquatic ecosystems sustainable, resilient, and healthy under current and future conditions." Ecological restoration has been described as the process of repairing damage caused by humans to the diversity and dynamics of indigenous ecosystems (Jackson et al. 1995). Practitioners of restoration have traditionally viewed their goal as the recreation of some past ecological state (Hobbs and Cramer 2008), although differences exist in defining success as some combination of restoring structure, function, diversity, and socioeconomic impacts (Wortley et al. 2013). Sometimes realism demands that managers repair the most egregious damage ("rehabilitation"), but not necessarily return to some historic state, or moving toward an alternative land use ("reallocation") (Aronson et al. 1993; Hobbs and Cramer 2008). A measure of success is the evaluation of an ecosystem's or community's "key ecological attribute," a characteristic of a community's biology that, when present, define the community as "healthy" and if absent or changed, presage the degradation or loss of the community sometime in the future (Gonzales and Hoshi 2015). North et al. (2009b) argue that restoration should focus on enhancing resilience, the ability to respond to disturbances in a way that maintains the ecological function of a particular ecosystem. This focus melds the drive for rehabilitation with the perspectives of those (O'Hara 2016) who contend that future novel situations require novel management.

Human activities dramatically changed these forests. As will be explained later in the specific forest-type sections, today's forests in Northeastern California and throughout the Great Basin are denser, with a higher proportion of basal area in smaller trees and more dominated by shade-tolerant species. Existing dry forests have permanently reduced capacity to withstand stressors without undergoing significant change (Noss et al. 2006). Current efforts emphasize restoring "processes that shape systems rather than any particular structure or composition of the past" (Hagmann et al. 2013; Millar et al. 2007; Stephens et al. 2010).

Many authors advocate the recreation of spatial heterogeneity that was produced by variable fire intensities, which not only enhances ecosystem values but also reduces the susceptibility to severe disturbance (Crotteau and Ritchie 2014; Hagmann et al. 2013; Larson et al. 2012). This heterogeneity existed on small and large scales and restoration efforts are expected to mimic them. Larsen (2012) defined two scales of heterogeneity: global, which encompasses a stand or even a forest, and local, which is at the scale of tree clump, open gaps, and single trees. Those advocating restoration often fall into one of two camps: "structural"—those who emphasize first restoring historical stand structure and composition through mechanical thinning, and "functional"—those who prioritize use of ecological processes, such as fire, for restoration (North et al. 2007; Stephenson 1999). Neither of these two philosophies fits all sites, although in the specific case of restoring fire to long-unburned sites, many researchers have recommended structural restoration (Covington et al. 1997; Fiedler et al. 1996; Knapp et al. 2017; Moore et al. 2004; North et al. 2007), so long as the process element is instituted in a timely fashion.

Resistance and Resilience

Millar et al. (2007) argue that management for persistence of a species necessitates taking up a broad ecoregion-based view and not be wedded to a particular location, composition, or population level. This practice gives managers flexibility to target achievable goals. Resistance and resilience are two options in the adaptive strategy of assisting ecosystems to accommodate changes due to climate and its resulting influences on disturbance patterns.

Understanding the definitions of resistance and resilience is essential for crafting management objectives and the practices to achieve them. With expectations of changing climates impacting future forest vigor, composition, and structure (Clark et al. 2016), maintaining or enhancing resistance and resilience is considered an important goal (Millar et al. 2007). A successful outcome assumes that the forest ecosystems exposed to disturbances will continue to provide ecological goods and services (DeRose and Long 2014; Puettmann 2011).

Desiring to make the forest more resistant to climate change is easier said than done, given the complications of projecting climate outcomes, the potential effects of said outcomes on the vegetation community, and the administrative and legal requirements for any management actions on public lands. Faced with a multitude of moving parts, silviculturists can focus on what they do best, which is manipulating composition and structure (DeRose and Long 2014).

Differentiating resistance and resilience revolves around timing or perspective: resistance is the ability of an ecosystem to withstand or influence the disturbance ("before" or "during") and resilience is the ability to recover from the disturbance ("after"). Just like any management action, managers should define the scale of disturbances—stands or landscapes—and develop metrics for analysis accordingly. Defining such a disturbance includes recognizing that, depending upon the severity, both resistance and resilience are impacted (DeRose and Long 2014) and that management actions can enhance resistance and resilience simultaneously (Puettmann 2011).

Several definitions emphasize this difference in perspective. Resistance "(to fire) is the ability of community to remain unchanged when challenged by disturbances" (DeRose and Long 2014; Grimm and Wissel 1997); "forestall(s) impacts and protect(s) highly valued resources" (Millar et al. 2007); is the "capacity of an ecosystem to retain its fundamental structure, processes, and functioning despite stresses, disturbances, or invasive species" (Chambers et al. 2013); and "[s]tand resistance to disturbance as the influence of structure and composition on the severity of the disturbance" (DeRose and Long 2014).

Resilience "is the ability of a forest to survive a wildfire relatively intact," which was usually the case in ponderosa pine ecosystems prior to settlement (Fitzgerald 2005); is "the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity and feedback" (DeRose and Long 2014; Walker et al. 2004); seeks to "improve the capacity of ecosystems to return to desired conditions after disturbances" (Millar et al. 2007); and is "the capacity of an ecosystem to regain its fundamental structure, processes, and functioning when subjected to stress and disturbances" (Chambers et al. 2013). DeRose and Long (2014) define resilience as "the influence of a particular disturbance on structure and composition" and state that in order to devise effective management practices, managers must define measurable attributes that are associated with their desired goals.

Management Activities That Can Promote Restoration

Resistance to fire is enhanced by fuels reduction, lower stand densities, and removing ladder fuels. Other management activities include thinning the stand to increase crown spacing and retaining large trees (Moore et al. 1999). DeRose and Long (2014) referred to strategic placement of area treatments (SPLATS) where 20 percent of the area treated can slow fire by 60 percent.

The multiple uncertainties regarding climate, insect and disease attack, and forest response can be overwhelming. Breaking down the analysis into discrete stages, focusing first on the vegetation influences, followed by attention to disturbance and disturbance influences, and finally on vegetation structure and composition in the context of a specific time horizon may be a prudent approach (DeRose and Long 2014).

Managers are usually expected to deal with immediate and long-term concerns at the same time. While there might be some event that triggers public and management concerns, such as preventing a wildfire, managers often approach the treatment, in this case thinning, with the intent to serve long-term goals, such as increasing a forest stand's ability to respond to potential future drought (DeRose and Long 2014; Lévesque et al. 2014). Silviculture only "buys time" as trees grow and enter a vulnerable stage again. Timely treatments increase resistance but economic and political barriers impede these timely treatments (DeRose and Long 2014).

In their chapter on maintaining and restoring ecosystems in Southern Nevada, Chambers et al. (2013) summarize the range of management actions available to managers seeking to enhance ecosystem resistance and resilience. The authors emphasize a "realistic approach" basing

their goals on the current rather than historical potential of ecosystems to support a given set of ecological conditions. They recognized that these vegetation classifications are fluid, as large-scale tree mortality can cause ecotones to shift and entire vegetation communities and regional disturbance patterns to change. While their recommendations are most appropriate for the Southern Nevada region, their characterization of ecosystems that are similar in character to those of the Lassen and the Modoc bears repeating below (table 2.1.3). It should be noted that Chambers et al. (2013) emphasize resistance strategies, which they term "prevention," to reduce disturbances such as fire and invasive species.

Forests with a long history of fire suppression are a prominent landscape feature of the Western United States. In this situation, the disturbance that DeRose and Long (2014) would identify as a management goal is to reduce the impact of catastrophic wildfire and the strategy to adopt would be resistance. Looking to restore more open stand structures consistent with historic disturbance patterns, Fiedler et al. (1996) advocated a resistance strategy of combining thinning with prescribed burning to reduce susceptibility to wildfire, and a resilience strategy of manipulating growing space to favor tree regeneration and control competition from thin-barked, shade-tolerant species. The authors recognize the different opportunities and constraints of the two treatments—the specificity and ability to remove large trees of harvesting, the efficiency in removing small trees, especially firs, the reduction in fuels and stimulation of seedbeds of prescribed burning—and advocate the judicious combination of both practices.

Table 2.1.3—Resilience and resistance characteristics of the major ecosystem types in Southern Nevada and guidelines for appropriate management actions (excerpted from table 7.2, Chambers et al. 2013).

Ecosystem	Resistance and resilience	Guidelines for appropriate management actions
Mixed-conifer	**Resilience**—Moderate to high. Relatively high precipitation, long growing seasons, and moderate growth and establishment rates. Potential to migrate upslope with climate warming. **Resistance**—Moderate to low. Multiple nonnative invaders adapted to environmental conditions; competition with invaders from established native plants can be high.	**Protection**—Control inappropriate recreational activities and overgrazing; detect and eradicate invasive species. **Prevention**—Warranted to decrease fuel loads, restore understory composition, and decrease invasion. Potential for wildland fire use and prescribed fire where risk of large or high-severity fire is low and fire spread can be controlled, and for tree thinning followed by surface fire or pile burning in wildland urban interface and areas with higher fuel loads. **Restoration**—Warranted following surface disturbance or in areas with insufficient fire-tolerant understory species for site recovery after fire. Seed burial (drilling) or transplanting natives adapted to local site conditions and climate warming preferred.
Piñon and juniper	**Resilience**—Moderate. Moderate precipitation, long growing seasons, moderate to slow growth and establishment. Potential for die-off at lower elevations with climate warming. **Resistance**—Low. Many nonnative invaders adapted to environmental conditions; competition from established shrubs and herbaceous species dependent on site productivity and ecological condition.	**Protection**—Control inappropriate recreational activities and overgrazing; detect and eradicate invasive species; suppress fires at lower elevations and that threaten ecosystem integrity. **Prevention**—Warranted to decrease fuel loads, restore understory composition, and decrease invasion. Focus is on mesic sites in early to intermediate stages of tree expansion, and in moderate to high ecological condition. Potential for wildland fire use and prescribed fire on productive sites at high elevation; mechanical treatments more appropriate on sites with low productivity. **Restoration**—Warranted following surface disturbance and in areas with insufficient fire-tolerant understory species for site recovery after fire. Seed burial (drilling) or transplanting natives adapted to local site conditions and climate warming preferred.

Ponderosa Pine

Restoring Ecosystem Function

To restore ecosystem structure and function, Youngblood et al. (2004) recommended that managers use: (1) reference conditions (such as the ones in their study) as guidelines, but recognize that variability occurs across all situations and thus treatments must be implemented accordingly; (2) keep existing live and dead old-growth ponderosa pines; (3) create and maintain clumps (or at least random distributions) of trees; (4) slowly reintroduce fire to control fuels and create both spatial and vertical heterogeneity; and (5) retain coarse woody debris. The interaction between the spatial patterns in structure and composition of dry forests and fire resistance means that restoring the one improves the other (Hagmann et al. 2013).

Improve Fire Resilience

Focusing more specifically on improving fire resilience in ponderosa pine forests, (Fitzgerald 2005) repeated the principles of: (1) reducing surface fuels, (2) removing ladder fuels, (3) retaining large trees that are more fire resistant, and (4) increasing the size and randomness of spacing between tree crowns, as put forth by Agee (2002).

Reintroducing Fire

Reintroducing fire into long-unburned areas could result in higher-than-desired mortality due to the accumulated surface and ladder fuels; many researchers (Covington et al. 1997; Fiedler et al. 1996; Fitzgerald 2005; Mast et al. 1999; Moore et al. 1999) recommend prior nonfire mechanical treatments, including thinning (fig. 2.1.13). Subsequent fires would then be less likely to escape and managers would be able to control the fuel loads in a safer manner.

Opening up the canopy in such a fashion is likely to change the microclimate of the stand. The resulting drier forest floors and increased within-stand wind speeds could potentially lead to increased fire intensity and spread (Fitzgerald 2005; Weatherspoon 1996), an outcome that may seem to contradict the goal of reducing severe fires. Fitzgerald (2005) points out that once the heavy fuel loads are reduced, such fire patterns can result in a presettlement-like forest structure (fig. 2.1.14).

Figure 2.1.13—Researchers have recommended preburn mechanical fuels treatments, such as this one within a riparian zone on the Modoc National Forest, before reintroducing fire into long-unburned areas. Balancing streamside fuels reduction with riparian zone protection from erosion and high temperatures is a challenging task (photo by James Brogan, Forest Service).

Figure 2.1.14—Prescribed fire can assist in reducing ground-level and ladder fuels, reduce stand density, create snags, such as this one next to Blue Lake on the Modoc National Forest, and restore those forest functions facilitated by fire (photo by John Cichoski, Forest Service).

Benefits of Silviculture

Thinning vs. Fire vs. Mortality

Even without catastrophic fires, untreated, dense stands face other risks. In a long-term study on the Blacks Mountain Experimental Forest (Crotteau and Ritchie 2014), two treatments (high and low structural diversity) were implemented on 12 stands with primarily ponderosa and Jeffrey pines in the overstory. Subsequent to harvest, half of the stands were burned and the treatments were compared to neighboring long-untreated (65 years) stands in a Research Natural Area (RNA). The authors observed that the RNA (burn, no-thin) had a higher mortality rate among large trees (24+ inches/60+ cm in diameter), resulting in a projection of few large trees into the future. Many of those that would remain were deemed to be at high risk to fire and beetle attack, casting their future into doubt as well. Thinned stands displayed less overstory mortality and an increase in overstory structure as ingrowth moved into the canopy. Thinned and burned stands experienced some delayed mortality due to the amount of elapsed time between the thinning and prescribed burning (but still much less than the untreated stands), but the fuels reductions were expected to improve the resilience of the stands to future fires (Crotteau and Ritchie 2014).

Benefits of Prescribed Fire

Prescribed burning can help lower stand density, reduce fuel loads, and generally restore ecosystem function to historically fire-maintained forests. Fires in 1990 and 1994 in the Ishi Wilderness in the Southern Cascades of California promoted ponderosa pine regeneration and demonstrated that even the infrequent use of fire can create presettlement structure and composition (Taylor 2010). Like most pine stands treated with prescribed fire (e.g., Boyer 1987; Crotteau and Ritchie 2014), a negative post-burn effect resulting in reduced stand growth can occur. Nonetheless, the effect of the treatments (burn and thin) decreases in influence over time. Crotteau and Ritchie (2014) suggest that these results support the claim that diverse, multiuse stands can produce as much as even-aged stands. Where fire is not available as a management tool, even logged stands may be more resilient than unburned, unlogged stands that have not been treated for more than 100 years.

Benefits of Thinning

Thinning, by reducing competition and resultant individual tree stress, is often recommended as a tool to restore dense stands to a more resilient state (Daniel et al. 1979; Nyland 2016; Smith et al. 1997) (fig. 2.1.15), including those stands that have been unburned and otherwise unmanaged (Covington et al. 1997). The remaining trees can then take advantage of the released growing space. Examples of this strategy are found throughout the region. Stands with high basal area and tree density, factors identified as negatively influencing diameter growth, are highly prone to lethal beetle attacks. Thinning of young eastside ponderosa pine stands in the 1960s and 1970s reduced the basal area to a level where remaining trees experienced less stress and were subsequently less vulnerable to insect attack (Fiddler et al. 1989). Conversely, researchers have related increased radial growth in thinned stands to lower beetle-caused

Figure 2.1.15—This view north of Parker Creek looking south toward the South Warner Wilderness, Modoc National Forest, is an example of the open nature of ponderosa pine and western juniper woodlands (photo by Ken Sandusky, Forest Service).

mortality levels in ponderosa pine stands in Oregon and South Dakota and higher levels of resin flow from wounds in Arizona (Kolb et al. 2007). Large trees may be at risk of mortality even without reintroduction of fire, and thinning may increase growing space per tree and increase survival of the large trees (Crotteau and Ritchie 2014). Kolb et al. (2007) concluded that "judicious thinning" could improve the health of residual large trees.

Forest Health Benefits

Forests with high density force trees to compete with each other for light, water, and nutrients (Oliver and Larson 1996) and thus have fewer resources available to defend against insects and disease. Accordingly, management that results in large increases in radial stem growth rate of ponderosa pine can also increase resistance to bark beetle attacks (Kolb et al. 2007). In Northern New Mexico, a study of managed second-growth ponderosa pine stands found that the thinned stands displayed characteristics of greater resilience, such as maintaining growth during drought and recovering after drought, and greater mean tree size and diameter growth, than stands that had not been treated (Thomas and Waring 2015). The authors concluded that even faced with severe drought stress, trees in thinned stands still had enough photosynthate to allocate to diameter growth. These stands with lower density and higher individual tree size were also believed to be more resistant to attack by insects, such as bark beetles. See Chapter 6.1, Wright, this synthesis, *Ecological Disturbance in the Context of a Changing Climate: Implications for Land Management in Northeastern California* for more discussion about bark beetles.

Importance of Large Trees

Restoration treatments often target retaining and developing large-diameter, fire-resistant pines, usually by thinning from below or broadcast burning, or a combination thereof, to recreate perceived historical stand structure and promote floral and faunal diversity (Crotteau and Ritchie 2014; Youngblood et al. 2004, and see Chapter 4.1, Hanberry and Dumroese, this synthesis, *Biodiversity and Representative Species in Dry Pine Forests*). Although the focus on large trees may underestimate the ecological relevance of smaller trees, whose presence can be hard to reconstruct (Fulé et al. 1997; Hagmann et al. 2013; Larson et al. 2012; Mast et al. 1999; North et al. 2007; but see Moore et al. 2004), large old pines provide the structure (or "backbone") of ponderosa pine and mixed-conifer forests (Franklin and Johnson 2012).

North et al. (2007) advise managers desiring to restore a presettlement forest structure after a period of suppression not to retain too many small trees. Some management recommendations require that all prefire suppression trees be retained in order to recreate historical structure. However, these trees grew up in a suppression environment, and retaining all of them may result in a denser forest than would have occurred under a normal fire regime (Abella et al. 2006; Larson et al. 2012).

Juniper Woodlands

Historically, piñon-juniper ecosystems have not been consistently or sustainably managed. More emphasis has been placed on removing woodland overstory to create or improve forage for livestock (Gottfried and Severson 1994; Tausch 1999). Thus, not a great deal of information is available for developing silvicultural prescriptions specific to the type. Ellenwood (1994) recommended two-step shelterwood and single-tree selection treatments which retained tree cover and sustained productivity. It is challenging, however, to manage this type economically as the market for small-diameter products is limited (Gottfried 2004). The selection method can reduce stand density while maintaining multiple ages and horizontal and vertical diversity. Managers can also retain trees in openings, but over the long term must decide whether the ultimate purpose of the openings is forage or tree regeneration. Openings intended for tree regeneration could temporarily benefit livestock and wild ungulates.

It would require large decreases in overstory to benefit livestock forage, but individual species of ground vegetation may respond differently (Gottfried and Severson 1994).

Restoration of juniper ecosystems to sagebrush benefit from a landscape-level focus (Tausch and Hood 2007), and include short-term and long-term components. In the near term, the principal risk to sagebrush ecosystems is from land use, whereas long-term risks might be equally derived from land use and climate change [effects] (Bradley 2010). Once trees are established, removal by harvesting, chaining, or fire is the principal tactic. Prescribed fire can be used to control tree establishment; however, the outcome depends upon the composition and structure present before the burning. Plant response after removal of western juniper depends on the seed pools in the soil, initial plant communities, soils, size of openings, and later management activities (Miller and Wigand 1994).

In their west-wide study of piñon-juniper ecosystems, Miller et al. (2008) recommended proactive management to reduce piñon-juniper woodlands over at least 50 percent of the study area. Failure to do so will result in the continued decrease in the historic sagebrush landscape. If the sagebrush component is still present, prescribed burning is more likely to be successful. Chaining can result in a 4- to 7-fold increase in biomass of herbaceous species, but the site could revert to a tree-dominated state within 25 years.

The choice of method requires calculating tradeoffs, especially in areas associated with greater sage-grouse (see Chapter 4.3, Dumroese, this synthesis, S*agebrush Rangelands and Greater Sage-grouse in Northeastern California*). Thinning will reduce basal area with less subsequent erosion, but chaining exposes more mineral soil to regeneration and is considerably cheaper. Seeding is necessary on sites with few understory plants or a limited seedbank; such sites are vulnerable to entry by invasive weeds or grasses (Tausch and Hood 2007). A vigorous juniper regeneration cohort can prove problematic in such a situation, however. In his study in Central Oregon, Eddleman (1986) found a large number of western juniper trees located under existing sagebrush or large juniper trees in the overstory. Mechanical clearing by itself would only serve to release the juniper regeneration. Management activities, such as post-treatment burning, may be necessary to prevent reversion to the previous woodland state.

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