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Natural Range of Variation for Red Fir and Subalpine Forests in Northwestern California and Southwestern Oregon

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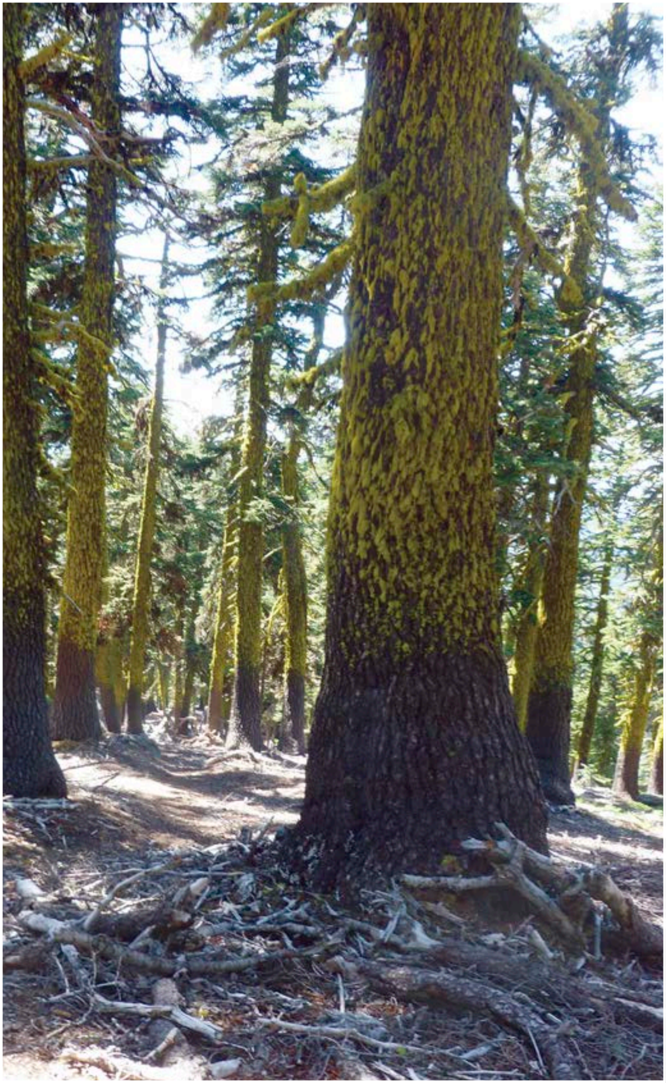

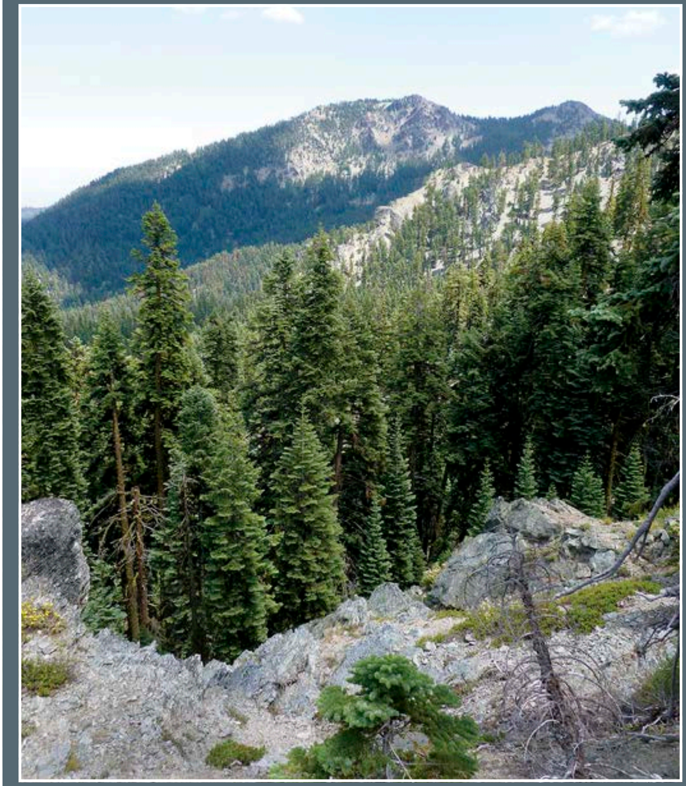

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Cover: Red fir forest in the Yolla Bolly–Middle Eel Wilderness, between Black Rock and north Yolla Bolly Mountain on the Shasta-Trinity National Forest. Photos courtesy of Julie A. Kierstead and Gabrielle Bohlman.

Abstract

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This assessment uses historical observations and datasets, as well as studies conducted in contemporary reference landscapes (i.e., those with active fire regimes and minimal management impacts) to define the natural range of variation (NRV) for red fir (*Abies magnifica*) and subalpine forests in northwestern California and southwestern Oregon. Comparisons between historical and current conditions in the assessment area suggest that many of the ecological attributes of red fir and subalpine forests are within the NRV; however, some conditions and processes are also approaching or are projected to be outside of the estimated limits of the NRV in the near future.

In contemporary red fir forests, the absence of fire for much of the 20th century has lengthened the fire return interval and fire rotation. Future climate and wildfire projections also indicate an extension of the fire season and an increase in fire size and severity in these forest types. Recent increases in insect- and disease-related tree mortality in red fir forests suggest that the scale and intensity of native insect and pathogen infestations may be increasing to levels outside of the NRV. Red fir forests have experienced an increase in tree densities, particularly in the smallest size classes, and concurrent decreases in the density of large-diameter red fir trees. Simplification of forest structure has occurred at both the stand and landscape scales, with shifts away from a heterogeneous, partially open canopy structure to one characterized by more continuous closed-canopy conditions. These alterations have likely resulted from historical logging, long periods of fire exclusion, and changes in climate.

Subalpine forests of the assessment area are generally within the NRV with respect to function, structure, and species composition. Exceptions include potential changes in forest structure, such as increases in the density of small trees and decreases in large-diameter trees. Fire frequency, fire severity, and insect- and disease-related mortality are also projected to increase and likely exceed the NRV with predicted climate change.

Climate envelope models consistently project a substantial loss in the area currently occupied by red fir and subalpine forests in the assessment area by the end of the 21st century. This suggests that the greatest changes to these high-elevation forest types in coming decades will occur as a direct consequence of climate change and its indirect impact on disturbance intensity and frequency.

Keywords: *Abies magnifica,* red fir, *Tsuga mertensiana,* subalpine, natural range of variation, NRV, climate change, forest ecology, forest management, forest structure, fire regime, California, Oregon.

Preface

Effective land and resource management planning requires recognition that ecosystems are naturally dynamic.

Effective land and resource management planning requires recognition that ecosystems are naturally dynamic. One tool that is increasingly being used to address and apply this concept is a natural range of variation (NRV) assessment. These assessments identify a range of ecosystem conditions that fluctuate, over long time periods and across multiple spatial scales, in response to changes in climate, disturbance regimes, and human influence (Wiens et al. 2012). They can be used by land managers and interested stakeholders as a foundation for the development of broad, science-based management goals and desired conditions.

The National Forest Management Act of 1976 requires the U.S. Department of Agriculture, Forest Service (U.S. Forest Service) to develop and maintain land and resource management plans for every national forest and grassland managed by the agency. The process for development, amendment, and revision of land and resource management plans is guided by a series of planning regulations known as the planning rule. The most recent planning rule (2012) emphasizes concepts of sustainability and ecological integrity and links these concepts to the NRV. The Forest Service defines NRV (FSH 1909.12) (USDA FS 2015a) as:

> The variation of ecological characteristics and processes over scales of time and space that are appropriate for a given management application. In contrast to the generality of historical ecology, the NRV concept focuses on a distilled subset of past ecological knowledge developed for use by resource managers; it represents an explicit effort to incorporate a past perspective into management and conservation decisions (adapted from Wiens et al. 2012). The pre-European influenced reference period considered should be sufficiently long, often several centuries, to include the full range of variation produced by dominant natural disturbance regimes such as fire and flooding and should also include short-term variation and cycles in climate. The NRV is a tool for assessing the ecological integrity and does not necessarily constitute a management target or desired condition. The NRV can help identify key structural, functional, compositional, and connectivity characteristics, for which plan components may be important for either maintenance or restoration of such ecological conditions.

This report is one in a series of NRV documents, completed by the U.S. Forest Service Pacific Southwest Region Ecology Program, to support development of Forest Service land and resource management plans. Each assessment in the series focuses on one or more of the major terrestrial vegetation types found on Forest Service lands within two large geographic regions. This assessment focuses on red fir (*Abies magnifica*) and subalpine forest types in northwestern California and southwestern Oregon. This area encompasses portions of the southern Cascade Range of Oregon and California, the north Coast Range of California, and the Klamath and Siskiyou Mountains that stretch from the north Coast Range inland to the Cascade (see fig. 1.1 on p. 2). Meyer and North (2019) provide an assessment of red fir and subalpine forests in the Sierra Nevada bioregion, which includes the Sierra Nevada Range, southern Cascade Range of California, the Warner Mountains, and the White and Inyo Mountains.

Note that while these assessments necessarily rely on historical information to develop NRV estimates, information from the past should not necessarily be used to return managed landscapes to a single static preexisting condition (Wiens et al. 2012). Instead, understanding of the NRV can be used to identify ecological conditions that may improve the resilience of managed ecosystems in the face of future climatic shifts and associated changes in disturbance regimes.

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Chapter 1: Introduction

Natural range of variation (NRV) assessments provide baseline information on ecosystem conditions that, when compared to current conditions, can be used to determine temporal trends and the degree of departure from an ecosystem's natural or reference condition (Landres et al. 1999, Morgan et al. 1994). In this document, we describe the spatial and temporal variation in red fir and subalpine forest ecosystem attributes in presettlement and contemporary reference landscapes. We focus on three broad ecosystem categories: function, structure, and composition. Within each of these categories, we identify ecological indicators (e.g., fire frequency, tree density, species diversity, etc.) and estimate a range of values from within the selected NRV reference period. To assess departure from NRV, we compare the range of values from the NRV period (or contemporary reference sites) to current, and whenever possible, projected future conditions.

Assessment Scope

This assessment focuses on red fir and subalpine forests in the northwestern California and southwestern Oregon assessment area (hereafter, "the assessment area"). This area encompasses portions of the southern Cascade Range of Oregon and California, the north Coast Range of California, and a complex of mountains referred to as the Klamath and Siskiyou Mountains that stretch from the north Coast Range inland to the Cascades (fig. 1.1). Geology in the assessment area is very diverse and plays a major role in influencing the distribution, structure, and composition of red fir and subalpine forest types throughout the assessment area (Sawyer 2006, Whittaker 1960).

Within the assessment area, red fir and subalpine forest types are found in the upper elevations of the montane forest zone. Unlike red fir and subalpine forests in the Sierra Nevada bioregion, which occur in more continuous montane and subalpine belts, these forest types are often found in isolated patches owing to the limited amount of high-elevation habitat in the region.

Although distinct, red fir and subalpine forests share some similar characteristics, including deep and long-lasting snowpack, relatively short growing seasons, and longer and more variable disturbance regimes than lower elevation forest types in the region. They have been influenced in the past by similar environmental (e.g., climatic) and anthropogenic factors, and are projected to exhibit sensitivity to future changes in climate. Based on these similarities, we have combined the discussion of red fir and subalpine forest types in several sections of this assessment. However, we also recognize that these high-elevation forest types possess many characteristics that make them ecologically distinct. To address these differences, we assess departure from NRV for red fir and subalpine forests in separate sections.

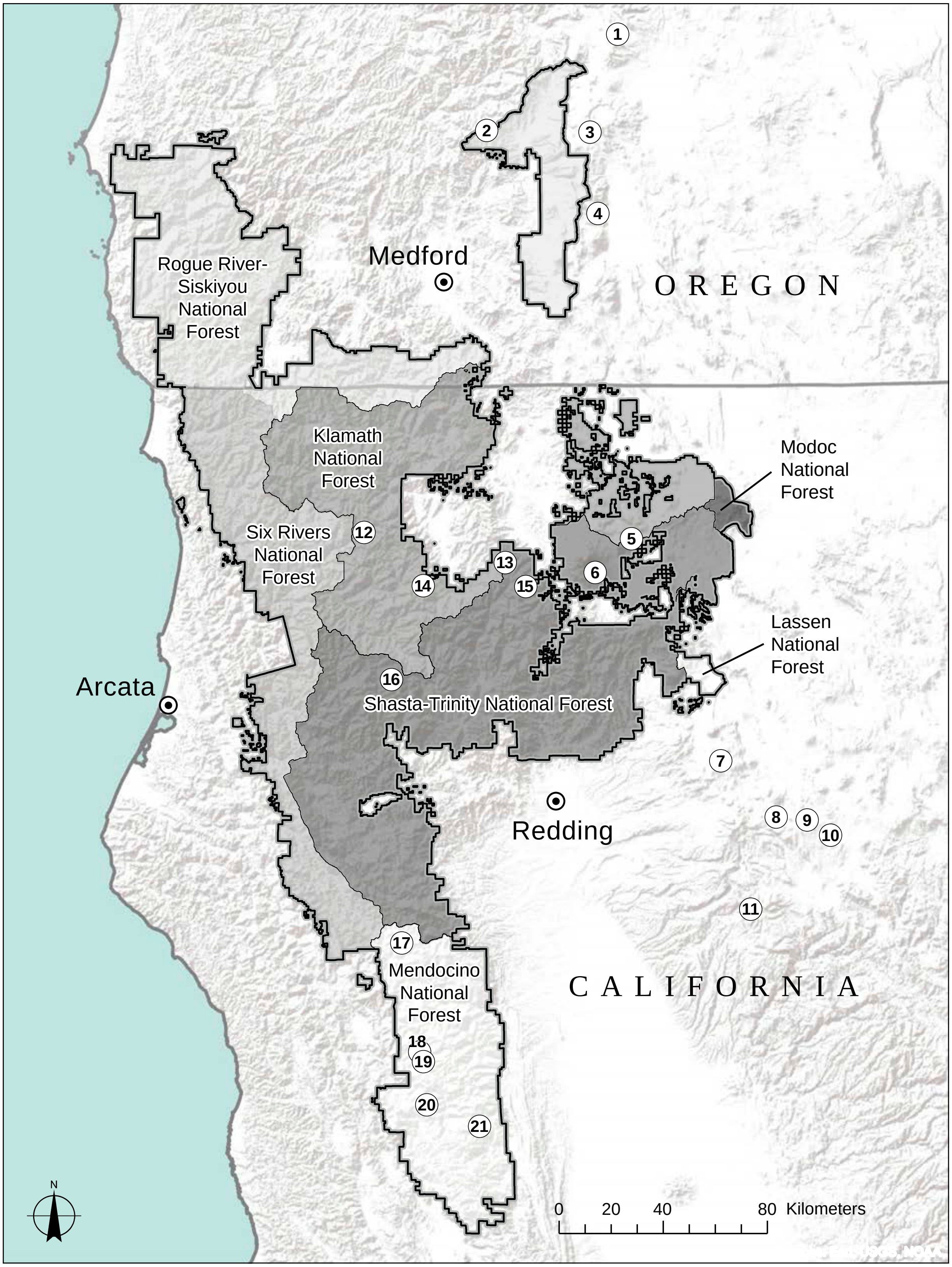

Figure 1.1—Northwestern California and southwestern Oregon assessment area. Numbered circles correspond to contemporary reference sites referred to in this assessment and listed by Map ID in table 2.1.

Historical Influences

Holocene Forest Development

Climatic variability throughout the Holocene Epoch, which began about 12,000 years before present (YBP), had a major effect on the distribution and composition of red fir and subalpine forests within the assessment area (table 1.1). Over this long time period, changes in temperature and precipitation, as well as biotic responses, were not synchronous or uniform across the assessment area (Briles et al. 2005). Nevertheless, several useful generalizations about the response of high-elevation forests, such as red fir and subalpine forests, to climate change can be made:

- The abundance of red fir and subalpine forests generally increased in response to cooler, wetter conditions and decreased in response to warming and drying.
- Dominant mid- to high-elevation tree species, such as red fir and mountain hemlock (*Tsuga mertensiana*), were found at lower elevations (compared to present day) during periods of cooler, wetter conditions and at higher elevations during warmer, drier periods.
- Small-scale heterogeneity in topography and substrate within the assessment area likely provided enough environmental variability for persistence under varying climatic conditions and facilitated rapid responses to both long-term (i.e., millennial) and short-term (i.e., centennial) changes in climate.
- This heterogeneity also resulted in some areas that were relatively buffered from changes in climate (i.e., climate refugia).
- Fire was an important process in high-elevation forests throughout the Holocene, and fire activity varied in response to climatic conditions, generally increasing with warmer and drier conditions.

The abundance of red fir and subalpine forests generally increased in response to cooler, wetter conditions and decreased in response to warming and drying.

The following sections summarize information from several paleoecological or palynology studies conducted within the assessment area to broadly outline the response of red fir and subalpine forests (collectively referred to hereafter as high-elevation forests) to changing climatic conditions over the past 12,000 years.

Early Holocene—

The early Holocene stretches from about 12,000 to 8,000 YBP (table 1.1). Although it was generally considered a period of postglacial warming, it was also characterized by greater seasonal variability, with warmer, drier summers and cooler, wetter winters than present (Mohr et al. 2000). Elevations that currently support high-elevation forests experienced a period of structural change during this time period, with vegetation shifting from subalpine parkland to relatively closed forests that included red fir and mountain hemlock (Briles et al. 2011, Mohr et al.

Table 1.1—General overview of climate, vegetation, and environmental conditions during the Holocene in the higher elevation forests of the assessment area

Period	Time period (YBP)[a]	Climate conditions	Vegetation and environmental changes
Early Holocene	12,000 to 8,000	Warmer, drier summers and cooler, wetter winters than present	• Development of more closed forests of mesophytic species, including Shasta red fir and mountain hemlock, at elevations that presently support high-elevation forest types • Shasta red fir, white fir, and mountain hemlock migrate upslope (still at lower elevations than present day) • Increasing fire activity throughout
Mid-Holocene Xerothermic[b]	8,000 to 4,000	Warmer and episodically drier than the early Holocene and present	• Declines in mesophytic conifer species, such as fir and mountain hemlock at elevations that presently support high-elevation forest types • Red and white fir, mountain hemlock, and subalpine conifers (whitebark pine, lodgepole pine) were likely rare and restricted to more mesic sites • Substantial increase in the frequency of fire events
Late Holocene	4,000 to present	Relatively cooler and often wetter than before	• Increases in mesophytic species, including red and white fir, at elevations that presently support high-elevation forest types • The upper and elevational limits of red fir, mountain hemlock, whitebark pine, lodgepole pine, and other subalpine conifers shift downslope • Decreased fire activity
Medieval Warm Periods[b]	1050 to 850 750 to 600 (900 to 1100 CE 1200 to 1350 CE)	Warmer and drier	• Modest increase in fire frequencies • Coast and Cascade Ranges may have become slightly drier • Increased red fir in the Klamath Mountains suggests wetter conditions
Little Ice Age	550 to 70 (1400 to 1880 CE)	Cooler and moister	• Mountain hemlock reaches greatest abundance • Decreased fire frequency

YBP = years before present; CE = current era.

[a] Years before present is calculated using 1950 CE as the present.

[b] Periods that may serve as possible analogues for climate in the near future.

Source: Briles et al. 2005, 2008, 2011; Marlon et al. 2006; Mohr et al. 2000; Sea and Whitlock 1995.

2000). In relation to modern species distributions, shifts in species composition in the Klamath Mountains during the early Holocene were equivalent to an 800- to 1000-m shift upward in elevation (Briles et al. 2011). Fire activity increased during this time, perhaps as a result of increasing woody vegetation and fuels development (Briles et al. 2005, Mohr et al. 2000).

Mid-Holocene—

The mid-Holocene, which occurred roughly between 8,000 and 4,000 YBP, is often referred to as the Xerothermic Period. During this time period, climates became much warmer and drier, reaching an optimum around 6,000 YBP. Sites that support high-elevation forests at present were primarily dominated by pines, oaks, and montane shrubs, with a notable lack of fir (table 1.1). Two studies of fossil pollen from lake deposits in the Klamath Mountains (Briles et al. 2011, Mohr et al. 2000) characterized present-day, high-elevation forests as open with lower elevation woodland taxa (*Quercus garryana* and *Juniperus occidentalis*), mid-elevation conifers, and a montane chaparral-type understory dominated by huckleberry oak (*Q. vacciniifolia*). Similarly, West (2004) conducted a study of high-elevation fossil pollen deposits in Lassen Volcanic National Park and found evidence of pine-dominated forests during the early and mid-Holocene. Elevational or latitudinal shifts in red fir and subalpine forests have not been documented within the assessment area for this time period. However, studies suggest that during this period of warmer, drier conditions, red fir and white fir (*Abies concolor*), mountain hemlock, and subalpine conifers such as whitebark pine (*Pinus albicaulis*) were rare and likely confined to more mesic habitats (Briles et al. 2011).

Increased charcoal deposits during the warmer periods of the mid-Holocene indicate an increase in fire activity (Briles et al. 2011, Mohr et al. 2000). High fire-event frequencies have been detected around 8,400 YBP in the Klamath Mountains (16 fire events per 1,000 years) (Mohr et al. 2000) and about 9,200 YBP in the Siskiyou Mountains (8 fire events per 1,000 years) (Briles et al. 2008). Prolonged summer drought and abundant chaparral-type understory may have contributed to this increase in fire incidence (Mohr et al. 2000).

Late Holocene—

At the close of the Xerothermic Period, precipitation gradually increased, and cooler conditions dominated beginning about 4,000 YBP (table 1.1). Coincident with these climate changes, red fir and mountain hemlock increased in abundance and demonstrated downslope movement of their upper and lower elevation limits (Briles et al. 2005, Mohr et al. 2000). An abrupt increase in fir and decline in pine and oak abundance was recorded at sites that currently support high-elevation forests in both the Klamath Mountains (around 4,000 YBP) (Mohr et al. 2000) and the southern Cascades (about 3,100 YBP) (West 2004); this suggests cooling temperatures and increased winter snow depths during this period. The lower elevation limit of whitebark pine, lodgepole pine (*Pinus contorta*), and other subalpine conifers also moved downslope during this relatively recent cooler and wetter period, leading to the formation of contemporary red fir and subalpine forests around 2,000 YBP

(Mohr et al. 2000, Woolfenden 1996). This period of cooler conditions resulted in a general decrease in fire activity. An analysis of high-resolution charcoal data from two lakes in the Klamath Mountains, documented low fire-event frequency values (five to nine events per 1,000 years) around 4,800 YBP (Mohr et al. 2000).

Medieval Warm Period and Little Ice Age—The warm, relatively dry periods that occurred between about 1,050–850 YBP and 750–600 YBP are collectively referred to as the Medieval Warm Period. Paleoecological evidence from the Klamath Mountains and northern Sierra Nevada suggests increases in fire frequencies and shade-intolerant species during this time period (table 1.1) (Beaty and Taylor 2009, Briles et al. 2011, Crawford et al. 2015, Daniels et al. 2005, Marlon et al. 2006, Mohr et al. 2000). In their study of charcoal deposits in high-elevation lakes of the Klamath Mountains, Mohr et al. (2000) recorded some of the highest fire-event frequencies on record (10 fire events per 1,000 years) around 1,000 YBP. This warming period was followed by a shift to cooler temperatures known as the Little Ice Age, initiated apparently by a series of massive volcanic eruptions that caused atmospheric reflection of solar radiation (Miller et al. 2012a). The Little Ice Age lasted from about 1400 to 1880 CE, and the period between 1650 and 1850 was the coolest since the early Holocene (Stine 1996). During this time period, there is evidence of an increase in shade-tolerant taxa, such as red fir and mountain hemlock, and a decrease in fire frequency (Briles et al. 2011, Crawford et al. 2015, Marlon et al. 2006). It is important to note that, despite being considered a climate anomaly, the Little Ice Age is commonly used as a historical reference period for restoration planning across the Western United States.

Evidence from the fossil record indicate that some of the rare subalpine species in the assessment area, such as Brewer spruce *(Picea breweriana*), are relicts of past climate change (Ledig et al. 2012). Prior to the Holocene, Brewer spruce had a much wider distribution, occurring as far east as Idaho and Nevada; however, as the cool, moist forests shrank toward the coast and higher elevations, it became restricted to the Klamath Mountains (Ledig et al. 2012).

Recent Climate Trends

Recent climate trends over the past several decades indicate that temperatures within the assessment area have increased, primarily because of increases in mean minimum temperature (Butz et al. 2015a, Rapacciuolo et al. 2014). In a comparison of historical (1900–1939) and modern (1970–2009) climate trends, Rapacciuolo et al. (2014) reported an increase in mean (+0.18 °C) and minimum (+0.47 °C) temperature and a decrease in maximum (-0.23 °C) temperature across northwestern California. Increases in nighttime temperatures have also been observed at a more local scale. For example, Butz et al. (2015a, 2015b, 2015c, 2015d) analyzed data collected since the early 1900s from 17 weather stations on the Klamath, Mendocino, Shasta-Trinity, and Six Rivers National Forests and found that 50 percent of the stations analyzed experienced significant ($p < 0.05$) increases in mean minimum (i.e., nighttime) temperatures ranging between 0.56 and 2.22 °C. Using weather station data and data produced by the PRISM Climate Group, Daly et al. (2009) analyzed temporal trends and variations in temperature and precipitation at Crater Lake National Park over two time periods: (1) 1895 to 2007, selected to represent more long-term trends, and (2) 1971 to 2007, chosen to represent more recent trends. They found evidence of increasing annual minimum temperatures since the 1970s and increasing frequency of summer drought conditions extending into late summer and early fall (Daly et al. 2009).

Precipitation has also slightly increased in the assessment area over the past several decades; however, this pattern is more variable, with both increases and decreases in annual precipitation apparent depending on location (Rapacciuolo et al. 2014). For example, in the Klamath National Forest, the Trinity and Marble Mountains and the highlands south of Mount Shasta have gotten wetter, while the eastern part of the forest has not changed (Butz et al. 2015a).

In montane forests across the Western United States, rising winter temperatures over the past century have been correlated with decreased snowpack, earlier snowmelt, and a higher proportion of precipitation falling in the form of rain rather than snow (Moser et al. 2009; Mote et al. 2018, 2005; Stewart et al. 2005). In a recent analysis of snow station data from across the Western United States, Mote et al. (2018) found that snowpack declined by about 21 percent since 1915. Earlier work by Mote et al. (2005), comparing data from the 1940s and 1990s, documented decreases in snowpack (i.e., April 1 snow-water equivalent [SWE]) of about 29 percent in the Cascades of Oregon and 2 percent in California. Thorne et al. (2015) used the Basin Characterization Model (BCM), a regional water balance model, to evaluate hydrologic change between historical (1951–1980) and contemporary (1981–2010) time periods across the 5,135 watersheds in California. Their analysis documented declines in mean April 1 snowpack that ranged from -24 mm in northwestern California to -33 mm in the southern Cascade Range. At a more local scale, Butz et al. (2015a, 2015c) compared weather station data collected between the early 1940s and 2009 and documented decreases in annual snowfall of 33 to 89 cm on the Klamath National Forest and 31 to 175 cm on the Shasta-Trinity National Forest. Three of the five stations analyzed also experienced significant increases in temperature (minimum or maximum), while none of the stations showed a significant increase or decrease in precipitation (Butz et al. 2015a, 2015c).

In montane forests across the Western United States, rising winter temperatures over the past century have been correlated with decreased snowpack, earlier snowmelt, and a higher proportion of precipitation falling as rain rather than snow.

Precipitation patterns can influence both short- and long-term trends in snowpack, particularly at higher elevations. Van Kirk and Naman (2008) compared SWE data from historical (1942 to 1976) and contemporary (1977 to 2005) time periods in the Klamath Mountains and found significant decreases in SWE at most sites below 1800 m in elevation. In contrast, they found slight increases in snowpack at some high-elevation sites (i.e., >1800 m), despite increasing temperatures over the two time periods, which they suggest was the result of increases in wintertime precipitation.

Anthropogenic Influences

American Indian Influence

Humans have inhabited the assessment area for at least 12,000 years (Anderson 2018, Whitlock and Knox 2002). Over this period, human population sizes have fluctuated in response to climatically induced changes in available food and water supplies. As a result, the effects of indigenous land use practices in high-elevation habitats also varied over time and across the region, with periods of intense activity interspersed with periods of lower populations and reduced land use (Parker 2002). Evidence of spatial shifts are evident in the assessment area's archeological record, which indicates that tribal hunter-gatherer populations lived and foraged at higher elevations during the mid-Holocene (~8,000 to 4,500 YBP) when climatic conditions were warmer and drier and the forest was more open (Hildebrandt 2007). As conditions became cooler and wetter in the later Holocene (4,500 to 1,500 YBP), indigenous populations shifted to more open, lower elevation valleys, using higher elevation forests primarily for transitory hunting camps (Hildebrandt 2007).

The indigenous people of northwestern California sustained themselves by hunting, fishing, and gathering, which they augmented by protecting, encouraging, and cultivating wild plants (Krober 1975 in Crawford et al. 2015). In the summer months, they journeyed from permanent settlements in lowland areas to montane and subalpine habitats to hunt game; collect plant materials for food, medicine, and basketry; and attend social gatherings and spiritual retreats (Muir 1911, Potter 1998, Turner et al. 2011). Some medicinal plants were preferentially collected from higher elevations because they were considered to be more potent and more effective than those at lower elevations; these included the bark and pitch of subalpine fir (*Abies lasiocarpa*) (called the "medicine tree" in some languages); roots of false hellebore (*Veratrum viride*), devil's-club (*Oplopanax horridus*), Sitka valerian (*Valeriana sitchensis*), and Canby's licorice-root (*Ligusticum canbyi*); and whole plants of mountain dryas (*Dryas drummondii*) and yarrow (*Achillea millefolium*) (Turner et al. 2011). Summer and fall were also the prime seasons for harvesting huckleberries (*Vaccinium* sp.), wı΄la (black tree lichen, *Bryoria fremontii*), mushrooms (Anderson and Lake 2013), and edible seeds from whitebark pine cones (Turner et al. 2011).

Just as they did in lower elevation forests, American Indians employed a variety of management practices to obtain and sustain resources in upper elevation forests, including burning, pruning, selective harvesting, sowing, and weeding (Anderson and Moratto 1996, Turner et al. 2011). One of the most important management tools was fire, which was used to clear brush, create firewood, and promote the growth and productivity of wild plants (Sawyer 2007, Turner et al. 2011). Ethnographic research from an area known as Iwamkani or "mountain with huckleberries" in the Klamath language (situated west of Crater Lake in the southern Cascades of Oregon) provides evidence that the Klamath and possibly other tribal groups actively used and managed subalpine plant communities in the past. Early written accounts of the area's tribes, compiled by members of the Wilkes Expedition in the 1840s, include observations of women igniting the vegetation of ''the prairie & mountain ravines'' (Peale 1841 cited in Turner et al. 2011). American Indians in this area burned subalpine habitats to eliminate competing vegetation, remove dead woody materials, encourage plant growth, keep trail networks open, channel deer along predictable passageways, and as part of a tradition of mountain signals or ceremonial fires (Jacobs, n.d., Sapir 1909 cited in Turner et al. 2011). Small, controlled burns of the subalpine understory, referred to as "cold fires" by contemporary tribal elders, were often done to prohibit large and destructive canopy fires, sometimes referred to as "hot fires" (Turner et al. 2011).

A limited number of historical accounts from the assessment area also provide insight into the role of fire in these systems and the potential link to American Indian management practices. For example, in his observations of the Klamath Mountains, John Leiberg noted:

> It is not possible to state with any degree of certainty the Indian's reasons for firing the forest. Their object in burning the forest at high elevations on the Cascades may have been to provide a growth of grass near their favorite camping places, or to promote the growth of huckleberry brush and blackberry brambles, which often, after fires, cover the ground with a luxuriant and, to the Indian, very valuable and desirable growth [Leiberg 1900: 278].

> The aspect of the forest, its composition, the absence of any large tracts of solid old growth of the species less capable of resisting fire, and the occurrence of veteran trees of red fir, noble fir, white pine, alpine hemlock [mountain hemlock], etc., singly or in small groups scattered through stands of very different species, indicate without any doubt the prevalence of widespread fires throughout this region long before the coming of the white man [Leiberg 1900: 277].

Although it is known that American Indians used fire as a management tool, the spatial extent and ecological importance of this practice makes it difficult to separate prehistorical human from climatic influences on high-elevation forest vegetation and fire regimes (Skinner and Chang 1996). Studies conducted in the Sierra Nevada and southern Cascades suggest that presettlement forest structure likely resulted from the interaction between American Indian burning and climate (Fry and Stephens 2006, Klimaszewski-Patterson et al. 2018, Taylor et al. 2016). Some research suggests that, even if there were high levels of indigenous burning, the red fir and subalpine forests of the region may have been too wet to burn except in times of multiyear drought (Marlon et al. 2006). If this was the case, fluctuations in indigenous populations and associated land use practices may have had a much lower impact on the assessment area's red fir and subalpine forests than on low-elevation, dry forest types (Marlon et al. 2006).

European-American Settlement (1848) to Present

European-Americans began exploring the assessment area in the early 1820s; however, it was not until the discovery of gold in 1848 that human impacts greatly intensified as thousands of miners made their way into the unexplored regions of northwestern California and Oregon (Sawyer 2006, Whittaker 1960). Widespread mining operations, intensive logging, major water diversions, and railroad development led to profound changes to many ecosystems in the region. However, unlike the lower elevation forests in the assessment area, the heavy snows, high elevations, and rugged terrain of high-elevation forests decreased many of the direct impacts associated with early settlement of the region (Laacke and Tappeiner 1996). In his assessment of logging operations in the southern Cascades of Oregon, Leiberg (1900: 276) noted that during a period of intensive logging in the yellow pine forests:

> No logging or lumbering has been done in stands of the [mountain]-hemlock type, owing in part to the high elevations at which the type grows, which renders access to its stands difficult and expensive, and in part to the circumstance that, with the exception of the white pine and portions of the noble-fir growth, the species which compose it are generally considered undesirable and unsaleable.

Aside from some localized areas of harvest associated with construction of the railroad, these forest types were generally too far from early markets for timber harvest and were often not burned or cleared for mining (Laacke and Tappeiner 1996, Oosting and Billings 1943,). In the late 1800s, John Leiberg (1900) noted that fir in the region was used primarily for fencing and fuel, with little to none being used for railroad ties or telegraph poles. It was not until the 1950s that timber harvest in red fir forests became more extensive (Laacke and Tappeiner 1996, Taylor and Skinner 2003).

Early settlers set fires unintentionally (e.g., neglected campfires), as well as intentionally to burn out brush and small trees for grazing, to clear areas for prospecting, and to improve conditions for hunting (Whittaker 1960); however, the extent of these fires in high-elevation forests is unknown. In their study of charcoal data from lake-sediment cores in red fir forests of the Klamath Mountains, Whitlock et al. (2004) noted increased charcoal levels that coincided with the arrival and early activities of European-American settlers, suggesting regional burning at this time. Early observations by forest surveyors such as John Leiberg (1900: 279) of the U.S. Geological Survey also noted human-ignited fires in the southern Cascades in the early days of the settlements:

> The earliest settlers found that burned-over tracts in the timber attracted game; hence the forest was fired. Many of the conflagrations spread from camp fires, which the settlers rarely took the trouble to extinguish when breaking camp. They also set many fires for the purpose of destroying the underbrush to facilitate traveling through the forest. Where roads or trails were constructed, fires were set to help clear the way. The builders of a road up the North Fork of the Rogue River and across the Cascades to the Deschutes River, known as the John Day road, are responsible for large tracts burned on either side of the road.

Aside from these limited observations of fire use by settlers, several studies show that fire frequency began to decline shortly after European-American settlement of the assessment area (Fry and Stephens 2006, Taylor et al. 2016). This decline was due to early 20th century livestock grazing (see discussion below), which reduced fuel loads and continuity, as well as a policy of fire suppression initiated in 1905 with the establishment of the National Forest Reserve System (Fry and Stephens 2006, Taylor et al. 2016). The reduction in fire in the assessment area was noted again by John Leiberg (1900: 277) during his survey of the southern Cascade region:

> As time has passed, the frequency of forest fires in the region has much diminished. This is owing to a variety of causes, chief of which are the numerous fire breaks caused by the earlier burns; the gradual extinction of the game and consequent diminished number of hunting parties and lessened risk from unextinguished camp fires; the acquisition of valuable timber claims by private parties throughout the heavily forested sections and the measure of protection, prompted by self-interest, bestowed on their

> property and incidentally on adjoining areas, and, lastly, the destruction of the humus layer, the chief factor in the spread of forest fires in this region, by the earlier conflagrations and the insufficient accumulations of this material since then to support hot, large, and destructive fires.

Early written accounts such as those by Klamath Indian Agent O.C. Applegate (1899: 310–311 in Turner et al. 2011) also document a purposeful reduction in American Indian burning. In his Annual Report to the Commissioner of Indian Affairs, Applegate noted:

> A police force is maintained in the huckleberry country during this season [summer 1899] to preserve order and prevent the spreading of fires. No party of Indians is permitted to go on these excursions into the forest reserve without being duly instructed as to our game laws and fully impressed as to the importance of preventing the starting of fires. I am quite certain that the destructive fires which annually devastate large areas of our timberlands are not usually traceable to our Indians.

Increasing human populations, combined with wide-scale drought in the 1860s and 1870s, resulted in summer grazing of high-elevation montane meadows (primarily in the red fir and subalpine zones) where livestock could take advantage of abundant forage and water (Kosco and Bartolome 1981). High-elevation ridgelines were often favored as driveways to herd the sheep to summer range in high-elevation meadows (Agee 1994). Forests and meadows at all elevations were affected as cattle and sheep were driven virtually everywhere that forage was available, including remote locations (Kinney 1996, Menke et al. 1996). By the late 19th century, more than 6 million sheep grazed in California (Menke et al. 1996). In 1903, which was considered a typical year, an estimated 103,000 sheep and 7,500 cattle grazed along the backbone of the Siskiyou Mountains between present-day Interstate 5 and U.S. Highway 199 (Atzet and Wheeler 1982). By the early 1900s, sheep grazing was heavily curtailed in the newly established national parks and forest reserves. By 1930, sheep grazing declined significantly and was eventually replaced by cattle, coinciding with an overall decline in livestock grazing through the rest of the 20th century (Menke et al. 1996, Ratliff 1985).

Chapter 2: Methods

Temporal Scale

Natural range of variation (NRV) assessments are greatly constrained by a lack of quantitative historical datasets. Although some data (i.e., fossil pollen and charcoal deposits) provide limited insight into ecological patterns and processes that occurred thousands of years ago, most historical datasets extend only decades, or sometimes a few centuries, into the past.

At the broadest level, the reference period used in this NRV assessment extends from the start of the Holocene, about 12,000 YBP, to shortly after the gold rush era when European-American settlement in the assessment area began in earnest. Evaluation of this long time period allows for an assessment of the patterns and processes that occurred during warmer, drier periods in the past (i.e., the Xerothermic Period) as well as cooler, wetter periods (i.e., the Little Ice Age). Information from warmer, drier periods in the past can provide important insight into potential future conditions, particularly when considering climate projections for the assessment area, which predict much warmer conditions by the end of the current century.

Information from warmer, drier periods in the past can provide important insight into potential future conditions.

Specific NRV time periods (i.e., reference periods) were further refined for each indicator variable based on the availability of historical data as well as the timing and intensity of land use practices on the variable of interest. For example, for many variables related to red fir forest structure, the reference period extends from the early 19th century to the early to mid-20th century. This time period was based on the availability of stand reconstruction data, as well as the timing of timber harvest and widespread grazing in the assessment area (Beesley 1996, McKelvey and Johnston 1992). The specific reference period used to evaluate NRV is provided for each indicator variable in the summary table provided at the end of this report.

Note that the influence of prehistorical human populations on high-elevation forest vegetation and fire regimes throughout the Holocene is difficult to disentangle from the effects owing to climatic variation. Therefore, the NRV for red fir and subalpine forests throughout this long reference period includes the influence of American Indian land use practices, which have been present in forests of the assessment area for thousands of years.

Variables and Information Availability

Several ecological indicator variables lacked sufficient historical information to define NRV based solely on past conditions. Therefore, in some cases, data from contemporary reference sites were used as a surrogate for historical NRV information. Contemporary reference sites for high-elevation forests in the assessment were carefully selected based on their relatively pristine condition (e.g., national parks and wilderness areas), the absence of significant historical legacy impacts (e.g., logging), the recent reintroduction of key ecological processes (e.g., natural fire regimes), and the existence of either short- or long-term research information (e.g., experimental forests, research natural areas, natural reserves) (table 2.1). Much of the published science information on reference conditions in high-elevation forests has been extracted from contemporary reference sites that match these criteria. In some instances, reference information was obtained from a nearby bioregion (e.g., Sierra Nevada or Pacific Northwest), particularly when this information was unavailable for the assessment area.

In addition to contemporary reference sites, historical written accounts provided additional information about the NRV for red fir and subalpine forests in the assessment area. These historical accounts were based on idiosyncratic time periods and were primarily by early explorers, naturalists, geologists, foresters, botanists, and other individuals who recorded their observations in field notes, manuscripts, official reports, books, and other published sources. Although many of these historical accounts often contain an inherent bias and other limitations, they nevertheless offer a unique perspective on the historical conditions of red fir and subalpine forests not captured in other historical information sources.

Table 2.1—Areas within the assessment area that are considered historical or contemporary reference sites, listed by mountain range from north to south[a]

Map ID[b]	Name	Location	Type	Forest type represented[c]	Examples of relevant studies
	North Cascades National Park[d]	Northern Cascades, Washington	Contemporary (very little fire since 1926)	SA	Agee et al. (1990)
	Jefferson Wilderness Area[d]	Southern Cascades, Oregon, Deschutes National Forest	Contemporary (active-fire areas present)	SA	Simon (1991)
1	Mount Thielsen Wilderness[d]	Southern Cascades, Oregon, Umpqua National Forest	Contemporary (unknown fire history)[e]	SA	Goheen et al. (2002)
	Cascade Range Forest Reserve (historical)	Southern Cascades, Oregon	Historical	RF/SA	Leiberg (1900)
2	Huckleberry Lake Special Interest Area	Southern Cascades, Oregon, Umpqua National Forest	Contemporary site (fire excluded)[f]	RF	Dubrasich (2010)
3	Crater Lake National Park	Southern Cascades, Oregon	Contemporary site (fire-excluded and active-fire areas present)[g]	RF/SA	RF: Chappell (1991), Chappell and Agee (1996), McNeil and Zobel (1980) SA: Appleton and St. George (2014), Forrestel (2013), Jules et al. (2017), Murray (2007)
4	Cherry Creek Study Area	Southern Cascades, Oregon	Contemporary site (fire excluded)[f]	RF/SA	Foster (1998)
5	Antelope Creek Lakes Research Natural Area	Southern Cascades, Klamath National Forest	Contemporary site (fire excluded)[e]	RF/SA	Cheng (2004)
6	Red Butte–Red Fir Ridge Research Natural Area	Southern Cascades, Shasta-Trinity National Forest	Contemporary site (fire excluded)[f]	RF	Cheng (2004)
	Mount Shasta	Southern Cascades, Shasta-Trinity National Forest	Historical	SA	Brewer (1930), Muir (1918)
7	Thousand Lakes Wilderness[d]	Southern Cascades, Lassen National Forest	Contemporary site (fire excluded)[e]	RF/SA	Bekker and Taylor (2001, 2010)
8	Lassen Volcanic National Park[d]	Southern Cascades, California	Contemporary site (fire-excluded and active-fire areas present)[h]	RF/SA	Parker (1992, 1993), Pierce and Taylor (2011), Taylor (1990, 1997, 2000), USDA FS (n.d.)
9	Caribou Wilderness[d]	Southern Cascades, Lassen National Forest	Contemporary site (fire excluded)[e]	RF/SA	Taylor and Solem (2001)
10	Swain Mountain Experimental Forest[d]	Southern Cascades, Lassen National Forest	Contemporary site (fire-excluded and active-fire areas present)[i]	RF/SA	Taylor and Halpern (1991, 1993)

Table 2.1—Areas within the assessment area that are considered historical or contemporary reference sites, listed by mountain range from north to south[a] (continued)

Map ID[b]	Name	Location	Type	Forest type represented[c]	Examples of relevant studies
11	Cub Creek Research Natural Area[d]	Southern Cascades, Lassen National Forest	Contemporary site (fire excluded)[f]	RF	Beaty and Taylor (2001), Cheng (2004)
12	Marble Mountain Wilderness	Klamath Mountains, Klamath National Forest	Contemporary site (fire-excluded and active-fire areas present)[h]	RF/SA	Cheng (2004), Eastman (1930s photographs), Regalia (1978), USDA FS (n.d.)
13	Crater Creek Candidate Research Natural Area	Klamath Mountains, Klamath National Forest	Contemporary (fire excluded)	SA	Cheng (2004), Skinner (2003)
14	Russian Wilderness	Klamath Mountains, Klamath National Forest	Contemporary site (fire excluded)[f]	RF/SA	RF: DeSiervo et al. (2018) SA: Eckert and Sawyer (2002)
15	Mount Eddy Research Natural Area	Klamath Mountains, Klamath National Forest	Contemporary (fire excluded)	SA	Cheng (2004)
16	Trinity Alps Wilderness	Klamath Mountains, Shasta-Trinity National Forest	Contemporary site (unknown fire history)[e]	SA	Eckert and Sawyer (2002)
17	Yolla Bolly–Middle Eel Wilderness Area	North Coast Range, Mendocino National Forest	Contemporary site (unknown fire history)[e]	SA	Eckert and Sawyer (2002)
18	Black Butte and Brushy Mountain	North Coast Range, Mendocino National Forest	Contemporary site (unknown fire history)[e]	RF	Selter et al. (1986), Barbour and Woodward (1985)
19	Bald Mountain	North Coast Range, Mendocino National Forest	Contemporary site (unknown fire history)[e]	RF	Barbour and Woodward (1985)
20	Hull Mountain	North Coast Range, Mendocino National Forest	Contemporary site (unknown fire history)[e]	RF	Barbour and Woodward (1985)
21	Snow Mountain Wilderness	North Coast Range, Mendocino National Forest	Contemporary site (unknown fire history)[e]	RF	Barbour and Woodward (1985)

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[a] Many of the sites listed have been influenced by a century or more of fire exclusion. In these areas, natural range of variation (NRV) estimates were generally obtained from historical (pre-suppression) stand reconstructions and fire history studies. Stands that are currently characterized by a fire return interval that is within NRV were categorized as contemporary reference sites. NRV estimates in these areas were often derived directly from current conditions within the stands; this is due to the higher likelihood that these reference sites more accurately reflect historical conditions that were created and maintained by the natural fire regime.

[b] Map IDs correspond to locations displayed in figure 1.1.

[c] Forest types represented: RF = red fir, SA = subalpine.

[d] These areas are outside of the assessment area but are considered in this assessment because of their utility as reference sites for the red fir or subalpine forests in the assessment area.

[e] The authors did not disclose the disturbance history of their study sites.

[f] Study sites had little to no evidence of fires in the 20th century.

[g] Fire was excluded from Crater Lake National Park from 1902 (the year of park establishment) until 1978; between 1978 and 1988, the park implemented a managed wildland fire policy, during which five fires, (ranging from 29 to 760 ha in size) were allowed to burn.

[h] Although fire history studies suggest a dramatic decline in fires after 1902, both Lassen Volcanic National Park and the Marble Mountain Wilderness have areas that have burned at least twice in the past 50 years at low to moderate severity (i.e., considered “active-fire” landscapes).

[i] The two plots sampled at Swain Mountain Experimental Forest had an estimated fire return interval of 18.6 years (last fire date = 1938) and 15.7 years (last fire date = 1945).

Chapter 3: Red Fir Forests

Physical Setting and Geographic Distribution

Geographic Distribution

Red fir (*Abies magnifica*) is found in the upper montane forests of the assessment area, generally above the montane mixed-conifer and below the subalpine forest zones. Unlike red fir forests in the Sierra Nevada bioregion, which occur in more continuous montane and subalpine belts, red fir in the assessment area often occur in isolated patches within the many mountain ranges that dominate the region (Sawyer 2007). These mountains include the higher, southern Cascade Range, which define the easternmost range of red fir forests in the assessment area; the lower north Coast Range that delineates the western range; and the complex of mountains referred to as the Klamath and Siskiyou Mountains that stretch from the north Coast Range inland to the Cascade Range (fig. 3.1) (Whittaker 1960). Within the assessment area, red fir forests often blanket the highest ridges, occurring at elevations that range from about 1600 to 2200 m in the Siskiyou Mountains and southern Cascade Range to 1400 to 1830 m in the Klamath Mountains and north Coast Range (Laacke 1990). Red fir forests can also be found at lower elevations in cool flats where it intermixes with lodgepole pine. Red fir is practically endemic to California (Barbour and Woodward 1985). Within the assessment area, it reaches its northernmost extent in the southern Cascade Range of Oregon. In California, the range of red fir extends southward down the north Coast Range to Snow Mountain in Lake County, where it is restricted to six disjunct mountain peaks (Barbour and Woodward 1985, Selter et al. 1986). In the northern portion of the assessment area, the distribution of red fir stretches eastward into the Klamath Mountains and southern Cascade Range. Within the assessment area, red fir is most abundant in the forested regions west of the Cascade crest (fig. 3.1).

Red fir *(Abies magnifica)* is found in the upper montane forests of the assessment area, generally above the montane mixed-conifer and below the subalpine forest zones.

Varieties—

The two varieties of red fir that occur within the assessment area are Shasta red fir (*A. magnifica* var. *shastensis*) and California red fir (*A. magnifica* var. *magnifica*). Shasta red fir is the most common variety and occurs in the Klamath and Siskiyou Mountains, the north Coast Range, and southern Cascades of northern California and southern Oregon. This variety is generally considered a hybrid between California red fir, which is found in the northern and southern Sierra Nevada mountain range, and the more northern noble fir (*A. procera*) (Lanner 2010). Shasta red fir is differentiated from California red fir by its long, exerted cone bracts, lighter seeds, and fewer cotyledons. Despite these morphological differences, the two varieties of red fir are thought to be similar in terms of their ecological response to environmental variation (Barbour and Woodward 1985). For purpose of this discussion, the two varieties have been combined into one general red fir forest type.

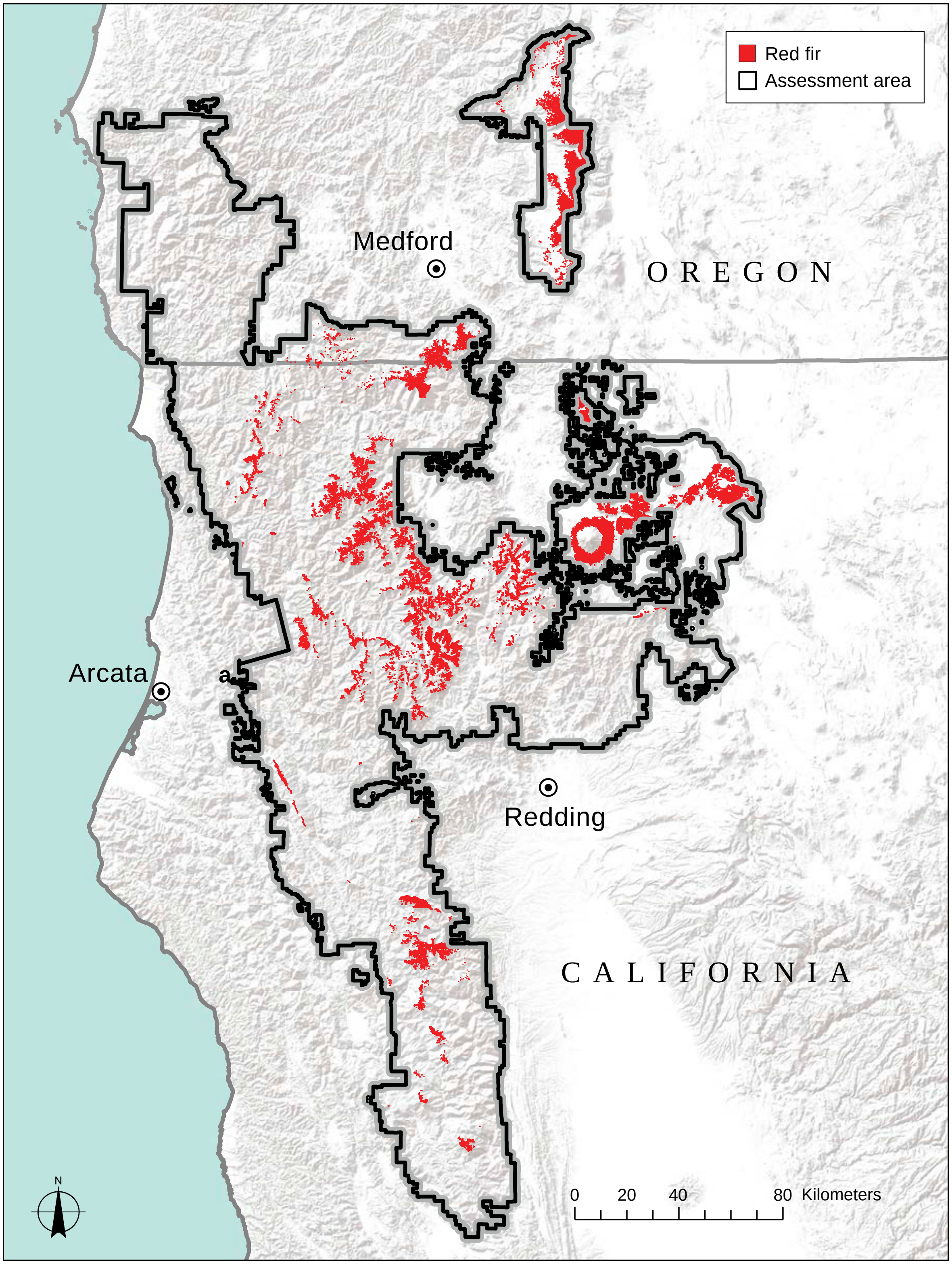


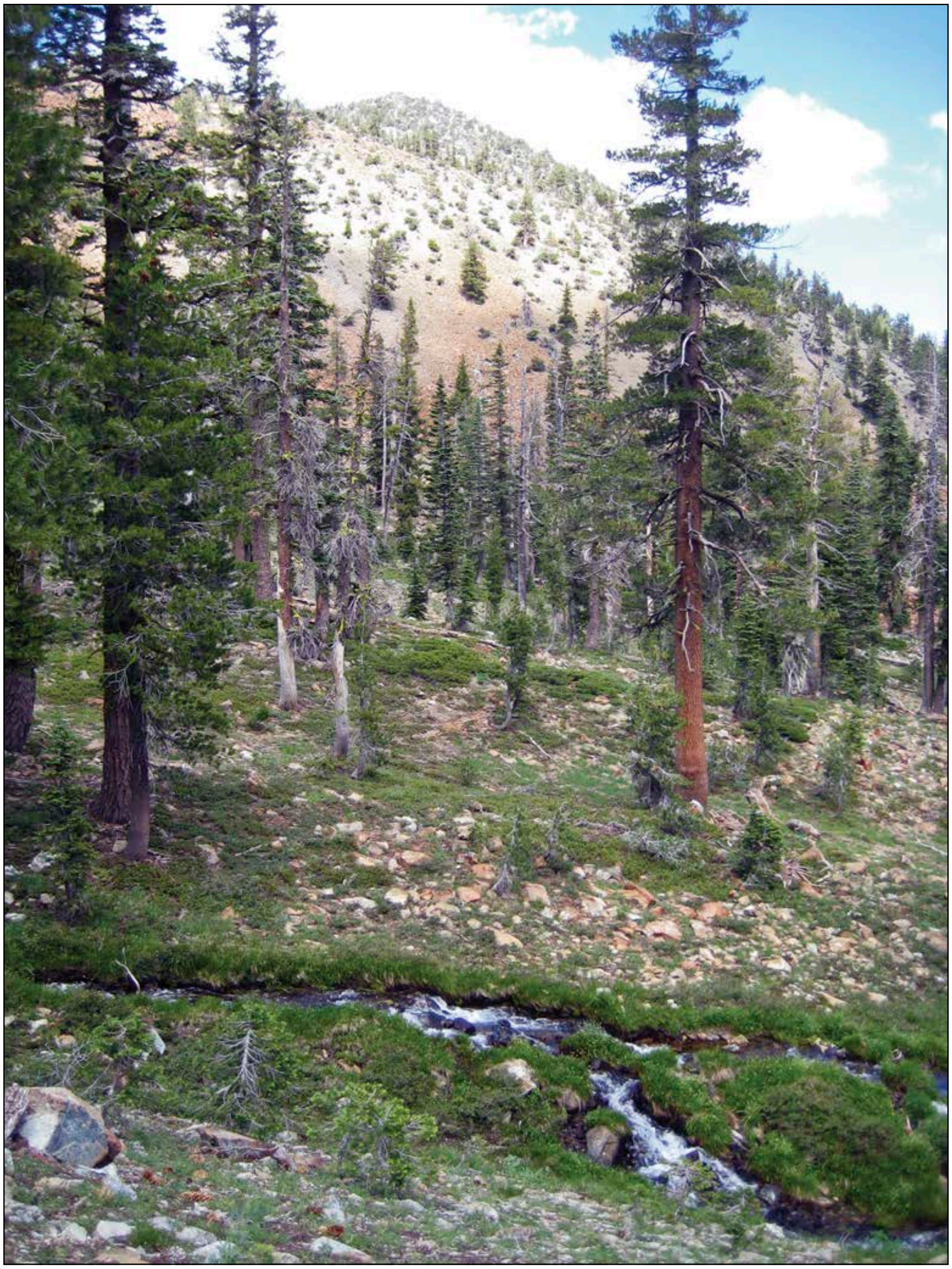
Figure 3.1—Map showing the distribution of red fir forests in the assessment area. Data were obtained from U.S. Department of Agriculture polygons of existing vegetation. Map displays forest types designated as red fir (California Wildlife Habitat Relationship type = RF) in California and red or Shasta red fir (forest type contains *Abies magnifica* or *A. shastensis*) in Oregon.

Climatic Associations

Red fir forests in the assessment area occupy cool sites with substantial winter snow (Agee 1993). The climate where red fir occurs is considered relatively mild for high-elevation forests; summer temperatures only occasionally exceed 29 °C, and winter temperatures rarely fall below -29 °C (Laacke 1990). Summers are characterized by a 4- to 5-month dry period, interrupted by occasional thunderstorms. The majority of precipitation in red fir forests occurs between October and March, with an estimated 80 percent or more falling as snow (Laacke 1990). The distribution and abundance of red fir has been shown to be strongly correlated with long-term mean April snow depth and snow-water equivalent (Barbour et al. 1991). Barbour et al. (1991) found that the lower elevational limit of red fir forests is correlated with the elevation at which precipitation transitions from rain to snow during the winter months. Red fir generally occupies sites that are cooler and snowier than commonly associated species such as white fir, which is consistent with the relatively low drought tolerance of red fir (Lutz et al. 2010). Growth of red fir has been shown to be positively correlated with mean snow depth from the prior winter and spring (Dolanc et al. 2013b), although the influence of these and other climate variables may be dependent on latitude and elevation (Torbenson 2014). In his comparison of forests at similar elevations in Lassen Volcanic and Yosemite National Parks, Parker (1995) noted that northern montane and subalpine forests occupied cooler and moister sites than those in the southern Sierra Nevada. These conditions, combined with higher annual and summer precipitation in these northern forest types, may reduce the severity of moisture deficits during the growing season (Parker 1995).

Geology and Soils

Red fir forests occur on soils derived from a variety of parent materials. Soils are typically classified as frigid or cryic, with minimal horizon development (Laacke 1990). In the Cascades, red fir forests are occasionally found on pumice deposits overlying older soils, while in the Klamath Mountains, red fir can occasionally be found on high-elevation ultramafic soils (fig. 3.2) (Laacke 1990). Tree growth and stand development is generally best on deeper soils, which are often associated with glacial deposits or Pleistocene lake beds, and poorest on steeper slopes with shallow soils (Fowells 1965 in Laacke 1990).



Julie A. Kierstead

Figure 3.2—Red fir and western white pine growing on ultramafic soils near Mount Eddy, Shasta-Trinity National Forest; elevation 2180 m.

Ecological Setting

Indicator Species and Vegetation Classification

Red fir is the defining species for this forest type in the assessment area. Common associates include white fir and lodgepole pine in the lower transition zones and mountain hemlock, lodgepole pine, and western white pine (*Pinus monticola*) in the higher elevation ecotones. In a limited number of high-elevation stands, red fir intermixes with Brewer spruce, Engelmann spruce (*Picea engelmannii*), Pacific silver fir (*A. amabilis*), subalpine fir (*A. lasiocarpa*), whitebark pine, and, to a lesser extent, foxtail pine (*P. balfouriana*) (Franklin and Dyrness 1988). In the northwestern portion of the assessment area, red fir shares dominance with noble fir. On serpentine soils at higher elevations, red fir is occasionally found with foxtail pine, western white pine, and Jeffrey pine (*P. jeffreyi*) (fig. 3.2) (Laacke 1990). Current vegetation classification systems recognize as many as 28 vegetation associations of red fir forest in the California portion of the assessment area (Sawyer et al. 2009) and at least 5 associations in the Siskiyou region of southwestern Oregon (Atzet et al. 1984). All red fir forest stands, including those only partially dominated by red fir (e.g., mixed red fir–western white pine, red fir–white fir, red fir–mountain hemlock), were included in this assessment to capture the full range of variation in red fir associations in the assessment area (fig. 3.3).

Red fir is commonly associated with white fir and lodgepole pine in the lower transition zones and mountain hemlock, lodgepole pine, and western white pine in the higher elevation ecotones.

Ecological Importance of Red Fir

Red fir forests provide a diverse array of ecosystem services, including watershed protection, erosion control, carbon sequestration, and habitat for a diverse array of species. More than 160 vertebrate wildlife species use red fir forests for foraging or nesting/denning habitat, including 8 amphibians, 4 reptiles, 104 birds, and 53 mammals (Barrett 1988). Of the more than 100 species of birds found in California's red fir forests, about half are associated with mature forests (Laacke 1990).

Natural Range of Variation Descriptions

Function

Fire—

Although fire has been an important disturbance process in red fir forests of the assessment area for thousands of years, detailed fire history information from the assessment area is lacking. The difficulty in characterizing past fires stems from the fact that fire scars can grow over, red fir tends to rot after being scarred, and evidence of low-severity fires is often obscured (Agee 1993). Therefore, the discussion below integrates local studies with information from areas adjacent to, but outside of, the assessment area (e.g., southern Cascades of California and Oregon), and occasionally draws from studies conducted in more distant portions of the red fir range (e.g., the Sierra Nevada).

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Figure 3.3—Red fir forests within the assessment area. The two top photographs are red fir forest in the Yolla Bolly–Middle Eel Wilderness, between Black Rock and north Yolla Bolly Mountain on the Shasta-Trinity National Forest, elevation 2151 m. The bottom photograph is red fir and mountain hemlock forest along the Siskiyou Crest, between the Klamath and Rogue River–Siskiyou National Forests, elevation 1920 m.

Fire type—The fire regime in red fir forests of the assessment area is highly variable, with fires exhibiting considerable variation in size, frequency, and severity (Agee 1993). Fires in these upper montane forests were predominantly slow-moving surface fires owing to the presence of compact fuel beds; natural terrain breaks; heavy snowpack; and cool, moist growing conditions (Skinner 2003, Taylor and Skinner 2003). Occasional passive crown fires occurred; however, they were often associated with dry and windy conditions and pockets of fuel created by tree fall, insect-related mortality, avalanches, snow-ice crushing, death of larger trees, and patches of montane chaparral in the understory (Skinner 2003).

Fires in these upper montane forests were predominantly slow-moving surface fires owing to the presence of compact fuel beds; natural terrain breaks; heavy snowpack; and cool, moist growing conditions.

In lower elevation red fir forests, which historically had shorter fire return intervals (FRIs) (see table 3.1), the primary factor limiting fire spread was likely a lack of sufficient fuel to carry a fire. In contrast, the longer FRI in upper and mid-elevation red fir forests generally resulted in sufficient fuel buildup, making fire occurrence dependent mostly on whether climate or weather variables were suitable for ignition and spread (Agee 1993). Taylor et al. (2008) examined the relationship between climate variability and fire extent in upper montane forests of the southern Cascades for the period of 1700 through 1900. They determined that widespread burning was strongly associated with dry and warm conditions in the year of the

Table 3.1—Average historical fire return interval (FRI) estimates for red fir forests in the assessment area[a]

Red fir type/group (aggregation)[b]	Mean FRI[c]	Median FRI[c]	Min. FRI[c]	Max. FRI[c]	No. of estimates	Forest types included
	- - - - - - - - -	- *Years* -	- - - - - - - - -			
High elevation	65	66	13	157	8	Red fir, red fir–western white pine/ mountain hemlock
Mid elevation	41	16	9	95	12	Red fir
Low elevation	25	15	4	56	9	Red fir–white fir
Klamath Mountains	43	20	6	126	9	Red fir, red fir–white fir, red fir–western white pine/mountain hemlock
Southern Cascades	44	41	11	79	20	Red fir, red fir–mixed conifer
Northern Sierra Nevada[d]	51	36	9	71	14	Red fir, red fir–white fir, red fir–western white pine/mountain hemlock
Regional FRI (all studies)	43	35	9	100	29	All
Regional composite FRI	37	27	8	87	19	All
Regional single tree FRI	57	53	10	126	10	All

[a] Individual FRI estimates and sources are presented in table A1.1 of the appendix.

[b] Averages for elevation and regional aggregations were calculated using single-tree and composite samples.

[c] Values reported represent grand means (i.e., the mean of the means presented in the studies assessed).

[d] Values from Meyer and North (2019) and includes the southern Cascades; average values were calculated using single-tree and composite samples.

fire, but that the strength of this relationship also varied over time in response to climatic variation over longer multidecadal time scales (Taylor et al. 2008).

Fire frequency—Historical FRI estimates for red fir forests in the assessment area are also highly variable (tables 3.1 and A1.1); therefore, it is important to consider local factors, such as topography, elevation, geology, and forest type, when making inferences about fire regimes in these forest types. In general, mean and median historical FRI values increased with elevation, and intervals tended to be longer in more mesic, high-elevation red fir forest types (e.g., red fir and mountain hemlock) (table 3.1). Lower fire frequencies in higher elevation forests were likely due to slower fuel accumulations and weather conditions that were less conducive to carrying fire (Skinner 2003).

The only reported FRI estimates for red fir forests specific to the assessment area come from a few studies in the Klamath Mountains. Fire chronologies from fire-scarred tree specimens in four upper montane basins in the Klamath Mountains suggest that presettlement fires were moderately frequent in red fir forests (Skinner 2003, Whitlock et al. 2004). Estimates of fire intervals in these forest types are scale-dependent; for example, Skinner (2003) found that 57 percent of fire dates were found only on one tree, resulting in a single-tree (point) FRI that was much longer (mean: 54.5 years; range: 5.8 to 276 years) than the composite FRI (mean: 20.4 years; range: 3.8 to 104.8 years). Another study of disturbance regimes in red fir forests of the Klamath Mountains estimated return intervals of 40 years (Atzet 1996, Atzet and Martin 1992); however, it is important to note that this return interval estimate encompasses all disturbance types (e.g., wind, disease, humans, etc.) and was derived from an analysis of stand ages rather than direct evidence of fire. At Crater Lake National Park in the southern Cascades of Oregon, Chappell and Agee (1996) estimated a mean composite historical FRI of 39 years (range: 15 to 71 years) (table A1.1), with considerable variability in fire-free periods (range: 15 to 157 years). Less than 100 km south of Crater Lake National Park, Sensenig (2002) calculated mean composite FRIs of 25 years (range: 6 to 89 years) for red fir forest types and a lower mean FRI of 11 years (range 2 to 66 years) for red fir–white fir forest (table A1.1). The mean/median single-tree FRI for red fir forests across all studies in the assessment area was 57/53 years (range: 10 to 126 years) and the mean/median composite FRI was 37/27 years (range: 8 to 87 years) (table 3.1).

At a more local scale, longer FRIs are generally found in areas where fuel accumulations are low and landscape features, such as volcanic outcrops and meadows, are present to break up fuel continuity. FRIs are typically shorter in stands where fuel accumulations are naturally more continuous, such as areas with extensive shrub fields that can promote the spread of fire from lower montane forests into upper montane red fir stands (Skinner and Chang 1996). Landscape position and context can also influence FRI patterns. For example, several studies from red fir forests in the southern Cascades (Bekker and Taylor 2001, Foster 1998, Taylor 2000) have documented a strong correlation between mean point FRI and elevation (i.e., less frequent at higher elevations). In reconstructions of annual area burned in the southern Sierra Nevada, mean and maximum FRI estimates for red fir forests tended to be greater on relatively mesic north-facing slopes (mean/maximum FRI = 30/50 years) compared to xeric south-facing slopes (mean/maximum FRI = 15/25 years) (Caprio and Graber 2000, Caprio and Lineback 2002). However, Taylor (2000) found that median FRI was similar across all slope aspects in red fir–mountain hemlock forests in Lassen Volcanic National Park. In this case, topography and landscape context (i.e., where the stand was in relation to other variables that influence fuel continuity and rate of fire import) likely played a strong role in determining the FRI.

Fire rotation is the time in years required for fire to burn an area that is equivalent to the study area's total extent and can be thought of as a landscape-level FRI (Agee 1993). Studies of presettlement (pre-1905) fire regimes in the Caribou and Thousand Lakes Wilderness Areas on the Lassen National Forest in the southern Cascades estimate that fire rotation within red fir forests varied from 50 to 93 years in lower elevation red fir–white fir forests and 76 to 147 years in higher elevation red fir–mountain hemlock/western white pine forests (table 3.2) (Bekker and Taylor 2001; Taylor 1997, 2000; Taylor and Solem 2001).

Table 3.2—Historical fire rotation estimates for red fir forests in the assessment area

Location	Forest type	Fire rotation[a]	Reference
		Years	
Southern Cascades (Thousand Lakes Wilderness)	Red fir–white fir	50	Bekker and Taylor (2001)
Southern Cascades (Thousand Lakes Wilderness)	Red fir–mountain hemlock	147	Bekker and Taylor (2001)
Southern Cascades (Thousand Lakes Wilderness)	Red fir–white fir	93	Taylor (1997)
Southern Cascades (Caribou Wilderness)	Red fir–white fir	68	Taylor (1997)
Southern Cascades (Caribou Wilderness)	Red fir and other upper montane forests[b]	76	Taylor and Solem (2001)
Southern Cascades (Lassen Volcanic National Park)	Red fir–western white pine	76	Taylor (2000)
Average across studies		85	

[a] Fire rotation is the length of time necessary to burn an area equal to the area or landscape of interest.

[b] Red fir and other upper montane forest types are aggregated for estimation of fire rotation.

Current and future deviation from NRV—Fires were moderately frequent in red fir forests before fire suppression became effective in the early 20th century (Skinner 2003). Since that time, few fires have burned in red fir forests (Bekker and Taylor 2001, Skinner 2003, Whitlock et al. 2004) with the exception of contemporary reference sites with active fire regimes (e.g., Crater Lake National Park). This absence of fire has led to a lengthening in FRI and fire rotation in contemporary red fir forests (Bekker and Taylor 2001). Taylor and Solem (2001) and Taylor (2000) estimated presettlement (1735 to 1849), settlement (1850 to 1904), and fire-suppression (1905 to 1994) fire rotations of 76, 117, and 577 years, respectively, in red fir and other upper montane forests in the southern Cascades. In their analysis of charcoal and tree-ring data from red fir forests in the Klamath Mountains, Whitlock et al. (2004) documented a significant decline in fire that coincided with the intensification of fire suppression efforts in the 1950s. Skinner (2004) noted that the 50-year period without fire in the second half of the 20th century was longer than any other fire-free period in the study area's fire history record.

Analysis of data in the California Fire Return Interval Departure (FRID) database (Safford et al. 2015) indicates that about 69 percent of red fir forests in the assessment area exhibit moderate departure from their presettlement FRI. These areas of moderate departure are burning less frequently than they would have historically and have current FRIs that are at least 1.5 times longer than expected under presettlement conditions. Safford and Van de Water (2014) derived percent FRID (PFRID) to estimate the current percentage of departure from the mean presettlement FRI. Currently, red fir forests in the assessment area have a weighted average PFRID value of 49.5 percent and an average FRI of about 79 years.

Like many western forest types, the absence of fire over the past 50 to 100 years has had an ecological impact on red fir forests in the assessment area, pushing many of these systems outside of their NRV. The mean fire frequency (FRI) in red fir forests of the assessment area is currently outside of the NRV. For example, the regional mean/median FRI for red fir forests in the assessment area is 43/35 years, which is shorter than the length of the fire suppression period (>100 years). The average FRI values presented in table 3.1 suggest that red fir forests at middle and lower elevations (i.e., those in the red fir–white-fir zone) may be at higher risk of departure from the NRV than red fir forests at higher elevations, which typically have longer FRIs. The period of time that fire has been absent from these higher elevation forest types is generally considered to be within the range of the natural fire-free period for red fir forests in the assessment area (Agee 1993, Sensenig 2002, Skinner 2003). Characteristics of these forests, such as slow fuel accumulation and dense and difficult to ignite fuel beds, may help promote conditions that are more similar to presettlement conditions, especially compared to those found in more fuel-limited fire regimes (Steel et al. 2015).

Note that the degree of departure from the NRV is scale dependent. Fire suppression may not have had a large impact at the stand level (i.e., individual red fir stands may still be within the NRV); however, when assessed across the landscape, a greater proportion of red fir forests are toward the higher end of their NRV in natural FRI, resulting in more homogenous landscape conditions (Chappell and Agee 1996).

At the landscape scale, a greater proportion of red fir forests are toward the higher end of their natural range of variation in natural fire return intervals, resulting in more homogenous landscape conditions.

Climate-driven increases in fire activity are already apparent in the Western United States (Westerling et al. 2006). Models specific to California project increased fire activity to persist and possibly accelerate under most future climate scenarios owing to increased growth of fuels under higher carbon dioxide (and in some cases precipitation), decreased fuel moistures from warmer dry season temperatures, and possibly increased thunder cell activity (Lenihan et al. 2003, Miller and Urban 1999, Price and Rind 1994, Westerling et al. 2006). In the Pacific Northwest, longer, hotter, and drier fire seasons are projected under future climate change scenarios, and the area burned by wildfires is projected to increase as a result (Wimberly and Liu 2014). Westerling and Bryant (2008) predict a 10- to 35-percent increase in large-fire risk by midcentury in California and Nevada, and Westerling et al. (2011) project increases in burned area that may exceed four times the current levels in forest lands by the end of the century. The dynamic ecosystem model (MC1) runs reported in Barr et al. (2010) project increases in annual fire area in the Klamath River basin of 11 to 22 percent from current levels of 2.7 to 3.0–3.3 percent by 2100, resulting in as many as 134 000 ha burned in an average year. These projected changes may result in shorter FRIs and fire rotation in red fir forests of the assessment area.

Seasonality—Historically, most fires in red fir forests occurred during the short dry season in late summer or early fall when high densities of lightning strikes were common (Agee 2007). Skinner (2003) estimated that 95 percent of fires in montane and subalpine forests of the Klamath Mountains historically burned in late July through early October. Studies of red fir forests in the southern Cascades documented similar timing. For example, Foster (1998) noted that, although fire scars were present in all seasons, the majority (69 percent) of fires in red fir forests occurred later in the year, coinciding with the dry season in south-central Oregon. In red fir–white fir forests in the Thousand Lakes Wilderness, the position of fire scars within presettlement annual growth rings indicated that 77 percent of historical fires burned during the late summer and fall, and the remaining 23 percent of fires burned during the early to midsummer (Bekker and Taylor 2001). In higher elevation red fir–mountain hemlock and red fir–western white pine stands where snowmelt is later, 99 to 100 percent of historical fires burned during the late summer to fall (Bekker and Taylor 2001, Taylor 2000).

Current and future deviation from NRV—The California Fire Perimeters Database (CDFFP 2015) contains information about fires (>1 ha) that occurred in red fir forests of the assessment area over the past 100 years. Analysis of these data indicate that about 75 percent of the fires recorded during this period occurred from late July to early October. The remainder burned between June and early July (22 percent) or late October to early November (2 percent). Of those fires that occurred in early June, more than half occurred in a single year (2008) during a period of severe drought when one storm ignited numerous fires across the region. As seen in figutre 3.4, this general pattern of fires occurring predominantly between late July and early October, with fewer fires occurring in early summer and late fall, has remained relatively consistent over the past 100 years.

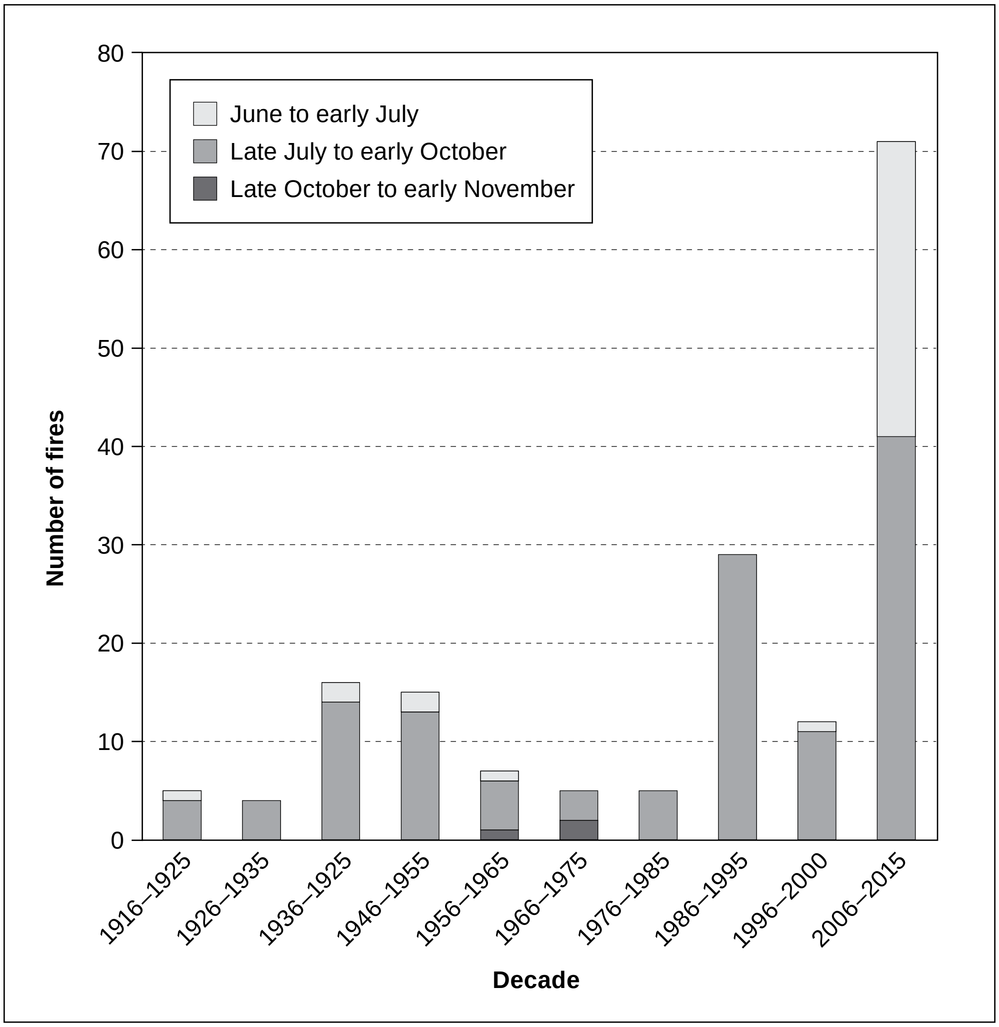


Figure 3.4—Seasonality of fires that have burned in red fir forests in the California portion of the assessment area over the past 100 years. Data on fires >1 ha in size were obtained from the California fire perimeter database (CDFFP 2015).

Although tree-ring reconstructions and contemporary fire data suggest that the seasonality of fires in red fir forests is not currently outside the NRV, climate projections suggest that more fires may occur later and earlier in the season in the future. While overall precipitation has been steady to slightly increasing over the last part of the century, snowpack depth and persistence in the assessment area have been decreasing, primarily because of warming temperatures and decreases in the snow-to-rain ratio (see "Recent Climate Trends"). These patterns result in earlier drying of fuels and extension of the fire season (Westerling et al. 2006). Liu et al. (2010) projected future changes in wildfire potential by modeling the Keetch-Byram Drought Index under a number of future Global Circulation Model (GCM)-based scenarios. Under their projections, fire season becomes a couple of months longer for much of the contiguous United States, including the assessment area, by the end of the 21st century. Flannigan et al. (2013) projected that fire season length would increase by more than 20 days for most of the assessment area by 2050, and all of the assessment area by 2100. Basing their work on 15 GCMs, Yue et al. (2013) projected a median increase of more than 3 weeks in the fire season for the Pacific Northwest, which includes the assessment area, by the middle of the 21st century.

Extent—Limited estimates of fire size suggest that fires burning in red fir forests ranged from small (a few hectares) to large (hundreds of hectares) (Chappell and Agee 1996). Presettlement (1729–1918) mean fire size in the Thousand Lakes Wilderness in the southern Cascades was estimated at 151 ha (range: 34 to 372 ha) in red fir–white fir forest and 140 ha (range: 124 to 155) in red fir–mountain hemlock forest (Bekker and Taylor 2001). In Lassen Volcanic National Park, historical mean fire size was 176 ha (median = 129 ha; range: 11 to 733 ha) in red fir–western white pine forest (Taylor 2000).

In one of the few fire history studies of montane and subalpine forests in the Klamath Mountains, Skinner (2003) noted that the majority of fire dates (89 of 156) were found on a single tree, and none met the criteria for an extensive fire year. Skinner (2003) concluded that larger fires were historically restricted to local basins. Other studies have shown similar results; for example, in his study of two red fir forest stands in the Cascade Range of southern Oregon, Sensenig (2002) found only one tree recording more than one fire in each of the two stands. In south-central Oregon, Foster (1998) also found no evidence of large fires in red fir forests that left evidence in contiguous plots. Farther south in Lassen Volcanic National Park, Taylor (2000) found evidence of some fire years that were coincident among sites and concluded that widespread fire events occurred during years with below-average regional precipitation. Early surveys of forest reserves in the southern Cascades of Oregon in the 1900s noted that before 1855, fires were generally small, with only limited evidence of larger ones (Leiberg 1900).

Fires in presettlement red fir forests were generally small (i.e., less than 4 ha).

Data from contemporary reference sites in the southern Sierra Nevada also suggest that fires in presettlement red fir forests were generally small (i.e., less than 4 ha). For example, nearly 80 percent of lightning-caused fires that occurred between 1951 and 1973 in the Emigrant Basin Wilderness Area on the Stanislaus National Forest were less than 0.1 ha, and none were larger than 4 ha (Greenlee 1973 cited in Potter 1998). In the red fir and lodgepole pine forests of Yosemite National Park, 6 percent of fires that burned between 1972 and 1993 were less than 0.1 ha, and 82 percent were smaller than 4 ha (van Wagtendonk 1994).

Current and future deviation from NRV—Data from the California Fire Perimeters Database indicate that over the past 108 years, the average size of fires (>1 ha) in red fir forests of the assessment area has been about 168 ha. Eighty percent of these fires have been less than 200 ha in size (fig. 3.5). As figure 3.6 suggests, the average size of fires tended to be larger than presettlement fires immediately following fire suppression (1908–1934) but appears to be close to or within the NRV between 1935 and 2015. Note that modern fire areas (since 1908) in figure 3.6 were calculated by removing those portions of fires that burned in other vegetation types.

Collectively, these studies indicate that current fire size is generally within NRV. However, recent trends suggest that fire size may be approaching or possibly exceeding the upper limit of NRV. For example, in their analysis of fires (>40 ha) in the Sierra Nevada and southern Cascades, Miller et al. (2009) found a significant increase in mean and maximum fire size in higher elevation fir forests between 1984 and 2004. Although not limited to red fir or high-elevation forests, Miller et al. (2012b) also documented similar trends of increasing mean and maximum fire size in conifer-dominated forests of northwestern California (Miller and Safford 2012, Miller et al. 2012b)

Estimates based on historical data and contemporary reference sites suggest that fires in red fir forests are dominated by low- to moderate-severity fire effects.

Severity—Estimates based on historical data and contemporary reference sites suggest that fires in red fir forests are dominated by low- to moderate-severity fire effects (tables 3.3 and A1.2). Although stand-replacing fires do occur, they are generally uncommon and relatively limited in size (Laacke and Tappeiner 1996). Leiberg (1900) provided limited historical evidence for this variability during his surveys of montane forests in the southern Cascades of Oregon when he noted that “fire has run through the forest for miles, burning a tree or a group of trees here and there.” High-intensity fires in red fir forests are often associated with pockets of fuel created by tree fall, dense conditions in younger stands, patches of insect-related mortality, avalanches, snow or ice crushing, or the death of larger trees as late-successional conditions are achieved (Skinner 2003).

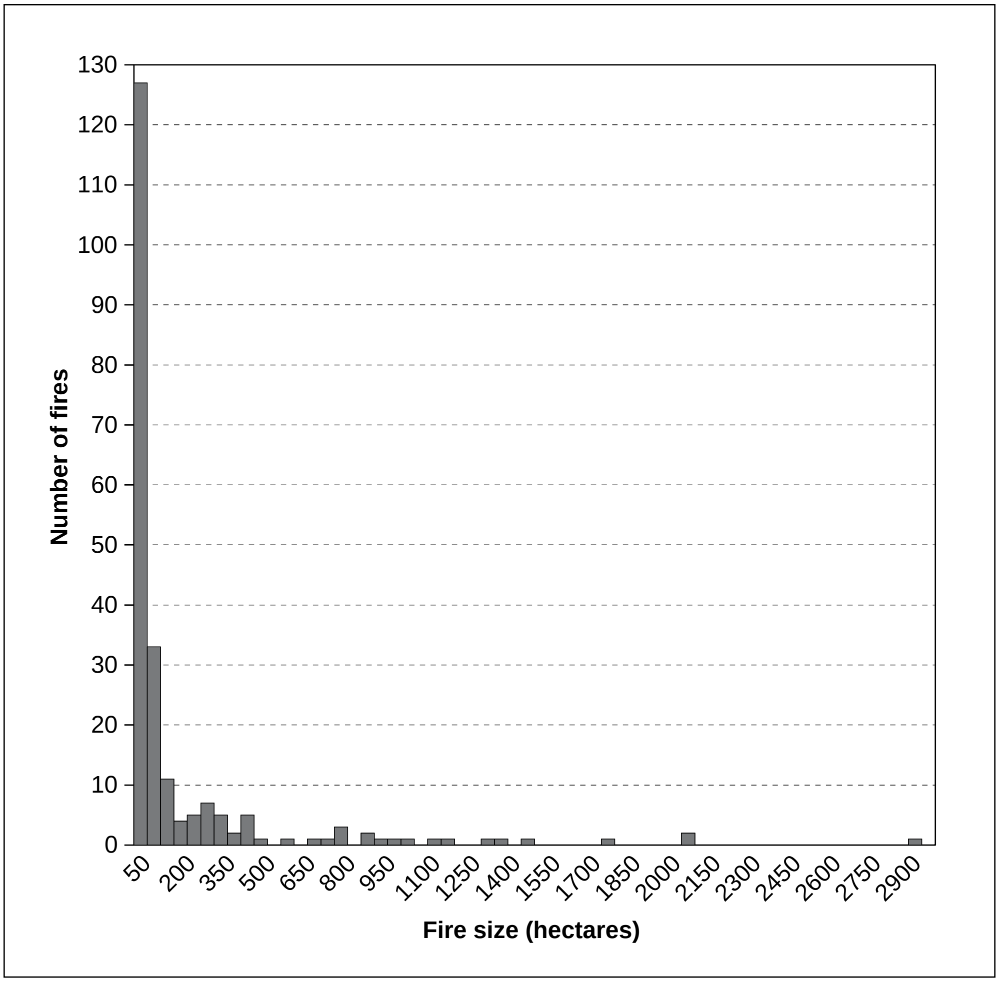


Figure 3.5—Distribution of fire size in red fir forests of the assessment area. Data were obtained from the California fire perimeter database (CDFFP 2015) and include all fires >1 ha in size that burned between 1910 and 2015.

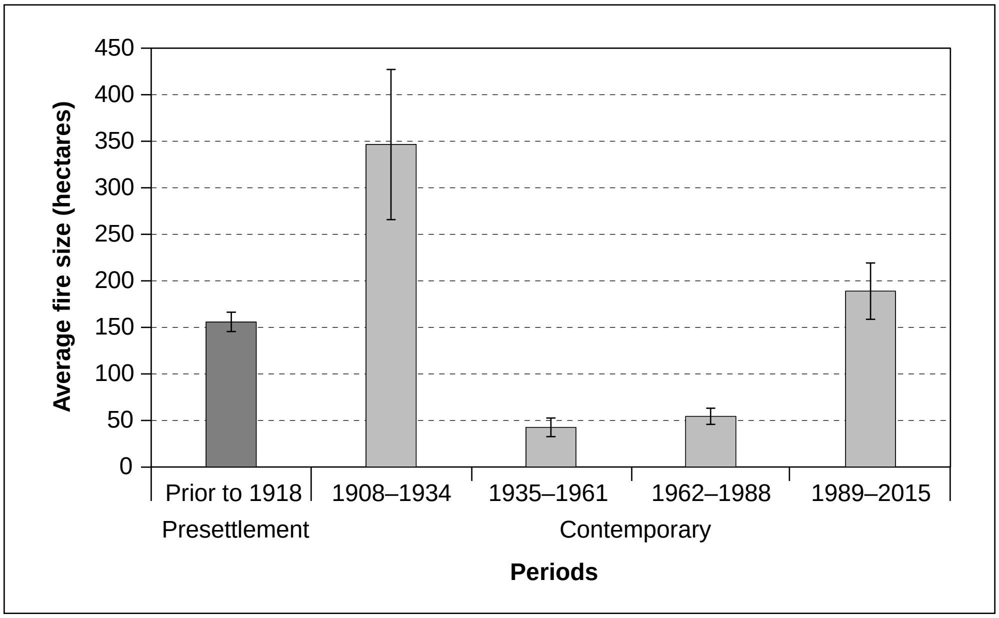


Figure 3.6—Comparison of mean fire size from estimates of presettlement fire regimes and wildfires that have burned within the assessment area since the institution of fire suppression in the early 20th century; the latter are divided into four equal time periods to show temporal trends. Only fires >1 ha are included. Presettlement estimates were taken from studies conducted in the Thousand Lakes Wilderness (Lassen National Forest) and Lassen Volcanic National Park (Bekker and Taylor 2001, Taylor 2000). Contemporary fire sizes were estimated by using the California fire perimeter database (CDFFP 2015). Error bars represent the standard error.

Fire severity patterns in red fir forests are variable, both temporally (i.e., among fire years) and spatially (i.e., within individual fires). For example, Foster (1998) examined two historical fires in the southern Cascades that occurred in red fir forests in 1871 and 1888 and found evidence of plots with high-severity fire adjacent to plots that showed no evidence of fire that year. In their study of two contemporary wildfires in the red fir zone at Crater Lake National Park, Chappell and Agee (1996) documented areas that burned at low, moderate, and high severity, but noted that moderate-severity fire effects dominated, covering an average of 51 percent (range 32 to 69 percent) of the burned area (fig. 3.7). They also found considerable variability in severity effects between the two fires; high-severity fire covered 33.9 percent of the burned area in the Crater Peak burn compared to 6.7 percent of the burned area in the Desert Cone Fire (table A1.2) (Chappell and Agee 1996). Historical estimates of high-severity fire in the southern Cascades range from 13 to 33 percent in lower elevation red fir–white fir forests (Bekker and Taylor 2001, Taylor and Solem 2001) to 19 to 28 percent in higher elevation red fir–western white pine forests (Bekker and Taylor 2001, Taylor and Solem 2001).

Weather conditions can also play a role in fire severity patterns. A good example of this was the 1999 Megram and Fawn Fires, which burned in the Klamath

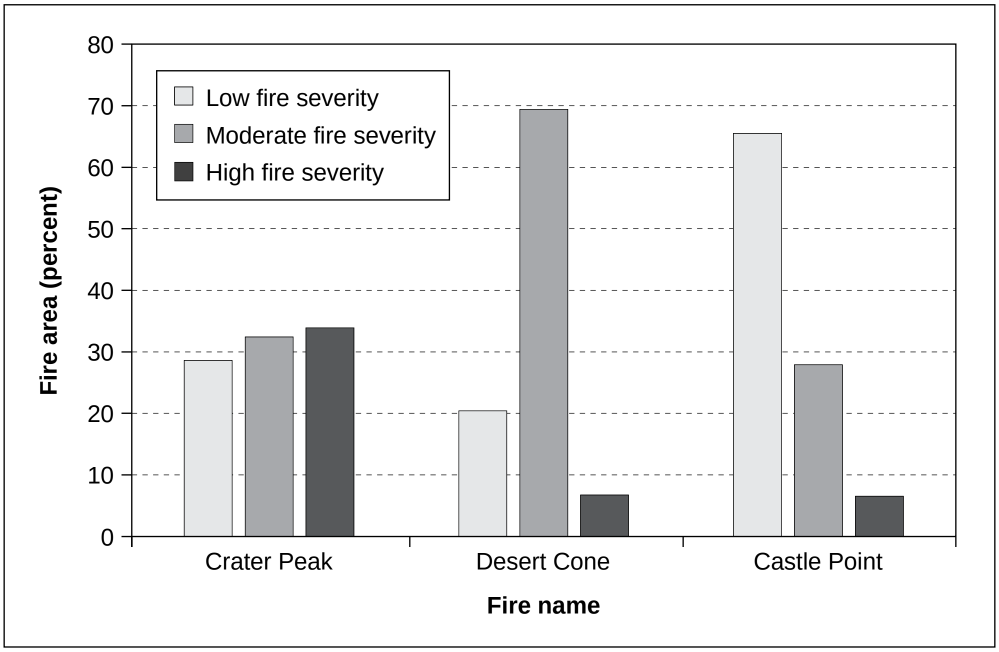


Figure 3.7—Distribution of fire severity in three fires in Crater Lake National Park. Prior to the 1978 Crater Peak Fire, the study sites had experienced at least four fires over the past 100 years. Chappell (1991) indicated that there was no evidence of fire (over a period of 190 years) before the 1982 Desert Cone Fire. The fire history at the 1986 Castle Point Fire study site was not noted. (Reproduced from Chappell and Agee 1996).

Mountains of the Shasta-Trinity and Six Rivers National Forests. During the first month of these fires, stable air masses produced by inversions resulted in predominantly surface fuel consumption, with occasional crown fires in steep terrain and heavier fuels; however, a series of low-pressure weather systems in the fall produced high northeast winds, resulting in rapid fire expansion, large crown fire runs, and large areas with high-severity fire effects (Jimerson and Jones 2000). The 1988 Prophecy Fire in Crater Lake National Park also burned at low to moderate severity for several weeks before strong winds caused the fire to burn more than 800 ha in a single day (Agee 1993). Stand structure and composition may also influence fire severity; mature red fir develops thick bark, making it more fire tolerant than young red fir cohorts and many of its associated species. In Crater Lake National Park, Chappell and Agee (1996) found that mature and old-growth red fir stands (>100 years old) burned at lower severity and had lower proportions of high-severity burned areas (4.5 percent) than young red fir stands (50 to 80 years old; 24 percent burned at high severity).

Inferences related to high-severity patch size are based almost exclusively on data collected in contemporary reference sites, primarily in Crater Lake National Park. In his study of two wildfires in the red fir zone, Chappell (1991) documented a high degree of variability in the size of high-severity patches. The Crater Peak Fire contained the largest high-severity patch at 15.3 ha, which was much larger than any other area of high severity within the two study sites. Within the Crater Peak Fire, three large high-severity patches (>2 ha) occupied 61 percent of the area characterized as high severity. Agee (1998) estimated an average high-severity patch size of 1.3 ha (median = 0.4 ha) in red fir forests of Crater Lake National Park. Collins and Stephens (2010) investigated fire severity patterns in two fires that occurred in the high-elevation, mixed-conifer forests of the Illilouette Creek Basin in Yosemite National Park. They documented a mean high-severity (>95 percent tree mortality) patch size of 9.1 ha (median = 2.2 ha), with most patches ≤4 ha in size.

Current and future deviation from NRV—Current averages for high-severity proportion and patch size may be approaching the upper limit of NRV, but historical information is very limited with respect to these variables (table 3.3, table A1.2). Haugo et al. (2019) compared fire severity patterns in contemporary large wildfires (1984–2015) with modeled historical fire regimes in the Klamath Mountains and eastern Cascade Range of Oregon. They found that the observed proportion of high-severity fire in red fir forests was 38 percent in the Klamath Mountains and 53 percent in the eastern Cascades. These values were above and outside of the range (5th to 95th percentile) of high-severity fire expected under a historical fire

Table 3.3—Average proportion of fire severity classes in red fir forests of the assessment area based on historical and contemporary reference site information[a]

Aggregation/group	Study locations	Low	Moderate	High	Forest types
		- - - - - Percentage - - - - -			
Historical estimates	Thousand Lake and Caribou Wildernesses (Lassen National Forest)	26	36	23	Red fir, red fir–white fir/western white pine
Reference sites	Crater Lake National Park	38	43	16	Red fir, red fir–mixed conifer
Modeled[b]	North Coast Range of Oregon and northern California, Cascade Range; eastern Cascades and Klamath Mountains of Oregon[c]	49	34	17	Red fir
Contemporary	Northwestern bioregion (California only); eastern Cascades and Klamath Mountains of Oregon	13	42	37	Fir, high-elevation conifer

[a] Individual fire severity estimates and sources are presented in table A1.2 of appendix.

[b] Based on LANDFIRE Biophysical Setting Model estimates of historical reference conditions. High-severity fires cause >75 percent kill or top-kill of the upper canopy layer; moderate-severity fires cause 26 to 75 percent, and low-severity fires cause <26 percent.

[c] From simulation results presented in Haugo et al. (2019), which were also based on LANDFIRE Biophysical Setting Model state and transition models.

regime, which they modeled as 7 to 19 percent in the Klamath Mountains and 7 to 22 percent in the eastern Cascades (Haugo et al. 2019). Miller et al. (2012b) analyzed trends in fire severity patterns in high-elevation fir forests in northwestern California for fires (>400 ha) that occurred between 1987 and 2008 and found no clear trend in the percentage of area burned at high severity. They estimated an average proportion of 19 percent high severity for contemporary, long-unburned (i.e., those that burned between 1987 and 2008, but had no previously recorded fire since 1910) fir and high-elevation conifer forest types; the percentage of high severity was less (11 percent) for areas that burned twice between 1910 and 2018.

These estimates of contemporary severity are slightly above the range of severities estimated for presettlement and contemporary red fir stands in the assessment area (table 3.3, table A1.2). These trends, coupled with future projections, indicate that fire severity may increase during the mid-21st century in many high-elevation forests in response to warming temperatures and drier conditions during the fire season (Lenihan et al. 2003, 2008).

Insects and pathogens—

Native insects and pathogens have been present for millennia (essentially as long as their host trees) and are an integral component of red fir forest ecosystems (Hawksworth 1978, Mathiasen 2011). Historically, insect- and disease-related mortality reduced intertree competition for site resources, generated coarse woody debris between fires, and created gaps in the overstory canopy that released red fir trees in the understory and promoted regeneration (Hessburg et al. 1994). The scale and intensity of an infestation, and subsequent effect on tree growth and mortality, was related to the abundance of the insect or pathogen and susceptible host trees, as well as a range of other environmental factors (e.g., water and light availability, stand density, site quality, etc.). In red fir forest types, native insects and pathogens generally kill or impair the growth of individual trees or small clumps of trees, resulting in few major shifts in species composition or forest structure (Ferrell 1996).

There are several native insects and pathogens that interact to affect the growth and health of red fir trees, especially during periods of stress associated with extended drought, competition for limited resources, or following disturbance (Ferrell 1996). This suite of species includes the parasitic dwarf mistletoe (*Arceuthobium abietinum* sp. *magnificae* and *A. abietinum* sp. *wiensii*); several insects, including the fir engraver beetle (*Scolytus ventralis*), flatheaded fir borer (*Melanophila drummondii*), and roundheaded fir borer (*Tetropium abietis*); and several fungal pathogens such as Indian paint fungus (*Echinodontium tinctorium*) and those associated with Cytospora canker (*Cytospora abietis*), Annosus root disease (*Heterobasidion occidentale*), and occasionally Armillaria root disease (*Armillaria ostoyae*) (Ferrell 1996, Hadfield et al. 1986, Laacke 1990, Mortenson et al. 2015). The most common pathogens in red fir forests are red dwarf mistletoe and Cytospora canker, which along with the fir engraver beetle, frequently occur in combination (Laacke 1990).

In red fir forest types, native insects and pathogens generally kill or impair the growth of individual trees or small clumps of trees, resulting in few major shifts in species composition or forest structure.

Little quantitative data are available that describe the abundance and distribution of insects and pathogens in red fir forests before European-American settlement. Sedimentary pollen records from the southern Cascade Range in central Oregon and the Sierra Nevada provide evidence that mistletoe has been a persistent component in higher elevation forests since the early Holocene, likely fluctuating with changes in canopy cover and tree density (Brunelle and Anderson 2003, Sea and Whitlock 1995). Botanists and plant pathologists identified dwarf mistletoe as a significant pathogen in coniferous forests of the Western United States by the 19th and early 20th centuries (Hawksworth 1978). In the late 1950s, the California Forest Pest Council (1960) estimated that 36 percent of trees in red fir stands were affected by dwarf mistletoe, especially in older and denser forests, and often associated with Cytospora canker. These same surveys described increases in the abundance of fir engraver beetle throughout the range of red fir.

Current and future deviation from NRV—Despite the lack of quantitative historical evidence, some inferences can be made about potential changes in the scale and intensity of insect and disease infestations over time. As detailed in the discussion below (see "Tree Density" and "Tree Size and Size Class Distribution" sections), the exclusion of fire from red fir forests in the assessment area has resulted in a significant increase in tree densities, primarily owing to shifts to smaller size classes over time. Stand density has been shown to have a strong influence on insect- and disease-related tree mortality in western conifer forests. High stand densities allow for easier transmission of root diseases via root-to-root contact (Woodward et al. 1988) and increase the likelihood of mistletoe seed dispersal to susceptible hosts (Reich et al. 1991). The severity of fir engraver beetle attacks has been shown to increase in areas of high stand density, when trees are stressed because of competition (Ferrell et al. 1994). Climate also influences the survival and spread of pathogens, as well as the susceptibility of the host. Many pathogens in red fir forests are limited by cold winter temperatures; therefore, increases in minimum winter temperature may increase overwintering survival, and the subsequent severity of an infestation (Kliejunas 2011).

Several recent analyses indicate that dwarf mistletoe, Cytospora canker, and other pathogens may be increasing in abundance in many parts of the assessment area. In 2015, aerial surveys of the north Coast Range, Klamath Mountains, and Cascade Range noted severe chronic flagging (i.e., recent foliage loss and localized branch death in the crown) caused by Cytospora canker in fir stands at high elevations in areas with severe drought conditions (USDA FS 2015b). A comparison of recent U.S. Forest Service (USFS) Forest Inventory and Analysis plots between 2005 and 2010 show increases in red fir mortality rates throughout the range over time (Mortenson et al. 2015). Similar increases in mortality rates were observed in coniferous forests (including red fir) in Yosemite, Sequoia, and Kings Canyon National Parks between 1983 and 2004 (van Mantgem and Stephenson 2007). The primary factors associated with the increased red fir mortality were increased temperatures associated with climatic water deficit and the occurrence of dwarf mistletoe and associated Cytospora canker, although the role of other mortality factors (e.g., fir engraver beetle, Heterobasidion root disease, stand density) was not clear (Mortenson et al. 2015).

A study conducted by DeSiervo et al. (2018) also found evidence of recent widespread red fir die-off in the Russian Wilderness in the Klamath Mountains. They found that 20.7 percent of red fir sampled were recently dead or dying, owing to the interaction of dwarf mistletoe (evident on 20.4 percent of trees), fir engraver beetle (evident on 34.7 percent of trees), and a canker forming fungi (*Cytospora*). In their study area, red fir mortality increased with increasing small-tree density and minimum winter temperature, the latter of which increased by about 1.6 °C between the reference (1951–1980) and recent (2004–2014) periods. In the Sierra Nevada, Meyer et al. (2019) found that 26 percent of red fir trees sampled were either recently dead or showed signs of seriously impaired health; the highest crown loss was associated with topographic areas of greater moisture stress.

In 2012, the USFS conducted a nationwide strategic assessment of mortality risk resulting from insects and diseases (Krist et al. 2014). They used a suite of variables, including host tree parameters, climatic data, physiographic variables, soil information, and other data to create individual pest and pathogen models. They used these models to project basal area mortality risk for individual tree species over a 15-year period (2013 to 2027). Within the assessment area, their models projected an average basal area loss of 8 percent for red fir, primarily owing to fir engraver beetle (Krist et al. 2014).

Kliejunas (2011) conducted a risk assessment to determine the likelihood that current and projected climate trends would increase the adverse effects caused by pathogens in forests of the Western United States. He found high to very high potential (defined as the likelihood of damage multiplied by the potential effects) for dwarf mistletoe and Armillaria root disease under both drier and wetter future climate projections. His analysis suggests that increased temperatures will increase both the incidence of these pathogens and the infection and mortality of host trees, especially when hosts are stressed by moisture deficiency.

Collectively, these findings suggest that the scale and intensity of native insect and pathogen infestations may be increasing within red fir stands of the assessment area. These changes are most likely driven by changing climatic factors, such as warming temperatures, decreased snowpack, and extended drought (Mortenson et al. 2015, USDA FS 2015b, van Mantgem and Stephenson 2007), as well as increases in stand density and changing successional patterns owing to fire suppression (Conklin and Fairweather 2010, Hessburg et al. 1994).

Wind—

High winds are a common feature of many high-elevation forests. Wind-related disturbances in red fir forests are highly variable both spatially and temporally. Long periods of time can pass with little damage, then a major blowdown event will occur, resulting in breakage of boles and limbs, tree uprooting (Potter 1998), and widespread dieback of shrubs (Nelson and Tiernan 1983). Muir (1894) noted trees breaking off every few minutes throughout the night during a major blowdown event in forests of the Sierra Nevada in December of 1874. In the northern Sierra Nevada, sustained windspeeds of 113 to 126 km per hour were recorded during the Columbus Day storm of October 12, 1962, which caused substantial damage in red fir forests (Potter 1998). Taylor and Halpern (1991) measured radial growth patterns in red and white fir stands of the southern Cascades and found growth releases related to two windstorm events that occurred between 1960 and 1990. In one of the few studies examining the effect of wind damage on upper montane forests, Gordon (1973) found that wind (based on two extreme events) accounted for 60 percent of tree damage and 77 percent of gross stand volume loss within intact red fir–white fir stands adjacent to clearcut stands in the Swain Mountain Experimental Forest. The direct effects of wind (e.g., bole and limb breakage, uprooted trees) accounted for 71 percent of tree mortality in their study and indirect effects (e.g., tree struck by another wind-damaged tree) accounted for the remaining 29 percent mortality. Gordon (1973) found that wind damage was most prevalent in individual trees, rather than groups of trees and that wind had a disproportionate impact on larger trees in the dominant and codominant crown classes, most likely resulting from a combination of wind damage and root rot.

Wind events can have both a direct effect on red fir forest structure (i.e., through tree mortality), as well as an indirect effect on subsequent fire behavior and severity. For example, in the Trinity Alps Wilderness on the Shasta-Trinity National Forest, a high-intensity windstorm during the winter of 1995–1996 caused extensive blowdown on about 4856 ha, including high-elevation red fir forests (Jimerson and Jones 2000). In 1999, the Megram Fire burned through these areas of high windthrow, and an analysis conducted by Jimerson and Jones (2000) documented higher frequency of high-severity fire in areas with blowdown (46 percent) compared to areas without blowdown (29 percent). A windstorm in 2011 in the eastern Sierra Nevada toppled 400,000 trees in red fir and upper montane forests on the Inyo National Forest and Devils Postpile National Monument (USDA FS 2012). Large trees were disproportionately uprooted (86 percent) and snapped (14 percent), creating variable-sized canopy gaps in red fir forests with heavy postdisturbance fuel loading (Hilimire et al. 2013).

Current and future deviation from NRV—The frequency and effects of historical wind events are difficult to compare to current rates owing to the highly infrequent or unpredictable nature of these climatic processes. Nevertheless, current rates of wind events in red fir forests of the assessment area are broadly considered to be within the NRV. Changing weather patterns resulting from climate change may alter the scale and severity of wind events in red fir forests; however, the type and amount of change of these smaller scale events are difficult to predict because they are below the resolution of today's GCMs.

Avalanche—

Avalanche may occur in red fir and other high-elevation forests in the assessment area, especially on slopes exceeding 30 to 40 percent (Potter 1998). Most large avalanches occur during years of high snowpack and following heavy snowfall events, often coupled with high windspeeds (Agee et al. 2000, Davis et al. 1999). Avalanches can occur frequently in steep gullies and chutes or on rare occasion, have major impacts on slopes that show no evidence of past activity (e.g., older forested stands on lower slope positions located beneath open slopes with few trees) (Potter 1998). The 1860–1864 Brewer (1930) expedition noted evidence of avalanche in subalpine forests of the southern Sierra Nevada, and Muir (1894) observed the complete removal of an older upper montane forest stand by avalanche.

Despite past and recent observations of avalanche in high-elevation forests of the assessment area, postavalanche successional dynamics remain poorly documented in the region. In the European Alps, where postavalanche successional dynamics have received more research attention, high-elevation forests affected by avalanche are characterized by smaller and shorter trees, reduced stem densities, and shade-intolerant species; at the landscape scale, avalanches create structural heterogeneity, while also influencing the spread of other disturbances (Bebi et al. 2009). In upper montane forests of the Sierra Nevada and Cascade Range, moderate avalanche return intervals (i.e., every few decades) can result in the dominance of hardwood vegetation, while more frequent avalanches can result in a shift from tree- to shrub-dominated vegetation (Fites-Kaufman et al. 2007).

Current and future deviation from NRV—Current avalanche rates are likely within the NRV. Projected declines in snowpack in the assessment area in the 21st century could result in reduced probability of avalanche and increased landslide activity in the coming decades (Huggel et al. 2012). However, reductions in snowpack could be offset in the near future by potential increases in precipitation variability and the frequency of heavy, snowfall events and rain-on-snow events, which can elevate avalanche probability, especially at higher elevations (Bebi et al. 2009, Swain et al. 2018).

Structure

The following discussion examines two overarching elements of red fir forest structure (Franklin et al. 2002): (1) the type, number, and size of individual structural elements (e.g., trees, snags, and understory vegetation); and (2) the spatial arrangement of these elements (e.g., whether they are uniformly spaced or clumped). Red fir forests in the assessment area are relatively similar to stands in the southern Cascade Range and Sierra Nevada, in terms of forest structure (e.g., tree density, basal area, canopy cover) (Barbour and Woodward 1985). Therefore, the discussion below integrates studies from the assessment area with information from areas adjacent to, but outside of the assessment area (e.g., southern Cascade Range of California and Oregon), and occasionally draws from studies conducted in more distant portions of the red fir range (e.g., the Sierra Nevada) when bioregional information is lacking.

Average tree density was lower in presettlement red fir forests compared to current, fire-excluded red fir forests.

Tree density—

A broad comparison of all unlogged red fir stands in the assessment area suggests that average tree density was lower in presettlement red fir forests compared to current, fire-excluded red fir forests (table 3.4, fig. 3.8). Although the paucity of historical data within the assessment area limits a more comprehensive analysis, this trend appears similar across vegetation types (e.g., red fir, red fir–white fir, red fir–mountain hemlock, etc.) and geographic regions (north Coast Mountains, Klamath Mountains, and southern Cascades) within the assessment area (see app. 1: table A1.3.

Table 3.4—Average and standard deviation (SD) of total and relative red fir tree density and basal area in historical or contemporary reference sites and current red fir stands, including Forest Inventory and Analysis (FIA) data collected between 2001 and 2014[a]

Summary statistic	Tree density[b] Total	Tree density[b] Red fir	Tree density[b] Red fir	Basal area Total	Basal area Red fir	Basal area Red fir	Number of plots	Forest types
	- - - *Trees/ha* - - -		*Percentage*	- - - *m²/ha* - - -		*Percentage*		
NRV estimates:								
Historical (pre-1940) or contemporary reference red fir stands:[c]								
Mean	388	329	81	63	53	81	76	Red fir
SD	195	216	19	20	27	15		
Current estimates:								
Contemporary (post-1940) red fir stands:[d]								
Mean	1428	854	58	92	43	47	80	Red fir–white fir
SD	306	416	15	21	10	9		
Mean	902	504	85	54	50	89	109	Red fir
SD	514	346	13	26	17	12		
Mean	1043	688	65	86	52	59	125	Red fir–mountain hemlock, red fir–western white pine
SD	313	321	20	19	22	18		
Contemporary (2001–2014) FIA red fir stands:[e]								
Mean	289	139	48	21	10	49	98	Red fir–white fir
SD	301	—	—	19	—	—		

— = no estimate available.

[a] Natural range of variation (NRV) estimates from contemporary landscapes were identified as having current fire return intervals that were less than 1.5 times the mean presettlement fire return interval of 43 years, calculated for red fir forests across the assessment area in table 3.1. Individual stand structure variables and sources are presented in table A1.3 of the appendix.

[b] Tree density averages are based on studies that had a minimum tree diameter at breast height (d.b.h.) between 3 and 5 cm; refer to table A1.3 of the appendix for detailed information about all individual studies, including those that were not used in this summary table.

[c] NRV reference red fir stands are located within contemporary, active-fire regime landscapes that have a current FRI that is less than 1.5 times the mean presettlement FRI of 43 years (see table 3.1).

[d] Contemporary estimates come from unlogged, fire-excluded (i.e., unburned; missed two or more historical FRIs) red fir stands in the assessment area.

[e] FIA estimates are based on FIA plots throughout the entire assessment area, including only trees ≥5 cm d.b.h. FIA data represent an unbiased sample of current conditions in red fir forests. Therefore, inclusion of plots in managed (e.g., logged, burned, etc.) and unmanaged stands may result in lower density and basal area estimates than other inventories targeting red fir forest stands.

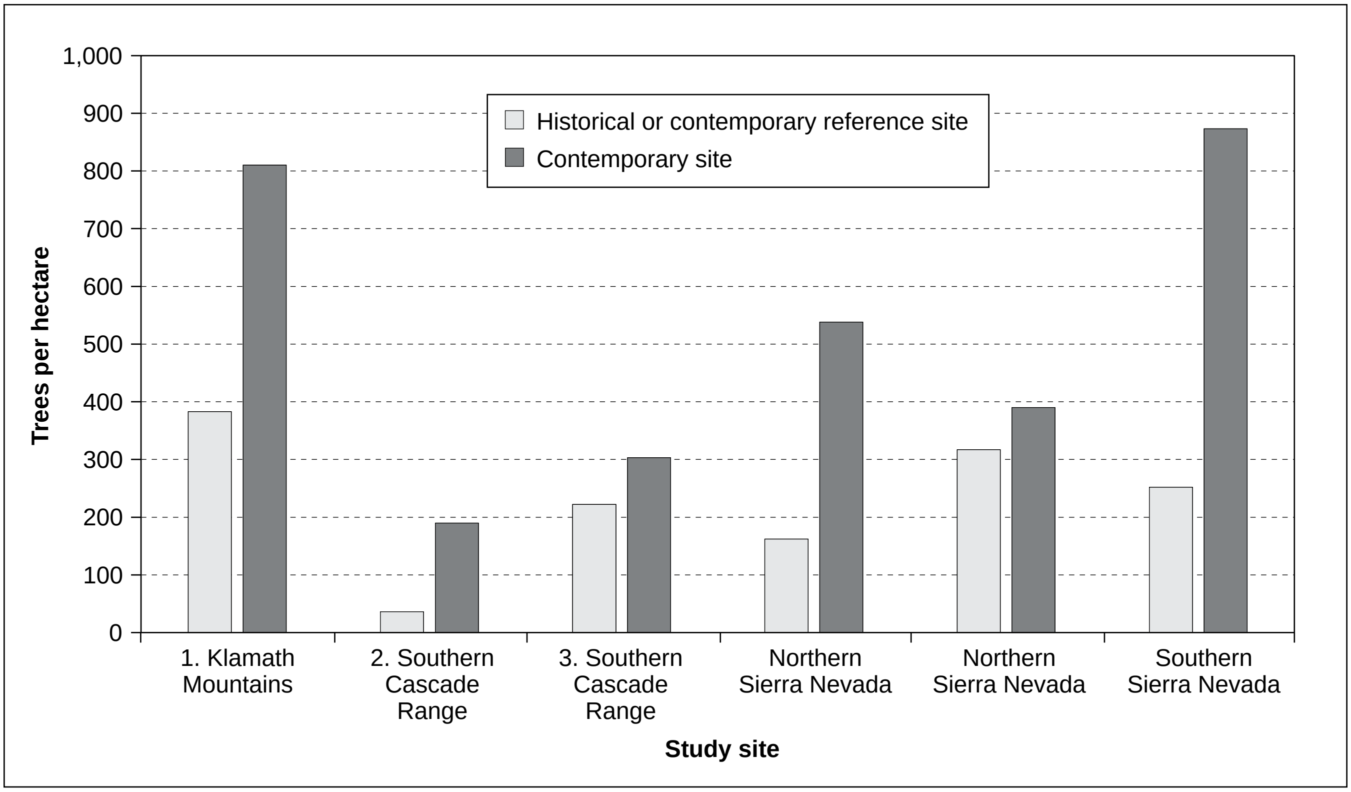


Figure 3.8—Comparison of current and historical tree densities at six red fir study sites in and adjacent to the assessment area. 1 = USDA FS (n.d.) (diameter at breast height [d.b.h.] >5 cm), Marble Mountain Wilderness. Natural range of variation (NRV) estimate from contemporary unlogged site with active fire regime; 2 = Dubrasich (2010) (d.b.h. >20 cm), Huckleberry Lake Special Interest Area, Oregon. Historical period is 1825; 3 = USDA FS (n.d.) (d.b.h. >5 cm), Lassen Volcanic National Park. NRV estimate from contemporary unlogged site with active fire regime; 4 = Taylor (2004) and Taylor et al. (2014): Lake Tahoe Basin (d.b.h ≥10 cm). Historical period is pre-1870; 5 = Dolanc (2014): northern and central Sierra Nevada (d.b.h. >10.2 cm). Historical period is 1929–1936; 6 = Meyer et al. (2019): Yosemite, Sequoia, and Kings Canyon National Parks and Giant Sequoia National Monument (d.b.h. >5 cm). NRV estimate from contemporary unlogged site with active fire regime.

The few studies that have been conducted within the assessment area that provide estimates of presettlement stand structure for red fir forests are primarily internal white papers and unpublished datasets. One unpublished analysis, conducted by Dubrasich (2010) as part of a large-scale effort to define reference conditions on the Umpqua National Forest in Oregon, used backdated stand reconstruction methods to estimate presettlement conditions in unlogged red fir stands in the Huckleberry Lake Special Interest Area. This analysis suggested a fivefold increase in stand density over a 185-year period with tree density increasing from an estimated 36 trees/ha in 1825 to 190 trees/ha in 2010 (fig. 3.9).

In 2016, the Pacific Southwest Region Ecology Program installed 17 plots in the Marble Mountain Wilderness on the Klamath National Forest as part of a regionwide effort to define the NRV in red fir forest stand characteristics (USDA FS, n.d.). Plots were established in unlogged, fire-excluded stands (n = 6) as well

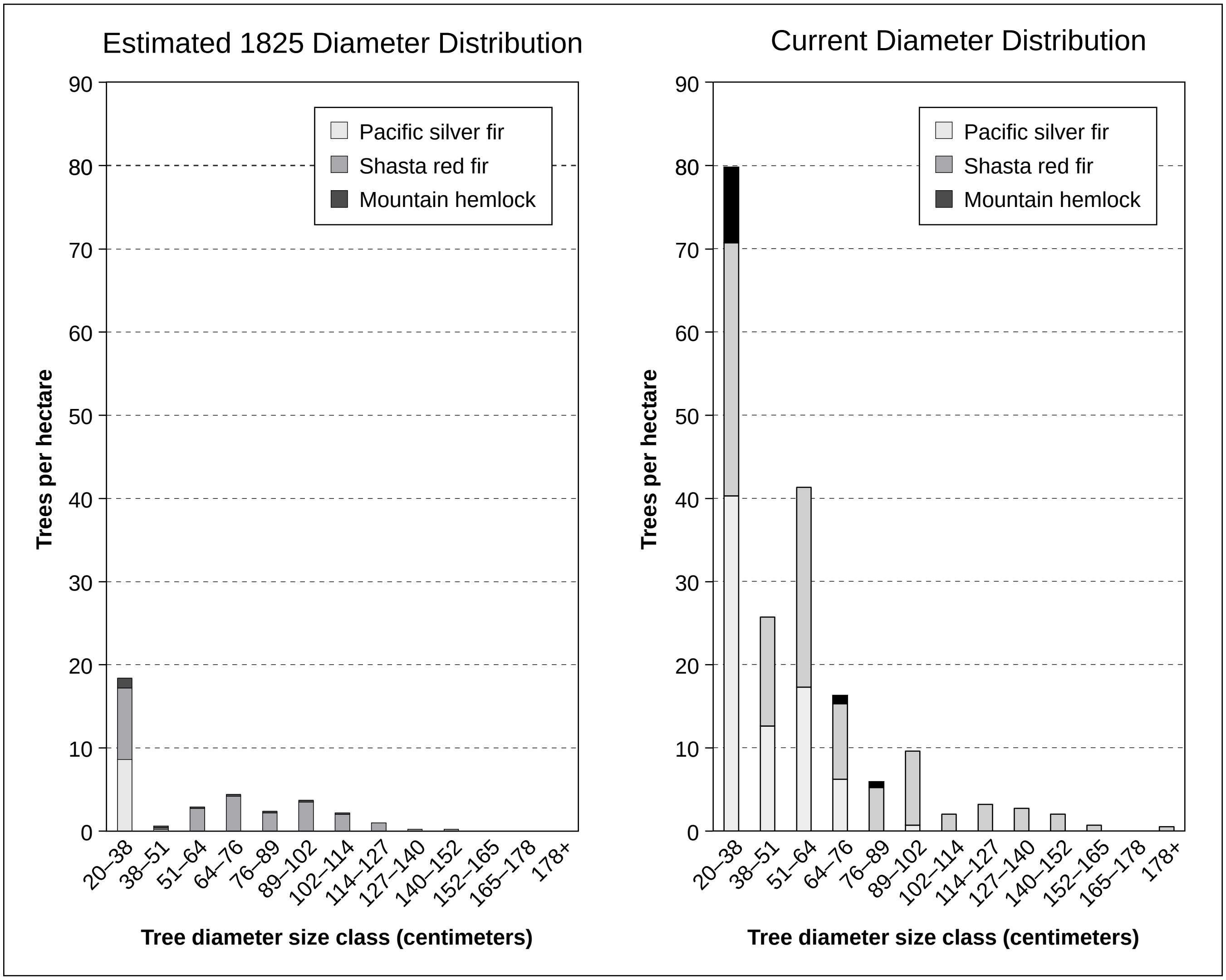


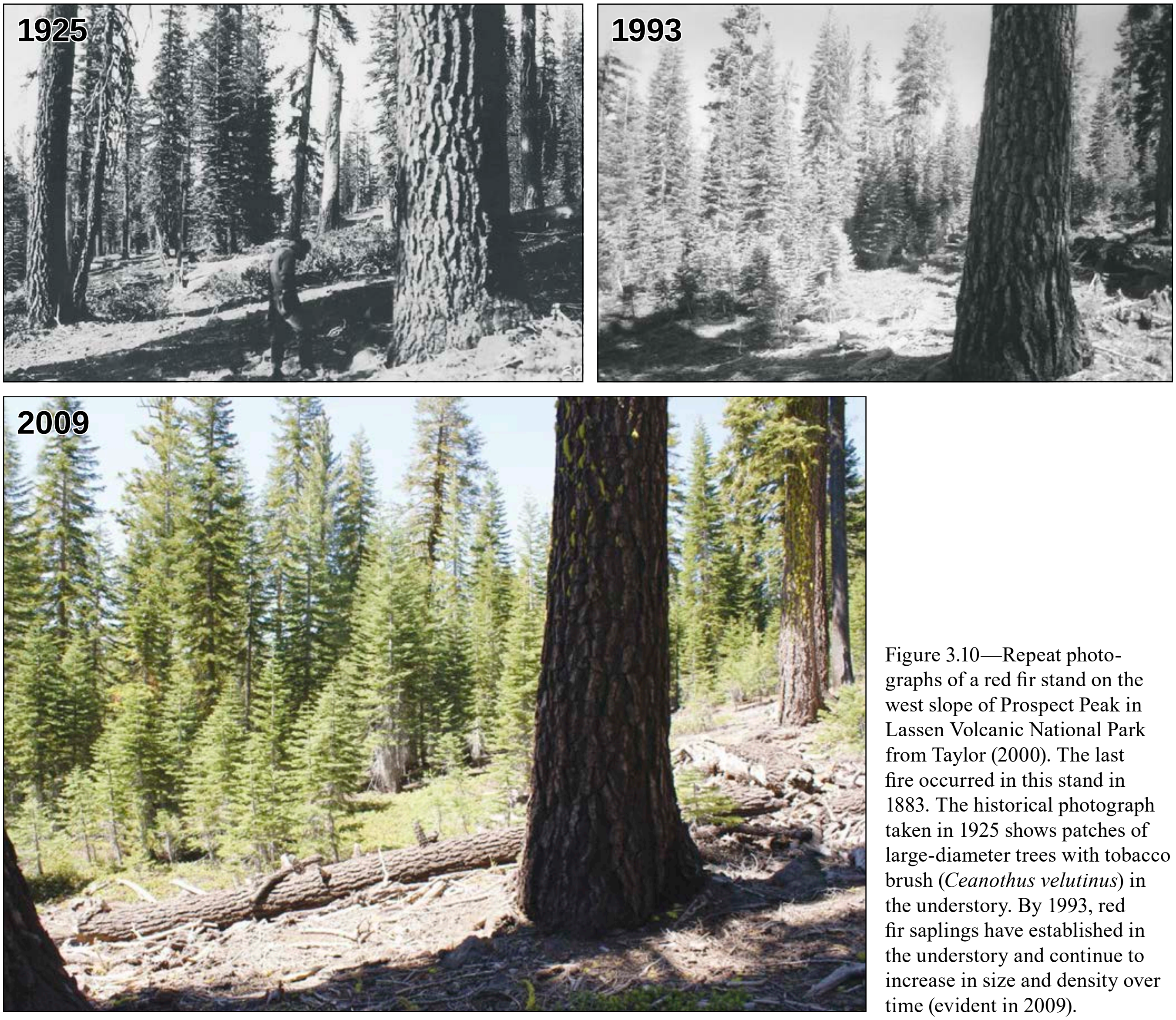
Figure 3.9—Comparisons by Dubrasich (2010) of stand conditions presettlement (1825) and in contemporary (2010) unlogged red fir stands in the Huckleberry Lake Special Interest Area on the Umpqua National Forest in Oregon. Presettlement stand conditions were estimated using backdated stand reconstruction methods. One important factor to consider when interpreting the magnitude of these differences is the inability of reconstruction studies to accurately account for very small trees in the historical reference period, particularly for red fir, which can decay rapidly.

as contemporary reference landscapes (n = 11), the latter of which was defined as unlogged sites that were dominated by red fir and had little to no FRI departure (i.e., stands had burned at low to moderate severity in at least two overlapping fires over the past 50 years). Preliminary analysis of these data suggests higher tree densities in fire-excluded red fir forests (mean = 810 trees/ha; range = 320 to 2,191 trees/ha) compared to contemporary reference stands with active fire regimes (mean = 388 trees/ha; range = 190 to 650 trees/ha).

Studies conducted in red fir forests outside of the assessment area have shown similar trends in tree density. In one unpublished study conducted near Lassen

Volcanic National Park, Alan Taylor retook a series of photographs that were first recorded in the early 1920s (figs. 3.10 through 3.12). The early photographs show an open red fir forest structure, with patches of larger trees occurring within a matrix of understory vegetation. Photographs of these same stands, taken by Taylor in 1993 and 2009, document substantial increases in tree density occurring both within stands (fig. 3.10) and across the landscape (figs. 3.11 and 3.12).

Meyer et al. (2019) compared burned and unburned red fir stands in Yosemite, Sequoia, and Kings Canyon National Parks and Giant Sequoia National Monument; all the unburned plots that they sampled had missed at least two FRIs (>87 years), while most of the burned plots (88 percent) had burned in the past 20 years.



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Alan Taylor

Alan Taylor

Figure 3.10—Repeat photographs of a red fir stand on the west slope of Prospect Peak in Lassen Volcanic National Park from Taylor (2000). The last fire occurred in this stand in 1883. The historical photograph taken in 1925 shows patches of large-diameter trees with tobacco brush (*Ceanothus velutinus*) in the understory. By 1993, red fir saplings have established in the understory and continue to increase in size and density over time (evident in 2009).

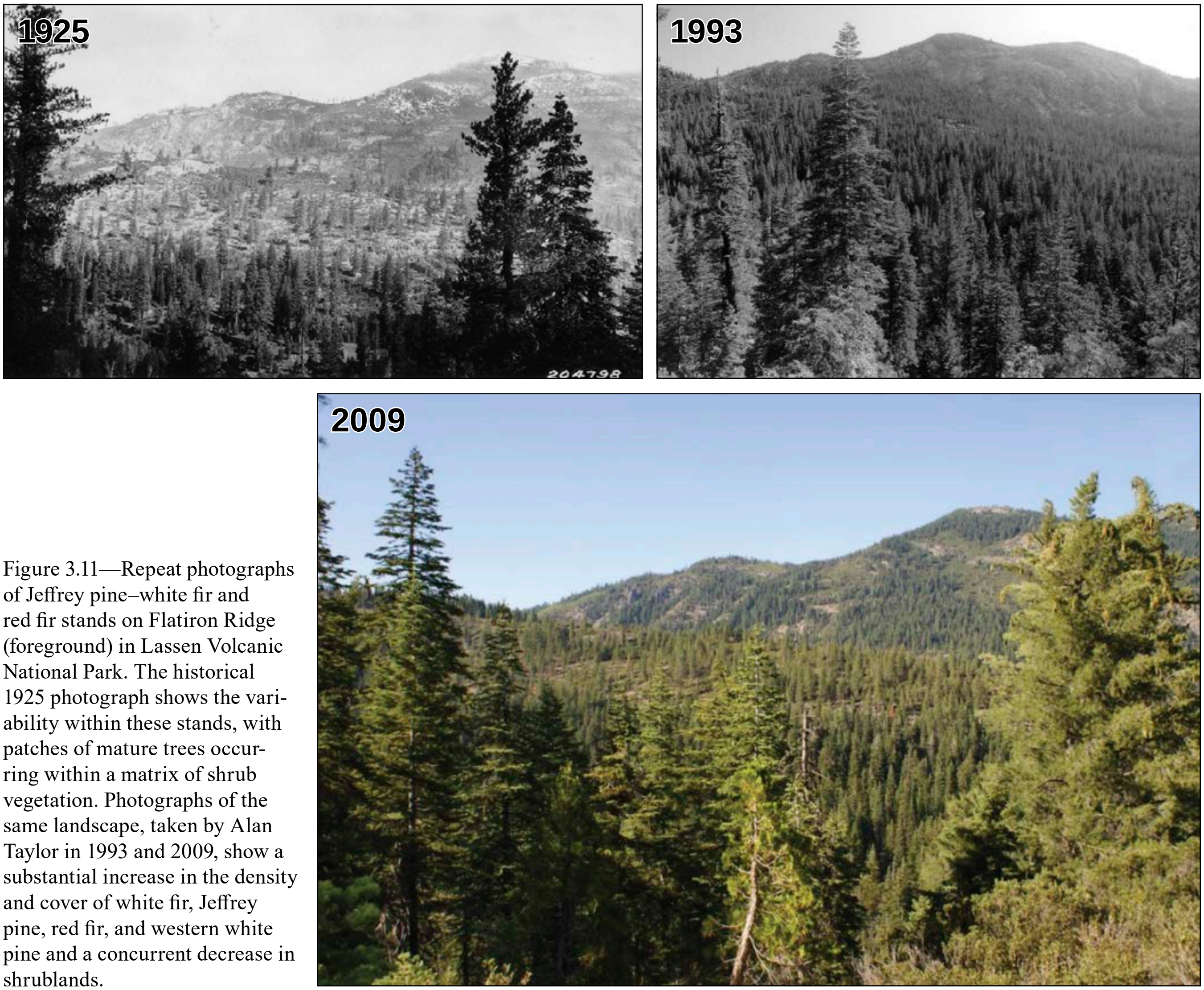

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Figure 3.11—Repeat photographs of Jeffrey pine–white fir and red fir stands on Flatiron Ridge (foreground) in Lassen Volcanic National Park. The historical 1925 photograph shows the variability within these stands, with patches of mature trees occurring within a matrix of shrub vegetation. Photographs of the same landscape, taken by Alan Taylor in 1993 and 2009, show a substantial increase in the density and cover of white fir, Jeffrey pine, red fir, and western white pine and a concurrent decrease in shrublands.

They found that burned plots had 3.5 times lower tree densities (mean = 252 trees/ha; ± 159 standard deviation [SD]) than unburned plots (mean = 873; ± 498 SD). Taylor et al. (2004, 2014) found similar results in the Lake Tahoe Basin where presettlement (pre-1870) tree densities in historical red fir–western white pine forests (average = 162; range: 118 to 208 trees/ha) were substantially lower than modern forests that were intensively logged in the late 19th century (average = 538; range: 214 to 842 trees/ha). In addition, inventories of red fir stands in the northern and central Sierra Nevada have also documented increases in tree density of about 23 to 51 percent between historical (1929–1936) and more recent (2001–2010) time periods (Dolanc et al. 2014a, 2014b).

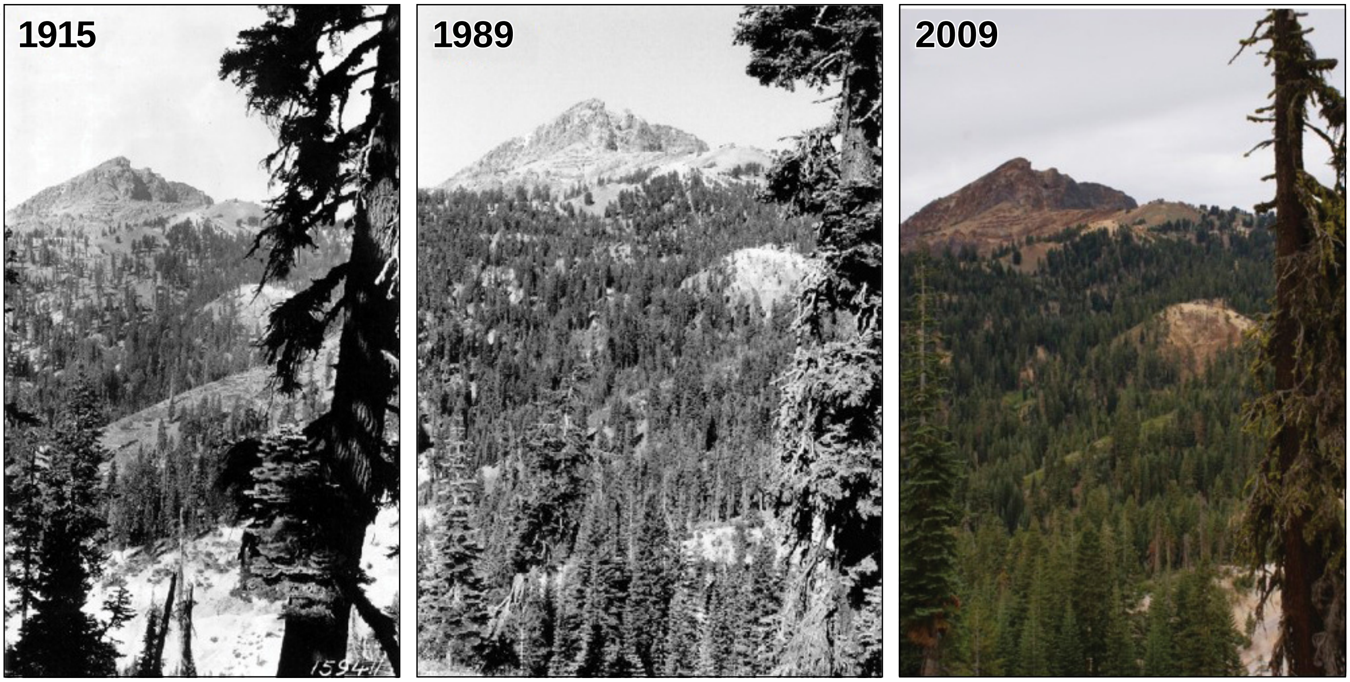

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Figure 3.12—Repeat photographs of red fir stands on the eastern slope of Brokeoff Mountain in Lassen Volcanic National Park. The historical photograph was taken in 1915 as part of the Vegetation Type Mapping survey of California survey conducted by Albert E. Wieslander (Kelly et al. 2005). The charred standing dead trees in the shrub field (middle foreground) of the historical photograph may have been the result of a moderate-severity fire. Photographs of the same landscape, taken by Alan Taylor in 1989 and 2009, show substantial infilling within the stands, particularly at higher elevations.

Historical data and observations can also provide some insight into presettlement stand densities. For example, historical inventories of red fir stands conducted by Albert E. Wieslander and others in the 1930s (Kelly et al. 2005) provide mean density (trees >10 cm d.b.h.) estimates that range from 372 trees per hectare in the northern portion of the assessment area to 262 trees per hectare in Lassen Volcanic National Park. Additional observations documented in letters from William Brewer in the early 1860s (Brewer 1930: 480–481) describe forest structure at high elevations along the crest of the Siskiyou Mountains. In these areas, which generally have thin, erodible soils, forests were relatively open:

> Just north of this ranch are several high peaks of the Siskiyou Mountains…. the whole region a mountainous one—the peaks five thousand to seven thousand feet high, some indeed much higher—and all furrowed into deep canyons and sharp ridges, many of the former over two thousand feet deep. The hills are covered with scattered timber, not dense enough to be called forests, or in places with shrubby chaparral.

Current and future deviation from NRV—Overall, there has been an increase in total tree density in red fir forests of the assessment area over the past century (table 3.4). This is evident in figure 3.8, which provides a comparison of tree density estimates obtained from either historical (i.e., late 19th and early 20th century) reconstructions or contemporary reference sites (i.e., unlogged, with an intact fire regime) and contemporary stands within the same study area. When interpreting the magnitude of these differences, it is important to consider the inability of reconstruction studies to accurately account for very small trees in the historical reference period, particularly for red fir, which can decay rapidly (Taylor 2004). Nevertheless, these and other studies show that current tree densities in red fir forests are higher than presettlement tree densities, particularly in stands that have experienced (1) 19th-century logging impacts (Taylor 2004); (2) fire exclusion, especially in lower elevation stands or areas that have missed multiple FRIs (Dolanc et al. 2014b); and (3) increases in daily minimum temperature and precipitation over the past several decades, which may favor increased regeneration and recruitment (Dolanc et al. 2013a). Future climate projections, coupled with continued fire exclusion, could lead to further increases in tree density in the future; however, note that wildfire and mortality trends (described above) may also counteract this trend to some extent.

Current tree densities in red fir forests are higher than presettlement estimates due to 19th century logging impacts, fire exclusion, and increases in daily minimum temperature and precipitation over the past several decades, which may favor increased regeneration and recruitment.

Tree size and size class distribution—

Published studies that describe historical tree size and size class distributions for red fir forests in the assessment area are not available. However, several studies conducted outside of the assessment area, in red fir forests of the southern Cascades and Sierra Nevada, suggest that the size class distribution of red fir forests has shifted toward smaller size classes over time, with an overall decline in large-diameter red fir trees.

In one study of red fir forests in Lassen Volcanic National Park, Taylor (2000) documented an abundance of small trees (diameter at breast height [d.b.h.] <14 cm) that were less than 140 years old, which corresponded to the number of years since the last documented fires (108 to 146 years prior). Repeat photographs taken in the 1920s and early 1990s of these stands provide additional evidence for increases in the density of understory trees, seedlings, and saplings in some stands (Taylor 2000) (fig. 3.10). A similar study conducted by Bekker and Taylor (2010) in the Thousand Lakes Wilderness on the Lassen National Forest also documented increases in understory tree density, which they similarly attributed to the effects of fire suppression. Historically, surface fires in red fir forests would have killed small trees in the understory. Because red fir is shade tolerant and has the ability to regenerate continuously (i.e., even without disturbance), an increase in the time between fires would likely increase the density of smaller trees that may historically have been killed by fire. Dubrasich (2010) found some evidence of this in a red fir stand on the Umpqua National Forest in which backdated stand reconstructions suggest large shifts toward smaller size classes over a period of 185 years of fire exclusion (fig. 3.9).

In a comparison of extensive historical (early 1930s) and modern (USFS Forest Inventory and Analysis) forest inventories in the northern and central Sierra Nevada, Dolanc et al. (2014b) found that the density of smaller (10 to 30 cm d.b.h.) red fir trees increased by about 70 percent (from 91 to 154 trees/ha) over the 70-year evaluation period, while the density of large (>60 cm d.b.h.) trees declined by about 36 percent (from 64 to 41 trees/ha). In a related study of unlogged upper elevation (2300 to 3400 m) forests of the central Sierra Nevada, Dolanc et al. (2013a) estimated that the density of small-diameter red fir trees increased by about 91 percent, while the density of larger (61 to 91 cm d.b.h.) trees marginally decreased by about 20 percent over a period of 73 years. Lutz et al. (2009) documented similar results in upper montane forests of Yosemite National Park where they found that the average density of larger (61 to 91 cm d.b.h.) red fir trees declined between historical (1932 to 1936) and contemporary (1988 to 1999) sampling periods, although declines in the largest trees (>92 cm d.b.h.) were not significant possibly owing to the limited sample size.

Although quite variable, the size class distribution of trees in presettlement red fir forests generally followed a relatively flat or hump-shape distribution rather than the characteristic form of even-aged or uneven-aged stands.

The distribution of tree size classes within a stand is influenced by the interaction of several factors, including the regeneration requirements and strategies of the species in the stand, a site's disturbance history, and variation in environmental conditions (Parker 1992). Data obtained from historical stand reconstructions and contemporary reference sites indicate that although quite variable, the size class distribution of trees in presettlement red fir forests generally followed a relatively flat or hump-shape distribution rather than the characteristic form of even-aged or uneven-aged stands (Meyer et al. 2019, North et al. 2007, Taylor 2004, Taylor and Halpern 1991). This is evident in figure 3.13, which displays data from two studies that compare tree size distributions in current and presettlement or contemporary reference stands. An assessment of field data collected in red fir forests in the Klamath Mountains and southern Cascades (USDA FS, n.d.) found that the distribution of trees in contemporary, unlogged, red fir forests with an active fire regime was relatively flat; whereas stands that had missed at least two FRIs had much higher tree densities in the smaller size classes (fig. 3.13). A similar pattern was documented in reconstructed presettlement red fir and western white pine forests of the Lake Tahoe Basin, where the distribution of size classes followed a hump-shape pattern and very few presettlement trees were <30 cm d.b.h. (Taylor 2004, Taylor et al. 2014). In contrast, contemporary stands in the same study that were logged in

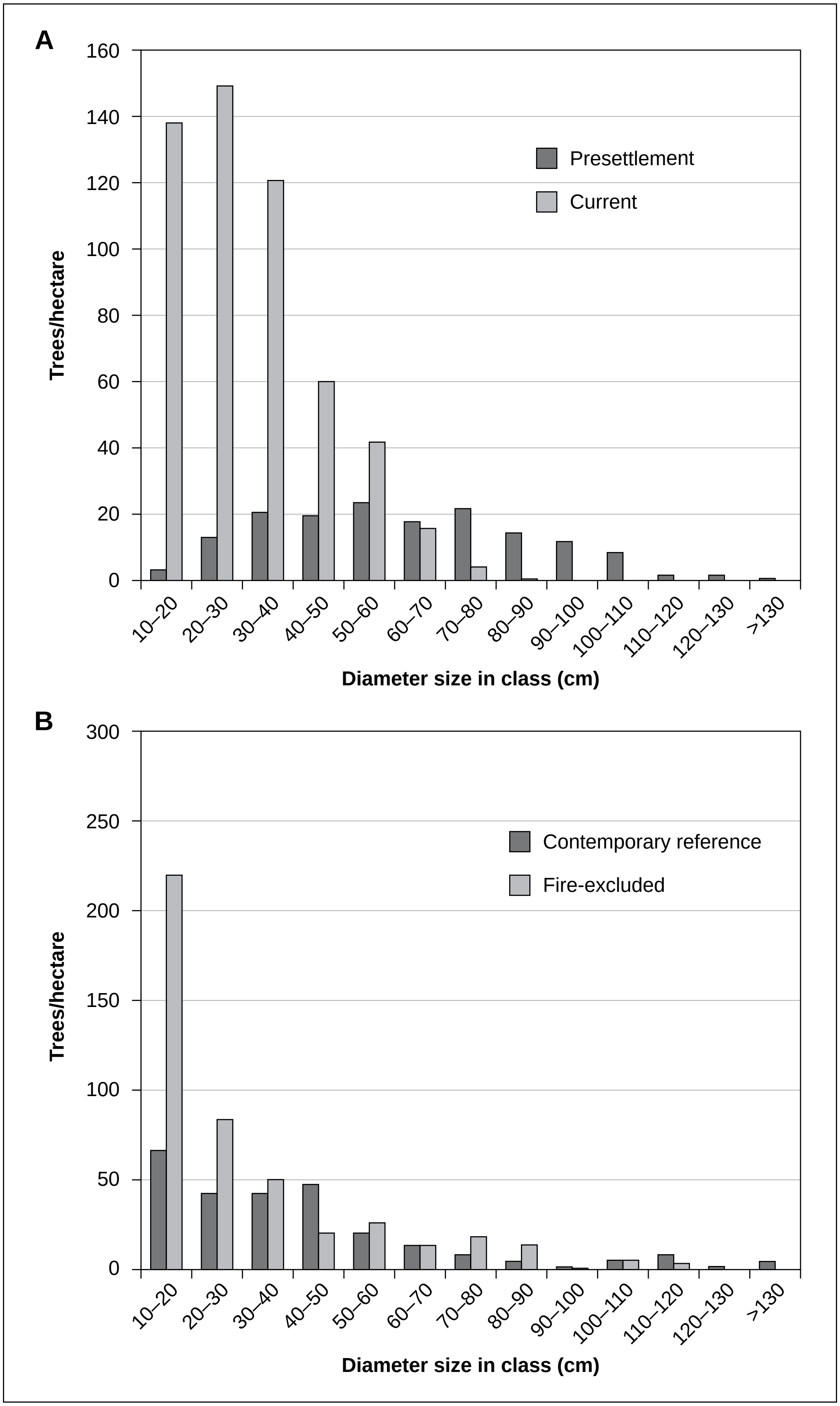

Figure 3.13—Distribution of tree diameter size classes in (A) presettlement and contemporary second-growth red fir–western white pine stands in the Lake Tahoe Basin, Sierra Nevada (data from Taylor 2004); and (B) contemporary reference and fire-excluded red fir stands in the Marble Mountain Wilderness, Klamath Mountains, and Lassen Volcanic National Park, southern Cascades (USDA FS, n.d.).

the late 19th or early 20th century were dominated by trees in significantly smaller size classes (fig. 3.13) (Taylor 2004, Taylor et al. 2014). In Yosemite and Sequoia and Kings Canyon National Parks, Becker and Lutz (2016) documented a greater number of red fir trees in the smallest size classes (i.e., a reverse-J shape distribution) in long unburned plots compared to recently burned plots, which had a low density of trees <15 cm. Maxwell et al. (2014) created a spatially explicit landscape-scale model of presettlement conditions in the Lake Tahoe Basin and determined that less than 0.02 percent of the landscape (all forest types included) was characterized by stands with high relative abundance of trees in the smallest size classes (i.e., <30 cm d.b.h.).

This pattern has also been documented in unlogged contemporary red fir forests after a century of fire exclusion in Crater Lake National Park (Chappell 1991), Thousand Lakes Wilderness (Bekker and Taylor 2010), and Lassen Volcanic National Park (Taylor 2000) in the southern Cascades, as well as in stands in the Sierra Nevada (Potter 1998). In Crater Lake National Park, one site that had evidence of at least four fires over the past 100 years had a much broader distribution of diameter classes than another site in the same study area that had no evidence of fire for at least 190 years and a greater number of trees in smaller size classes (Chappell 1991). In moist mixed conifer forests at Teakettle Experimental Forest in the Sierra Nevada, North et al. (2007) documented a shift away from an almost flat distribution of tree size classes (i.e., nearly equal stem numbers in each d.b.h. class) in 1865 to an uneven stand structure with most trees occurring in the smallest size classes (a reverse-J shape) in contemporary stands. A reverse-J shape size structure in red fir forests can be the result of relatively continuous regeneration in an uneven-aged forest, a two-tiered structure with breaks in the canopy facilitating understory tree growth, or extended survival of suppressed stems in the understory of a more even-aged forest (Parker 1992). Note that not all contemporary, fire-excluded red fir stands are characterized by this size class distribution. For example, Scholl and Taylor (2006) found no evidence of a J-shaped size distribution in unlogged red fir stands in the Lake Tahoe Basin, despite over 180 years of fire exclusion. Regalia (1978) also found evidence of a more even distribution of size classes in his study of unlogged, fire-excluded red fir forests at Haypress Meadows in the Marble Mountains and at north Trinity Mountain in the Trinity Alps. In Lassen Volcanic National Park, Parker (1992) found that reverse-J distributions were more frequent on northern exposures and upland flats; whereas more xeric slopes supported a size class distribution that gradually decreased. One potential hypothesis for the lack of small trees and saplings on xeric red fir sites is drought stress, heat load, or high solar irradiance, which can restrict successful establishment and increase small tree mortality in dry years (Parker 1992).

When compared to contemporary fire-excluded or secondary-growth red fir forests, historical red fir stands also had greater variation in size class distribution among stands. For example, Taylor (2004) compared presettlement diameter distributions in red fir stands in the Lake Tahoe Basin and found that 66 percent of sampled plots had significantly different size-class distributions; whereas 90 percent of sampled plots in contemporary stands had similar size class distributions. In their comparisons of contemporary reference red fir stands at Swain Mountain Experimental Forest, Taylor and Halpern (1991) and Taylor (1993) found significant differences in size class distributions between plots, even though red fir occurred in a wide range of diameter classes in both plots. One of Taylor's (1993) plots had more of a bimodal distribution (i.e., greatest number of individuals <85 cm and >165 cm d.b.h.), while the other had a gradually decreasing distribution (i.e., more individuals <55 cm).

In many contemporary red fir forests, there has been an increase in the density of small-diameter trees and an overall decline in large-diameter red fir trees over the past century.

Current and future deviation from NRV—In many contemporary red fir forests, there has been an increase in the density of small-diameter (<30 cm d.b.h.) trees and an overall decline in large-diameter red fir trees over the past century. This has resulted in a shift in the distribution of tree size classes within stands, generating more stands that are currently dominated by smaller size classes (i.e., a reverse J-shape distribution). These patterns have been attributed to (1) intensive past timber harvest (Taylor 2004); (2) periods of fire exclusion that are far outside the reference FRI (Dolanc et al. 2014b); (3) insect and disease-related mortality, which can be higher in denser stands (DeSiervo et al. 2018); and (4) recent increases in temperature and climatic water deficit associated with climate change, which can lead to higher drought stress in dense stands (Dolanc et al. 2014a). Projected future changes in climate, which may intensify large tree mortality, may intensify the tendency toward a reverse J-shaped size-class distribution in some red fir stands in the assessment area.

Basal area—

Estimates of basal area vary widely in both historical and contemporary red fir stands, with most contemporary red fir forests trending toward higher basal area, but having mean values that fall within the NRV (table 3.4, table A1.3). Averaged across all studies, estimates of historical basal area (mean = 63; range: 46 to 85 m^2/ha) was similar to the mean basal area estimate from contemporary fire-excluded forests (mean = 76; range: 17 to 110 m^2/ha); however, the number of available studies were also too low for a robust comparison (table A1.3). One unpublished study conducted within the assessment area offers limited evidence of a trend toward increasing basal area in contemporary forests. In unlogged red fir stands on the Umpqua National Forest, Dubrasich (2010) noted an almost five fold increase in basal area between 1825 and 2010, from about 11 m^2/ha in 1825 to 50 m^2/ha in 2010. This difference was primarily due to a large increase in small-diameter trees (fig. 3.9). In contrast, a comparison of unlogged red fir stands in both active-fire and long-unburned landscapes found no significant difference in basal area, even with significant increases in overall tree density (Meyer et al. 2019). Taylor et al. (2004, 2014) found no difference in basal area in their study of presettlement and contemporary red fir–western white pine stands in the Lake Tahoe Basin. They attributed this finding to the longer FRIs found in red fir forests, as well as potential limitations in forest reconstruction methods (i.e., the removal of evidence by decay, logging, or fire) (Taylor 2004, Taylor et al. 2014).

Current and future deviation from NRV—Modern basal area estimates for red fir forests in the assessment are likely within or slightly higher than basal area values in presettlement times. Variation in basal area often depends on the relative effects of fire exclusion (that may increase basal area) and logging or other forest mortality agents (that decrease basal area).

Canopy cover—

Canopy cover is commonly defined as the percentage of the ground surface that is shaded by the vertical projection of tree crowns. Unless specifically noted, it is generally a relative metric, where overlapping tree canopies are not double counted (i.e., the maximum canopy cover in a plot is 100 percent) (Safford and Stevens 2017).

Historical accounts and reconstructions of presettlement red fir forest conditions rarely provide estimates of canopy cover. However, data collected from contemporary reference sites suggest that historical red fir forests may have had lower average canopy cover than contemporary fire-excluded red fir stands, as well as a high degree of variability within an individual stand (figs. 3.10 through 3.12). Limited evidence for this general trend can be seen in figure 3.14, which provides a broad comparison of canopy cover estimates derived from contemporary active-fire reference landscapes (NRV) and contemporary fire-excluded stands (current). Included in this figure are field-based estimates of canopy cover from red fir stands in active-fire landscapes in the Marble Mountains of the Klamath Mountains (mean 30 percent, ± 15 percent [SD]); these canopy cover estimates were lower than neighboring fire-excluded areas (mean 47 percent, ± 23 percent [SD,) (USDA FS, n.d.), although this difference was not significant. An analogous comparison of field data in Yosemite and Sequoia and Kings Canyon National Parks found similar results, with significantly lower overstory cover (37 ± 17 percent) in contemporary active-fire reference landscapes compared to adjacent fire-excluded areas (62 ± 18

Historical red fir forests may have had lower average canopy cover than contemporary fire-excluded red fir stands, as well as a high degree of variability within an individual stand.

percent) (Meyer et al. 2019). Barbour and Woodward (1985) reported comparable canopy cover values (mean = 53 percent) for fire-excluded landscapes in their study of 16 unlogged red fir stands in the north Coast Mountains. However, they also documented substantial variability in overstory cover (range 26 to 84 percent) among stands. In a study of two red fir stands at Swain Mountain Experimental Forest, where fire had occurred regularly until 1924, Taylor and Halpern (1991) documented overstory cover values of 54.5 and 46.5 percent. Whittaker (1960) conducted elevational transects in the high-elevation forests of the Siskiyou Mountains and also noted substantial variation in forest cover, ranging from relatively open stands with abundant understory vegetation to dense overstory cover with little to no vegetation in the understory.

The USFS Forest Inventory and Analysis program collected data from 98 red fir stands across the assessment area. Analysis of these more recent data documented a surprisingly lower average canopy cover for contemporary red fir stands (mean 25 percent, ± 20 percent [SD]). This lower value may be due to the method used to calculate canopy cover in Forest Inventory and Analysis plots (see Safford and Stevens 2017 for discussion) or the fact that plots represent an unbiased sample that includes managed (e.g., logged) and unmanaged stands; areas that have burned, including those affected by high-severity wildfire; and low-productivity sites. In another study of contemporary fire-excluded red fir stands in the Klamath Mountains, Laurent et al. (1994) noted that the only openings in the dense canopy of red fir were places where snags or fallen trees had created openings.

A number of studies from the southern Cascades and Sierra Nevada provide evidence that canopy cover in presettlement red fir forests likely varied in response to disturbances like fire, wind, insects, and disease. For example, limited evidence from the assessment area suggest lower canopy cover values (mean: 35 percent, ± 14 percent [SD]) in areas with more frequent fire than areas that have had little to no fire over the past 100 years (mean: 51 percent; ± 11 percent [SD]; see fig. 3.14). Studies conducted in Yosemite National Park found that overstory canopy cover was substantially reduced following moderate- or high-severity fire (Kane et al. 2013), whereas lower fire severities generally reduced cover in the lower canopy layers, reducing understory ladder fuels and raising the average height between the ground and the stand's canopy (i.e., canopy base height) (Collins and Stephens 2010).

In their comparison of reference and contemporary red fir stands in the Lake Tahoe Basin, Taylor et al. (2014) determined that canopy base height was significantly greater (6.6 vs. 0.9 m) and density of available canopy fuel (i.e., canopy bulk density; 0.05 kg/m^3 vs. 0.09 kg/m^3) was significantly lower in presettlement stands than in contemporary stands. Maxwell et al. (2014) used landscape-scale modeling

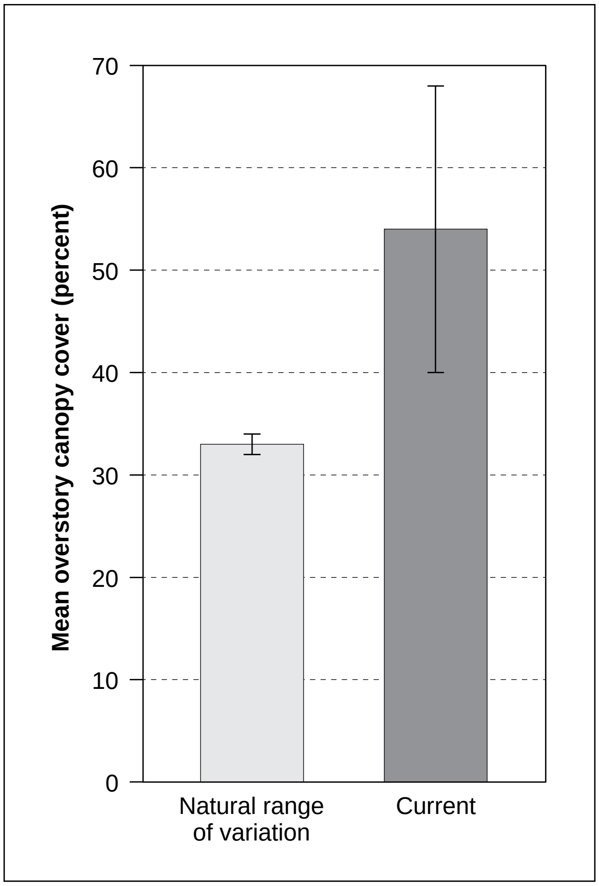

Figure 3.14—Mean (± standard deviation) percentage of canopy cover in stands that are representative of natural range of variation (NRV) (i.e., contemporary reference landscapes) and current red fir stands that are unlogged, but have likely been affected by fire suppression. Canopy cover estimates for contemporary reference stands were based on two datasets from Lassen Volcanic National Park (USDA FS, n.d.) and the Marble Mountain Wilderness (USDA FS, n.d.) and a single study from Swain Mountain Experimental Forest (Taylor and Halpern 1991). Values for contemporary, unlogged, fire-excluded stands were taken from studies conducted in the southern Cascades (Parker 1991) and the north Coast Range (Barbour and Woodward 1985), as well as unpublished data from Lassen Volcanic National Park (USDA FS, n.d.) and the Marble Mountain Wilderness on the Klamath National Forest (USDA FS, n.d.). Standard deviation was derived from averages presented in individual studies and does not represent the full range of variation within or among individual stands.

to derive reference values for three red fir forest types in the Lake Tahoe Basin: (1) stands dominated by small-diameter red fir, (2) medium-large-diameter red fir, and (3) red fir and western white pine. Their models provided slightly lower estimates of presettlement canopy base heights (2.4 m) and a range of mean canopy bulk densities (0.03 to 0.010 kg/m^3); the latter varied by red fir forest type, with higher canopy bulk densities in stands that had higher overall tree density and basal area, smaller diameter trees, and higher proportions of red fir.

Current and future deviation from NRV—Canopy cover comparisons between presettlement and contemporary stands in the assessment area are limited by the scarcity of quantitative information from historical or contemporary reference stands and the high degree of variability among stands. Nevertheless, these limited data, combined with studies documenting substantial increases in tree density over the past century (see discussion above) and concurrent decreases in spatial heterogeneity (i.e., see discussion below), suggest that contemporary fire-excluded red fir stands are likely to have more continuous tree cover than presettlement stands, as well as higher average canopy cover. Shifts in the distribution of tree size classes in

Contemporary fire-excluded red fir stands are likely to have more continuous tree cover than presettlement stands, as well as higher average canopy cover.

contemporary forests toward stands that are dominated by smaller trees also suggest a likely increase in canopy cover in the lower canopy strata.

A limited number of studies from outside the assessment area also suggest that contemporary, fire-excluded red fir forests may have lower canopy base heights and greater canopy bulk density than either presettlement reference stands or contemporary reference landscapes that have burned at low severity. In addition, contemporary red fir forests that have burned at lower severity are more likely to have greater canopy cover in higher strata and larger canopy base heights than those that have burned predominantly at high to moderate severity.

Snags—

A very limited number of studies suggest that snags may have been less abundant in presettlement red fir forests than in contemporary red fir stands. Historical forest inventories of four red fir stands in the central and northern Sierra Nevada (i.e., Sudworth 1899) documented an average of 17.5 snags per hectare (snags >30.5 cm d.b.h.), an average snag diameter of 57 cm, and an average snag basal area of 4.5 m^2/ha (Stephens 2000). For comparison, a study conducted by DeSiervo et al. (2018) in the Russian Wilderness in the Klamath Mountains documented substantially higher snag densities of 42 snags per hectare (snags >30 cm d.b.h.); they attributed this high number of snags to a widespread die-off event caused by interactions between insects, disease, high densities of small trees, and increases in minimum winter temperature.

Although limited by methodological differences in the minimum diameter of measured trees, a small subset of studies suggest higher snag densities in contemporary red fir forests. For example, two studies of unlogged fire-excluded red fir stands in the northern Sierra Nevada documented average snag densities of 96 snags per hectare (trees >5 cm) (Scholl and Taylor 2006) and 46 snags per hectare (trees >1 cm d.b.h.) (Barbour et al. 2002); in the latter study, the majority (67 percent) of dead standing trees were less than 76 cm d.b.h. (Barbour et al. 2002). Snag density and basal area estimates in contemporary, late-seral red fir forests of the southern and central Sierra Nevada were also higher, with averages of 33.4 snags per hectare (snags >1 cm d.b.h.) (Potter 1998) and 12.4 m^2/ha (snags >5 cm d.b.h.) (North et al. 2002), respectively.

Meyer et al. (2019) provided one of the only comparisons of snag densities in contemporary active-fire reference stands and long-unburned red fir forests in the southern Sierra Nevada. They found no significant difference in snag basal area (24.7 m^2/ha in burned vs. 20.3 m^2/ha in unburned) or the density of large (>76.2 cm) snags (13 snags/ha in burned vs. 9 snags/ha in unburned). In a separate study, average snag diameter was found to be similar between historical (57 cm) and contemporary unlogged (50 cm) red fir forests (Potter 1998, Stephens 2000).

Current and future deviation from NRV—The density and size of snags in contemporary red fir forests may be higher than under presettlement conditions; however, considerable variation in both historical and contemporary stands makes determination of this trend difficult. The abundance and distribution of snags in red fir stands is influenced by the intensity and scale of natural disturbance processes, like fire, insects, and disease, as well as the interaction of these processes with climatic conditions (e.g., drought-induced mortality). In presettlement red fir forests, these natural disturbances would likely result in an irregular fine-grained pattern of live and dead trees within a stand, with concentrations of snags present in scattered patches (Arno 2000). In contrast, future projections suggest that the scale and intensity of disturbances, like native insect infestations and wildfire, may be increasing within the assessment area (Lenihan et al. 2003, 2008; Mortenson et al. 2005). In combination with changing climatic conditions, these altered disturbance processes may increase mortality, and, consequently, the density of snags, in future red fir forests.

Tree regeneration—

Natural regeneration in red fir forests can occur beneath existing trees, as well as in openings created by fire, wind, insects, and disease (Barbour and Woodward 1985, Laacke and Tappeiner 1996). Although regeneration is most strongly associated with periodic disturbance, the shade-tolerant nature of red fir also allows for continuous, slower rates of regeneration (Chappell and Agee 1996, Taylor and Halpern 1991). Seedlings that establish beneath a dense forest overstory generally grow very slowly until an opening in the canopy (e.g., from the death of one to two overstory trees) allows suppressed saplings to considerably increase their growth rate (Regalia 1978).

Natural regeneration in red fir stands is positively related to good cone crops (i.e., abundant seed) and favorable microsite conditions, including gaps in the forest canopy, partial shade in the first few years after germination (especially in warmer or more exposed sites), high moisture availability, and bare mineral soil (Fites-Kaufman et al. 2007, Laacke and Tappeiner 1996, Selter et al. 1986). In unlogged, contemporary reference red fir stands in the southern Cascades, Taylor and Halpern (1991) documented higher seedling densities in gaps (mean: 379 seedlings/ha; <33 percent overstory cover) than intermediate (mean 102 seedlings/ha; 33 to 66 percent overstory cover) or closed (mean: 30 seedlings/ha; >66 percent overstory cover) canopy conditions. In their two study stands, more than 70 percent of seedlings established in quadrats that were classified as canopy gaps, which occupied an average of 48 percent of the sampled blocks. In red fir–mountain hemlock stands in Lassen Volcanic National Park, Taylor (1990) also found that canopy openness had a significant influence on the distribution and abundance of seedlings and saplings. In his study sites, both seedlings and saplings were more abundant in open quadrants

(28 percent of quadrants) than closed quadrants (72 percent of quadrants). Taylor (1990) determined that the patchy distribution of canopy gaps likely contributed to the highly clumped distribution of red fir regeneration.

In their study of seedling survival in the southern limit of red fir in the north Coast Range, Selter et al. (1986) determined that higher seedling survival occurred in patches that experienced more midday shade, shorter periods of direct sunlight, and lower surface temperatures from mid-summer to mid-September. In his study of lodgepole pine–red fir forests in Lassen Volcanic National Park, Parker (1993) found positive correlations between red fir seedling densities and canopy cover, and no evidence of red fir regeneration in gaps. However, Parker attributed the discrepancy between this finding and those described above (i.e., Taylor 1990, Taylor and Halpern 1991) to the fact that his study sites were dominated by an overstory of lodgepole pine, which has sparser foliage than red fir, and suggests that gap-phase regeneration may be more important in the latter forest types.

Disturbances like fire can play an important role in patterns of red fir regeneration and establishment (Agee 1993, Cheng 2004). Higher rates of seedling establishment on burned surfaces compared to unburned surfaces have been attributed to exposure of mineral soil, better access to beneficial mycorrhizal fungi, and reduction of competition for belowground resources (Chappell and Agee 1996, Laacke 1990). In his study of contemporary red fir forests in Lassen Volcanic National Park, Taylor (1997) estimated that 50 to 73 percent of red fir had established after the last fire, which had occurred 80 to 140 years prior. Scholl and Taylor (2006) examined tree diameter and age distributions in an unlogged, old-growth forest in the Lake Tahoe Basin and also found evidence of pulses of regeneration that coincided with several past fires. However, they also documented slow, continuous, and variable recruitment, which occurred over a 200-year period without fire. They determined that small canopy openings, associated with the death of single trees or small groups of trees, were a frequent and important small-scale disturbance that influenced regeneration in these stands. In contrast, they found no evidence for the large releases generally associated with large-scale episodic disturbances like windstorms or insect outbreaks (Scholl and Taylor 2006).

The effects of natural disturbances like fire on red fir regeneration can be highly variable and are related, at least in some part, to severity (Scholl and Taylor 2006). Post-wildfire in Crater Lake National Park, Chappell and Agee (1996) found the density of red fir seedlings was greatest in low- and moderate-severity burned patches and lowest in high-severity burned and unburned patches (fig. 3.15). They attributed the scarcity of seedlings in high-severity patches with the increased distance to live trees, which limited seed dispersal, and higher seedling mortality

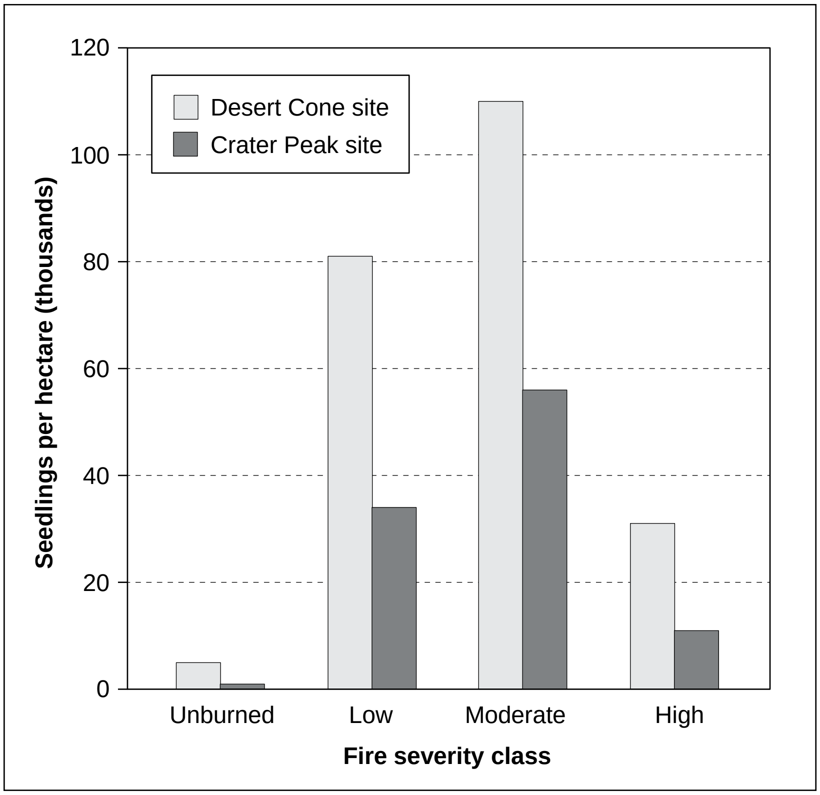

Figure 3.15—Figure recreated from Chappell (1991) showing the density of red fir seedlings by severity class in two fires (sites) in Crater Lake National Park.

owing to increased drought stress, elevated temperatures on the soil surface, high irradiance, and herbivory (i.e., from cutworms) (Chappell and Agee 1996, Agee 1993). The relationship between fire severity and postfire regeneration patterns was also evaluated in lodgepole pine–red fir stands in Lassen Volcanic National Park. In their study, Pierce and Taylor (2011) found high levels of variability among severity classes and no significant difference in red fir seedling densities in low-, moderate-, and high-severity patches; however, they also found no red fir saplings 24 years postfire in high-severity patches. Shrubs have been shown to ameliorate the microclimate in burned sites and allow for fir seedlings to become established and slowly (30 to 50 years) overtop the shrub canopy (Laacke and Tappeiner 1996). In areas where lodgepole pine is present, it can initially dominate high-severity patches; however, red fir regeneration has been observed under mature lodgepole pine, suggesting that over time, these sites can eventually revert back to red fir dominance (Parker 1993, Ziegler 1978 in Agee 1993).

No presettlement estimates of regeneration rates are available; however, studies of contemporary red fir stands suggest that there is considerable variation both within and among red fir forests of the assessment area. In their study of unlogged fire-excluded red fir stands in the north Coast Range, Barbour and Woodward (1985) recorded densities of red fir seedlings, which they defined as trees less than 1 year old, ranging from 2,900 to 94,700 seedlings per hectare (mean = 19,300 seedlings per hectare). These densities were about four times greater than the red fir regeneration that they measured in comparable stands in the southern Sierra Nevada; however; they were not able to attribute these differences to any ecological or genetic factors (Barbour and Woodward 1985).

Current and future deviation from NRV—Quantitative estimates of red fir regeneration are lacking for historical and contemporary reference sites in the assessment area. This, in combination with a high level of variability in regeneration rates both within and among contemporary red fir stands, makes an NRV determination difficult. An understanding of red fir regeneration strategies and how they interact with natural disturbance processes implies that contemporary red fir forests may have lower seedling densities than presettlement forests. For example, the shade tolerance of red fir allows for continuous regeneration in the absence of disturbance; yet several studies suggest that red fir seedling establishment and growth is also strongly associated with periodic fine-scale disturbance. In presettlement red fir forests, disturbances like fire, wind, or insects would have created favorable conditions for regeneration that included moderate-size gaps in the forest canopy or (in the case of low-moderate-severity fire) areas of bare mineral soil. The long absence of fire in many red fir forests of the assessment area has resulted in fewer overstory gaps and more continuous canopy cover in many mature stands (see discussion above). While regeneration is likely occurring beneath the canopy of these forests, it is likely less than what would be expected in a more heterogeneous stand with variably sized canopy gaps and tree clumps.

In presettlement red fir forests, disturbances like fire, wind, or insects would have created favorable conditions for regeneration that included moderate-size gaps in the forest canopy or (in the case of low-moderate-severity fire) areas of bare mineral soil.

Future disturbances such as wildfire and insect outbreaks are projected to increase in scale and severity. These altered disturbance processes may decrease the area within red fir stands that have favorable conditions for natural regeneration. Liang et al. (2017) used species-specific, spatially explicit forest landscape models to evaluate forest response to climate-wildfire interactions under historical and future conditions. Their findings also suggest a decrease in red fir recruitment, with climate warming and wildfires favoring recruitment of drought-tolerant species, like Jeffrey pine, over less drought-tolerant species like red fir.

Understory vegetation cover—

The percentage of understory vegetation cover in presettlement red fir stands was likely low but also highly heterogeneous. In an assessment of field data collected in red fir forests in the Marble Mountain Wilderness, shrub cover was highly variable (ranging from 0 to 60 percent), but tended to be higher in contemporary reference stands with active fire regimes (24 ± 20 percent [SD]) and lower in fire-excluded stands (16 ± 14 percent [SD]), (USDA FS, n.d.). Similarly, herbaceous plant cover was generally greater in these contemporary reference landscapes (16 ± 20 percent) than in adjacent fire-excluded landscapes (2 ± 2 percent; USDA FS, n.d.). Meyer et al. (2019) did a similar comparison in the southern Sierra Nevada and also found significantly higher shrub cover (9 ± 15 percent [SD]) in areas with an active fire regime than in neighboring fire-excluded stands (2 ± 7 percent [SD]). The LANDFIRE project, which relies on vegetation dynamics models and expert opinion to quantify the natural range of variability, estimates that the percentage of cover of shrub and herbaceous species in presettlement red fir forests of the assessment area likely contributed less than 30 percent cover each (LANDFIRE 2017).

In a study of contemporary unlogged fire-excluded red fir stands in the north Coast Range, Barbour and Woodward (1985) found that most stands lacked a significant shrub and herbaceous layer, with an average cover of 1.5 percent and 0.09 percent, respectively. In fire-excluded red fir stands in Lassen Volcanic National Park, repeat photographs taken in the 1920s and 1990s show decreases in shrub cover between the two time periods (fig. 3.11) (Taylor 2000).

Current and future deviation from NRV—Historical information about understory cover in red fir forests is largely lacking. However, general comparisons between contemporary reference sites and current red fir forests suggests that understory cover is likely lower than it would have been under presettlement conditions. This is likely due to an overall decrease in fire frequency, which has resulted in higher and more continuous overstory canopy cover, increased litter and duff depths, and reduced exposure of bare mineral soil. Understory cover in contemporary red fir forests may change in response to future alterations in the scale and severity of disturbance events (e.g., fire) and warming climates; these factors could increase the abundance of early-seral shrubs and herbaceous vegetation in red fir forests understories (Lenihan et al. 2008, McKenzie et al. 2004).

Spatial patterns (horizontal and vertical)—

The historical structure of red fir forests in the assessment area was a mosaic of single trees, tree clusters, and canopy openings, which were associated with variation in disturbance intensity, red fir life history traits, and site conditions (Agee 1993, Taylor and Halpern 1991). Small- and large-scale disturbances, such as fire-, windthrow-, and insect-related tree mortality, played an important role in stand initiation, as well as the creation of small canopy openings that allowed for the release of trees growing under dense overstory canopy (Oosting and Billings 1943, Pitcher

1987, Scholl and Taylor 2006, Taylor and Halpern 1991). Patches of trees, relatively similar in size, ranged from a few hundred square meters to tens of hectares in size, and overlap between these patches created a complex structural mosaic within stands (Taylor 1993, Taylor and Halpern 1991).

No historical descriptions of forest structure were found for red fir forests in the assessment area; however, some accounts do exist for comparable forest types in the southern Cascades and Sierra. In one unpublished study, Alan Taylor retook a series of photographs within and adjacent to Lassen Volcanic National Park that were first photographed in the early 1920s (figs. 3.11 and 3.12). The early photographs show the variability in historical red fir forest structure, with patches of mature trees occurring within a matrix of shrub vegetation. John Muir (1911: 216–217) observed the regularity of canopy gaps and tree clumps in historical red fir forests of Yosemite National Park:

> The principal tree for the first mile or two from camp is the magnificent [red] fir, which reaches perfection here both in size and form of individual trees, and in the mode of grouping in groves with open spaces between…A few noble specimens two hundred feet high occupy central positions in the groups with younger trees around them; and outside of these another circle of yet smaller ones, the whole arranged like tastefully symmetrical bouquets, every tree fitting nicely the place assigned to it as if made especially for it; small roses and eriogonums are usually found blooming on the open spaces about the groves, forming charming pleasure grounds.

Leiberg (1902: 36) observed a similarly high degree of spatial variation in red fir and upper montane forests in the northern and central Sierra Nevada:

> The tendency of the [red fir] tree in the region is toward open, park-like groves…The type as a whole is scattering and patchy. Everywhere along the main divide of the Sierra it is made of blocks of forest, separated by sedgy or weed-covered openings or by tracts of naked rock. In the central district the stands form long thin lines, here widening into a fairly compact or heavy body of timber a few hundred acres in extent, there narrowing into irregular, straggling groups or lines of trees. The great expanses of chaparral which occur almost everywhere throughout this district break and interrupt the stands of the type at frequent intervals. Wet glades and expanses of bare rock are common in these areas, and contribute toward the patchy character of these forests.

These historical observations, coupled with information from contemporary reference forests, suggest that late-seral red fir stands were historically characterized by a high degree of horizontal spatial heterogeneity. In his study of red fir–western white pine stands in the Lake Tahoe Basin, Taylor (2004) found that presettlement forests had greater structural complexity than contemporary stands that had been logged in the late 19th century. He used Shannon's diversity index to examine the number of different tree size classes (measure of richness), as well as the number of trees within each of these size classes (measure of evenness), and found that presettlement red fir stands had significantly greater structural diversity than contemporary stands. Taylor (2004) also investigated tree spatial patterns within his 0.5-ha study plots and found that in both presettlement and contemporary forests the largest trees (≥40 cm at stump height) were most frequently clumped at smaller spatial scales (distances of <9 m and <18 m, respectively) and more randomly distributed at larger scales. In contrast, he found that small and intermediate trees, were more randomly distributed in presettlement stands and more clumped in contemporary stands, regardless of scale (Taylor 2004). This latter finding means that on average, small and intermediate trees in contemporary red fir stands have more neighboring trees (i.e., within 3 to 24 m) than would be expected if trees were randomly located within the stand. In studies of contemporary unlogged red fir forests at Swain Mountain Experimental Forest in the southern Cascades, Taylor and Halpern (1991) and Taylor (1993) also found evidence for a more random distribution of larger trees (>55 cm). In their study sites, seedlings, saplings, and small trees were significantly clumped at spatial scales similar to the size of gaps in the overstory canopy (Taylor and Halpern 1991).

The historical role of disturbance in creating and maintaining spatial patterns in red fir forests can be inferred from a limited number of studies conducted in contemporary reference sites. In one study of unlogged red fir stands in the Lake Tahoe Basin in the Sierra Nevada, Scholl and Taylor (2006) found historical evidence of infrequent, moderate-severity fires that burned unevenly through the stand creating a diverse mosaic of relatively even-aged patches of trees. In their study area, small even-aged patches were distributed throughout the stand but were also separated from one another by other (different) even-aged patches. Scholl and Taylor (2006) attributed these spatial patterns to both small- (e.g., individual treefall) and large-scale disturbances (e.g., moderate-severity wildfire) that created an overlapping mosaic of dead and surviving trees and resulted in patchy regeneration and stand development patterns. In another study of red fir forests in the Klamath Mountains, Regalia (1978) found evidence of even-aged patches (defined as 90 percent of trees falling within 10 years of the plot's mean tree age) in only 23 to 29 percent of sampled plots. The author suggests that this provides evidence for a more uneven-aged structure within the stand where the intermediate to moderately high shade tolerance of red fir allows it to regenerate and survive under a shady overstory, as well as take advantage of small-scale disturbances (one to two trees falling) for release and growth.

Fire suppression has shifted contemporary red fir forest structure from one defined by spatially heterogeneous tree patterns (e.g., with clumps, gaps, and individual trees) to larger continuous areas of closed canopy.

Several recent studies from the southern Cascades and Sierra Nevada suggest that fire suppression has shifted contemporary red fir forest structure from one defined by spatially heterogeneous tree patterns (e.g., with clumps, gaps, and individual trees) to larger continuous areas of closed canopy. Repeat photographs taken in and around Lassen Volcanic National Park in the 1920s and by Alan Taylor in the early 1990s and 2000s suggest a loss of structural heterogeneity in red fir forests and a concurrent increase in forest density and cover over time (figs. 3.10, 3.11, and 3.12).

In contemporary red fir forests that have been affected by high-severity wildfire, larger gaps with little to no overstory cover may also be more common than expected under a presettlement fire regime. Kane et al. (2013, 2014) found evidence of this in their study of contemporary active-fire reference stands in Yosemite National Park where they used light detection and ranging (LiDAR) data to analyze spatial patterns in red fir forests. They identified three structural canopy classes within their study area: canopy-gap, patch-gap, and open-patch (fig. 3.16). Canopy-gap arrangements, which are typically referred to as closed-canopy forest, were characterized by continuous canopy punctuated by frequent and small gaps. These arrangements were most commonly found in unburned and undifferentiated (i.e., no satellite-detected change in postfire vegetation) red fir forests. Patch-gap arrangements were defined as spatially heterogeneous, partially open canopy forest, and had alternating tree clumps and canopy gaps in roughly equal proportions across the landscape. This patch-gap pattern was typical of red fir forests burned at low

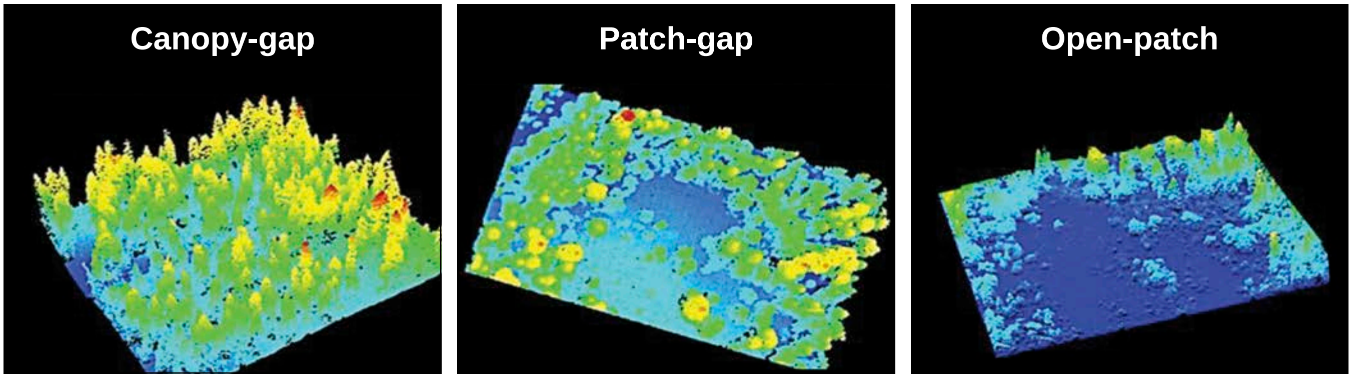

Figure 3.16—Horizontal structural classes identified by Kane et al. (2013, 2014) in burned and unburned red fir forests of Yosemite National Park. Three LiDAR-derived canopy gap structural classes that range from relatively continuous overstory cover punctuated by frequent small gaps (canopy-gap), to relatively equal proportions of tree clumps and canopy gaps (patch-gap), to scattered trees with large openings (open-patch). Figure created by Marc Meyer using FUSION software (McGaughey 2010).

severity. In contrast, open-patch arrangements (i.e., those with large canopy gaps) occurred in portions of the landscape that burned at moderate and high severity and were characterized by scattered trees within large open areas. In general, the total area in canopy decreased, and the number of clumps increased with increasing fire severity in red fir stands of Yosemite National Park (Kane et al. 2013, 2014). In two contemporary reference stands at Swain Mountain Experimental Forest in the southern Cascades, Taylor and Halpern (1991) documented a higher frequency of quadrants in gaps (48 percent of quadrants, <33 percent cover) than in intermediate (26 percent of quadrants, 33 to 66 percent cover) or closed (25 percent of quadrants, >66 percent cover) conditions.

Fire exclusion in contemporary red fir forests may have also altered vertical forest structure. In their study in Yosemite National Park, Kane et al. (2013) found that unburned red fir forests were characterized by an approximately equal mixture of four vertical structure classes: (1) sparse, with low tree densities separated by relatively large areas of understory vegetation (<2 m in height); (2) shorter, predominantly tree covered, but dominated by smaller trees; (3) multistory, characterized by trees of variable height; and (4) top story, with low densities of larger trees and distinct vertical separation between tall trees and lower forest strata (see fig. 5 in Kane et al. 2013). In stands that had burned at low severity, the proportion of red fir in the shorter and multistory classes decreased, while the proportion in the top story and sparse structural classes increased; this finding highlights the capacity of low-severity fire to remove understory ladder fuels while retaining larger trees (Kane et al. 2013). In contrast, areas that burned at moderate or high severity, had a greater proportion of red fir in the sparse and open (i.e., little to no trees, with trees and shrubs primarily less than 2 m in height) structural classes (Kane et al. 2013). These results suggest that contemporary fire-excluded red fir forests may have a relatively lower proportion of top story and sparse structural classes and greater proportion of multistory and shorter (and denser) structural classes than contemporary reference landscapes that have burned in the recent past.

In his study of 34 fire-excluded red fir–western white pine stands in Lassen Volcanic National Park, Taylor (2000) identified three predominant forest structural types that were represented in relatively equal proportions across the landscape: (1) a sparse overstory (65 trees/ha; trees >54 cm d.b.h.) with a dense understory of small trees (2,775 trees/ha; trees <34 cm d.b.h.); (2) a moderately dense overstory (92 trees/ha) and understory (831 stems/ha); and (3) an abundance of larger overstory trees (170 stems/ha) and a sparse understory of small trees (271 stems/ha).

Current and future deviation from NRV—Limited information from historical and contemporary reference sites suggest that presettlement forests were structurally more complex than contemporary second-growth forests (Taylor 2004). In these stands, large tree spatial patterns may currently be within the NRV; however, small- and intermediate-sized trees are generally more spatially homogeneous than expected under presettlement conditions (i.e., distributed in a more clumped rather than random pattern) (Taylor 2004). The alteration of disturbance processes such as fire (see discussion above) has also reduced the fine-scale structural complexity within many contemporary red fir stands. Some red fir forests that have experienced relatively frequent low-severity fire have been shown to exhibit greater structural heterogeneity than comparable unburned stands. In these stands, long periods without fire have led to substantial canopy ingrowth, a more homogenous canopy structure, a substantial reduction of area in canopy gaps, and a general shift toward more coarse-grained patterns (figs. 3.11 and 3.12) (Kane et al. 2013). Trends in fire and tree mortality, which are both projected to increase in scale and severity under future climate scenarios, may also result in less structural complexity and greater homogenization; however, these shifts would be toward more open fragmented forest structure, rather than more continuous closed forest conditions.

Long periods without fire have led to substantial canopy ingrowth, a more homogenous canopy structure, a substantial reduction of area in canopy gaps, and a general shift toward more coarse-grained patterns.

Seral class proportions—

The interagency LANDFIRE Program developed a nationwide map of potential vegetation (referred to as Biophysical Settings [BpS]) representing the distributions of vegetation types as they are hypothesized to have existed before European-American settlement. LANDFIRE BpS modeling suggests that historical reference conditions in red fir forests within the assessment area were dominated by mid- and late-seral stages (fig. 3.17). Within these mid- and late-seral stages, red fir forests of the assessment area had relatively equal proportions of open (<40 percent cover) and closed canopy conditions (>40 percent cover) (LANDFIRE 2017); this is in contrast to red fir forests in the southern Sierra Nevada, which were generally dominated by open canopy conditions (fig. 3.17).

Current and future deviation from NRV—Information describing the NRV in seral stage is limited for red fir forest of the assessment area. Haugo et al. (2015) used the LANDFIRE BpS modeling framework to compare the distribution of seral stages in contemporary red fir forests in the Cascade Range and Klamath Mountains of Oregon with the estimated NRV distribution. Although there were some differences between moist and dry red fir forest types, Haugo et al. (2015) found that there was an overall deficit of the open-canopy, late-seral class and a surplus of closed-canopy, mid-seral class in contemporary red fir forests (fig. 3.18).

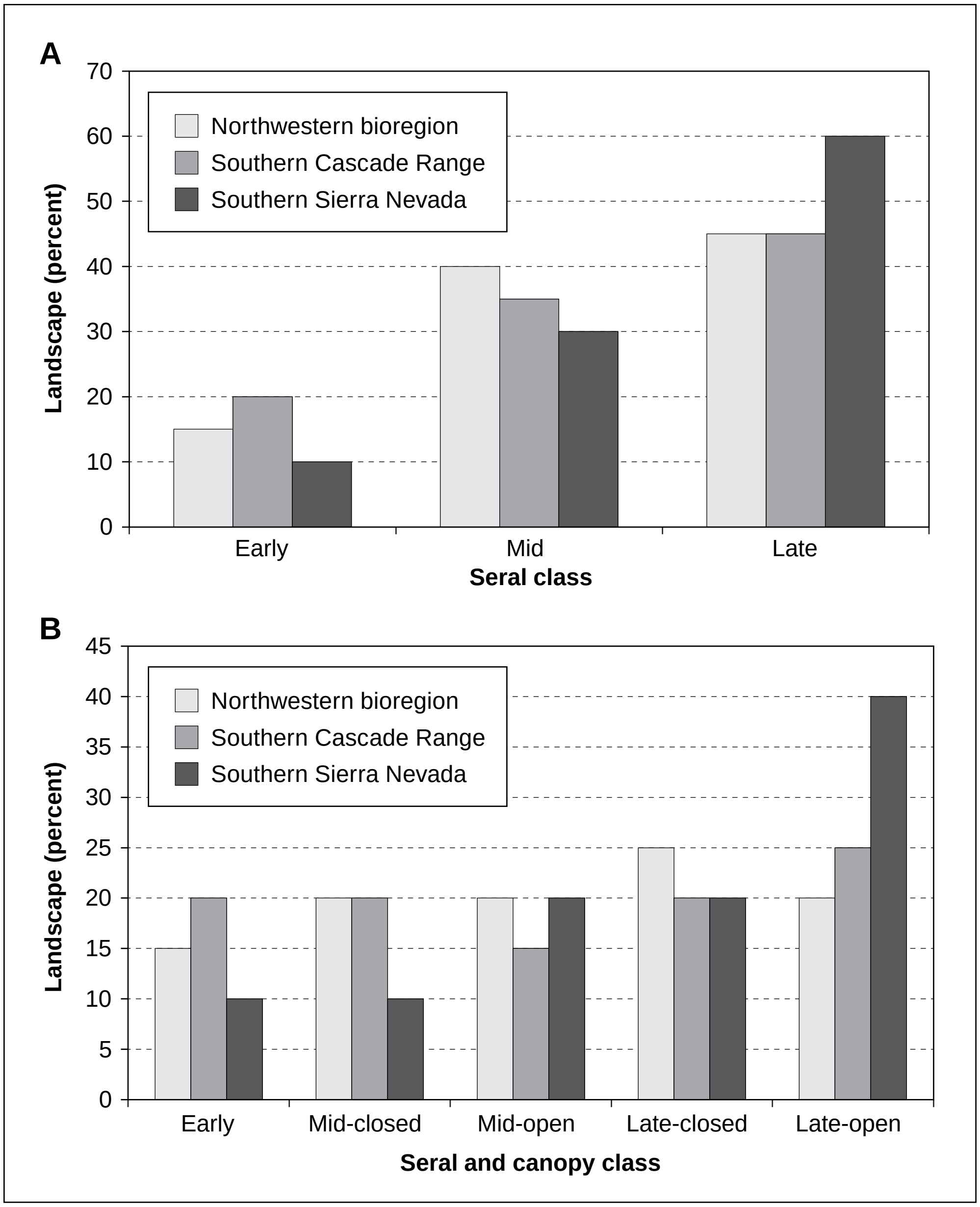

Figure 3.17—Percentage of the landscape occupied by red fir forests in different seral stages based on LANDFIRE Biophysical Setting models for the assessment area, southern Cascades, and southern Sierra Nevada (LANDFIRE 2017). The bottom figure displays the open and closed canopy subclasses within mid- and late-seral classes.

In many cases, fire is required for the development from early- to late-open-seral stage conditions; whereas the mid-closed-seral stage can develop from early-seral conditions in the absence of fire (Haugo et al. 2015). These data, in combination with the studies described above, which indicate substantial increases in tree density, shifts in size class distributions to smaller size classes, increases in canopy cover, and decreases in open stand conditions, suggest that the distribution of seral

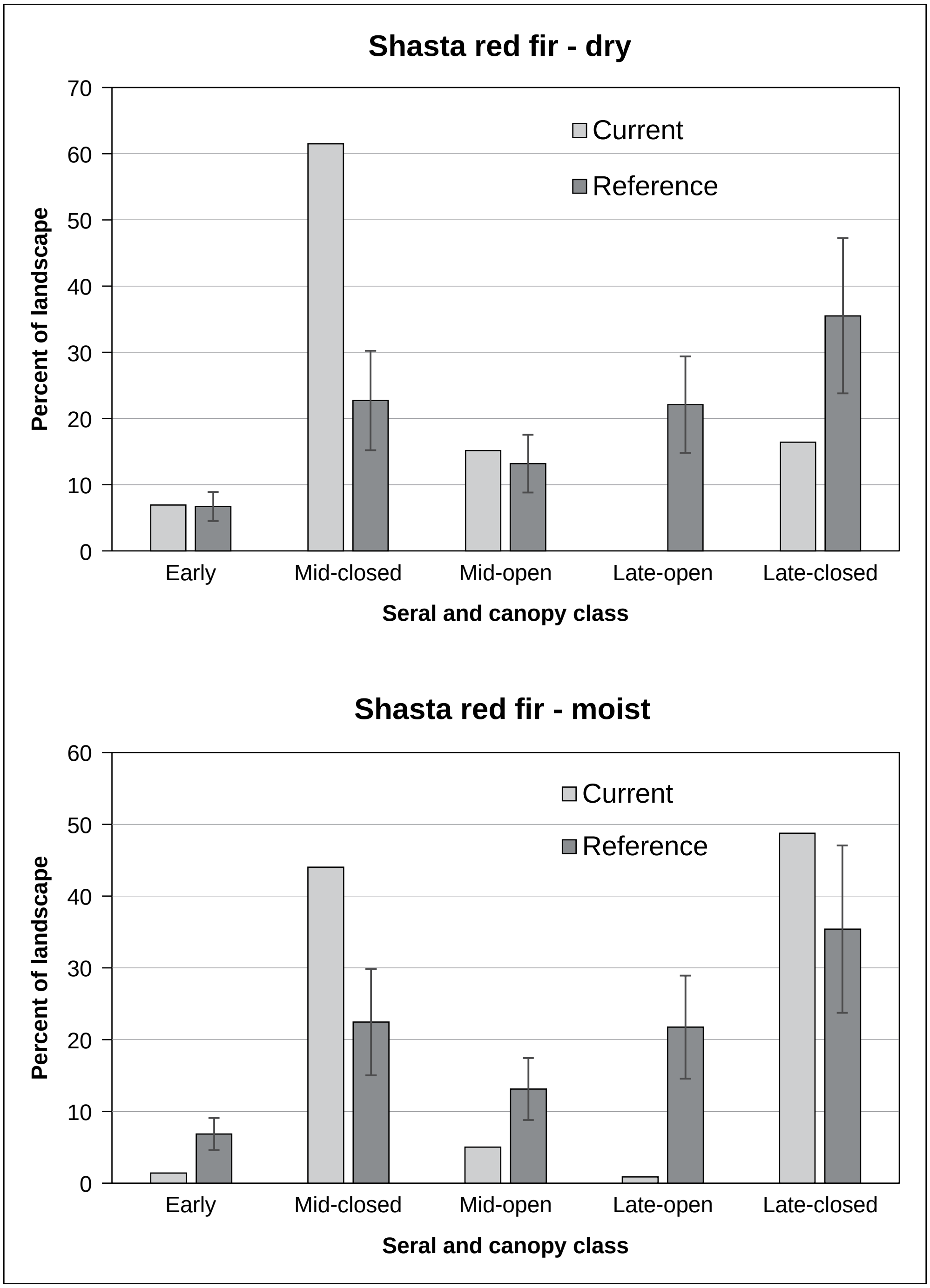

Figure 3.18—LANDFIRE Biophysical Settings models for southwest Oregon and central Cascades of Oregon. Results extracted from Haugo et al. (2015).

stages in contemporary red fir forests is currently outside of NRV. Continued fire exclusion could lead to further increases in the proportion of closed-canopy seral stages across the landscape; however, future wildfire and tree mortality projections (described in the sections above) may also counteract this trend, increasing the proportion of early-seral stages within the assessment area.

Composition

Overstory species composition—

Red fir maintains a high relative density and basal area in both historical and contemporary reference red fir forests of the assessment area (figs. 3.19 and 3.20). This includes mixed red fir–white fir, red fir–mountain hemlock, and red fir–western white pine forests that generally have a lower and more variable relative contribution and dominance of red fir than pure red fir stands. The ratios of different tree species and relative dominance of red fir varies with elevation; in the lower elevation transition zones, red fir intermixes with species from the lower mixed-conifer forests (e.g., yellow pine and white fir), while in the upper elevations, associated species include mountain hemlock, lodgepole pine, western white pine, and other subalpine forest trees. The limited number of quantitative estimates from historical and contemporary reference sites (table 3.34, table A3) suggests that the relative proportion of red fir either did not change or slightly decreased between historical and contemporary time periods in the assessment area. This aligns with other studies, which have suggested that species composition in red fir forests has likely not changed substantially over the past century (Meyer et al. 2019; Dolanc et al. 2014a, 2014b; Taylor 1990, 1997). Studies of contemporary unlogged red fir stands in the

Photo was taken as part of Vegetation Type Mapping survey of California conducted by Albert E. Wieslander and was provided courtesy of the Marian Koshland Bioscience and Natural Resources Library, University of California, Berkeley, http://www.lib.berkeley.edu/BIOS/vtm/.

Figure 3.19—Historical 1949 photograph of a red fir–white fir stand along Horse Creek road, about 3.2 km from the Siskiyou Mountain Summit. Red fir was estimated to comprise about 70 percent of the stand.

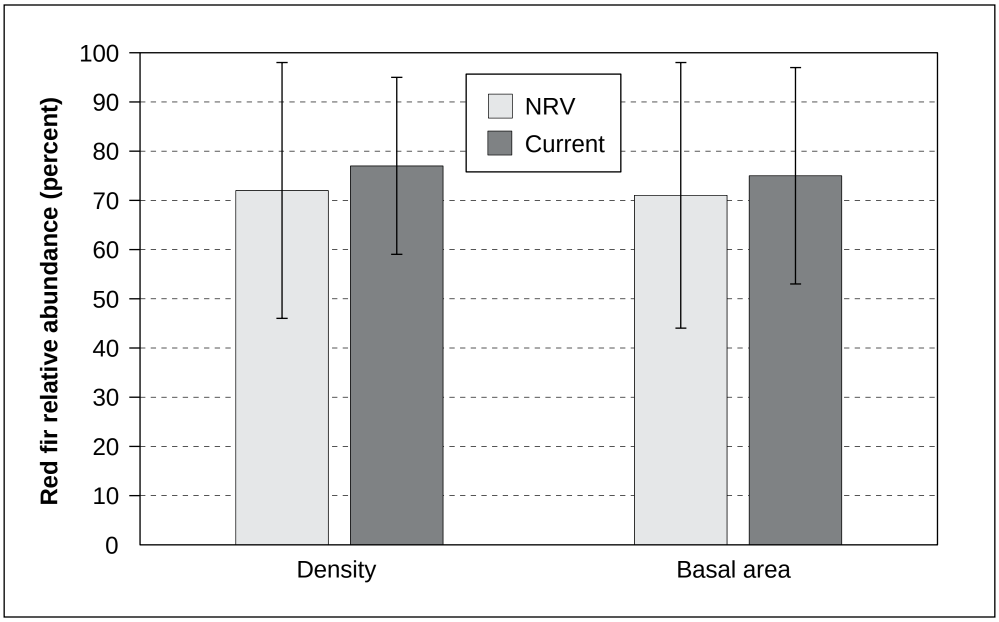

Figure 3.20—The relative abundance (percentage) of red fir in stands that are representative of natural range of variation (NRV) (includes both historical and contemporary reference landscapes) and contemporary red fir stands in areas that have not been logged, but may have been affected by fire suppression.

southern Cascades and north Coast Range have also shown continuing dominance of red fir in the overstory, despite almost a century of fire exclusion (Barbour and Woodward 1985, Bekker and Taylor 2001, Taylor 2000, Taylor and Solem 2001).

Within some of these red fir stands, there is evidence that the relative density of red fir has shifted, most likely as a result of logging practices or high-severity wildfires to which shade-intolerant species such as lodgepole pine initially are prone (Fites-Kaufman et al. 2007; Parker 1991, 1993). For example, a comparison of historical and contemporary red fir–western white pine stands in the Lake Tahoe Basin showed evidence of an increase in the relative density of lodgepole pine following late 19th century logging (Taylor et al. 2014, 2016).

The USFS Forest Inventory and Analysis program collected data from 98 red fir stands across the assessment area between 2001 and 2014. These data represent an unbiased sample of current conditions in red fir forests and include plots in both managed and unmanaged stands, as well as those burned by wildfire. Analysis of these data shows substantially lower relative density and basal area estimates for red fir (mean 48 to 49 percent of total tree density and basal area of all species) compared to historical and contemporary reference stands, which averaged 81 percent (table 3.4). This may be because the most common species in the sampled stands after red fir was white fir, which had a mean relative abundance that ranged from 31 to 37 percent, suggesting that many of these contemporary stands may have been located in the red fir–white fir ecotone that is more common at lower elevations.

Current and future deviation from NRV—Comparisons between historical and contemporary red fir stands in the assessment area indicate that overstory species composition (i.e., the relative abundance of red fir) is currently within the NRV.

Understory species composition—

Similar to contemporary stands, understory species composition in historical red fir forests varied across the assessment area in response to underlying environmental conditions, disturbance processes, and the influence of the overstory tree canopy. Although widely variable, red fir forests generally support a less diverse understory flora than many lower elevation forests, and this appears to be valid for both historical and contemporary stands (Barbour and Woodward 1985, Laacke and Tappeiner 1996). This is often the result of dense shade and thick, compacted litter accumulation beneath mature red fir canopies, which restricts the growth of understory plants (Fites-Kaufman et al. 2007).

Observations from contemporary reference sites in and adjacent to the assessment area suggest that the understory in presettlement red fir forests was heterogeneous. Understory species abundance and diversity likely ranged from sparse to dense and well developed, and was often dependent on past disturbance events (Cheng 2004). For example, in Crater Lake National Park, Chappell (1991) estimated species richness in red fir stands after two fires and observed the highest number of shrubs and herbaceous species in areas that had burned at moderate to high severity (statistical differences were not tested) (fig. 3.21). In their comparison of active-fire and fire-excluded red fir forests, Meyer et al. (2019) documented greater understory species diversity in burned than in unburned stands.

In a study of unlogged red fir stands in the north Coast Range, Barbour and Woodward (1985) found that most stands lacked a significant shrub and herbaceous layer. They documented 10 shrub species in their plots with an average cover of 1.5 percent; the most common shrub was the low-growing pinemat manzanita (*Arctostaphylos nevadensis*). Barbour and Woodward (1985) also documented 22 species of herbaceous plants (average cover = 0.09 percent) with white veined wintergreen (*Pyrola picta*) being the most common. They compared species richness in these stands to red fir stands in the southern Sierra Nevada, and although not significantly different, they noted that the north Coast stands had higher shrub species richness and lower herbaceous species richness (Barbour and Woodward 1985). In a recent assessment of red fir in the Sierra Nevada, Meyer and North (2019) noted that shrub and herbaceous understory species that were frequent historically were still relatively common in contemporary red fir forests (Oosting and Billings 1943, Potter 1998).

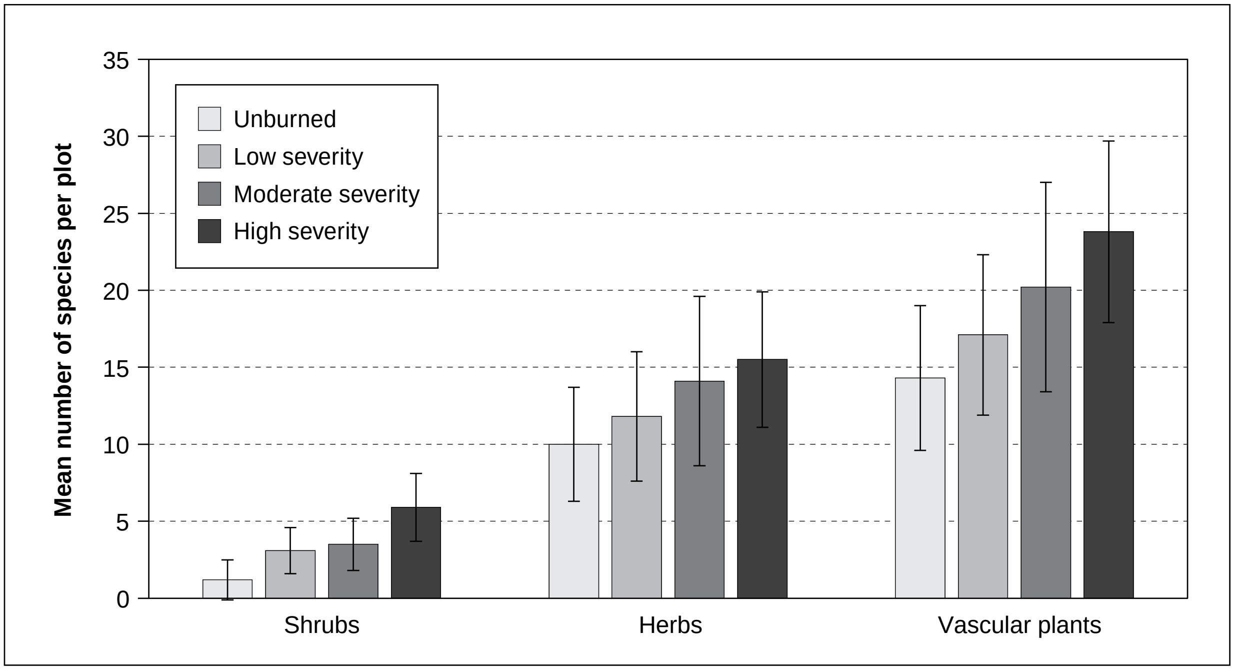

Figure 3.21—Mean species richness (± standard deviation) in relation to fire severity after two wildfires (Desert Cone and Crater Peak) in red fir forests at Crater Lake National Park. Data reproduced from Chappell (1991).

Current and future deviation from NRV—Historical information about understory species in red fir forests is largely lacking. Early observers focused on documenting tree species, making only passing reference to herbaceous understory species and shrubs. However, general comparisons between contemporary reference sites and fire-excluded contemporary red fir forests suggests that understory species composition in red fir forests is relatively similar between historical and current stands.

Chapter 4: Subalpine Forests

Geographic Distribution and Physical Setting

Geographic Distribution

Subalpine forests in the assessment area represent the highest distinct forest zone, occupying the transitional zone between forested and nonforested alpine landscapes (Agee 1990). Subalpine forests are restricted within the assessment area owing to the limited amount of high-elevation habitat (fig. 4.1). Elevations in the region rarely exceed 2800 m, so many mountain peaks and ridges are dominated by red or white fir forests rather than alpine or subalpine vegetation. Subalpine forests generally occupy areas that exceed 1500 m and are situated above the upper montane forest zone where they grow to the tops of the highest peaks and ridges; true alpine vegetation is uncommon in the region (Sawyer 2007, Tesky 1992). In California, the subalpine forests of the Klamath Mountains and north Coast Mountains collectively represent about 20 percent of the state's total area of subalpine habitat (Millar and Rundel 2016). In the Klamath Mountains, subalpine forests occupy high ridgetops above 2100 m in the Siskiyou, Marble, Salmon, and Trinity Mountains. This forest type is most restricted in the north Coast Range, where it is limited to the highest elevations in the north and south Yolla Bolly Mountains (Sawyer 2007). Subalpine forest is scattered throughout the southern Cascade Range of Oregon and California. The southern Cascades of California represent a small fraction (1 percent) of the state's subalpine forest, with most of the habitat found in narrow bands around Mount Shasta and Mount Lassen (Millar and Rundel 2016). In general, subalpine forests are found at lower elevations in the more northern latitudes of the assessment area, and at slightly lower elevations in the western mountain ranges (i.e., the Klamath Mountains) than the eastern ranges (i.e., the southern Cascades) (Millar and Rundel 2016).

The subalpine forests of the assessment area are noted for their high floristic diversity. This is thought to result from variation in parent material, climatic gradients, and disturbance regimes, as well as their geographic location at the intersection of the Pacific Northwest, California, and Great Basin floras (Sawyer 2007, Whittaker 1960). Most subalpine stands are characterized by less than three conifer species; however, there are a few notable exceptions. For example, up to 10 conifer species have been documented in subalpine stands near Cliff Lake and Twin Lakes in the Trinity Mountains (Sawyer 2007).

Mountain hemlock is frequently found in subalpine forests of the assessment area. Although it is relatively restricted in the Coast Range, where it is limited to only a few stands, mountain hemlock is common in subalpine forests of the Klamath Mountains and southern Cascades (Sawyer et al. 2009). Within subalpine

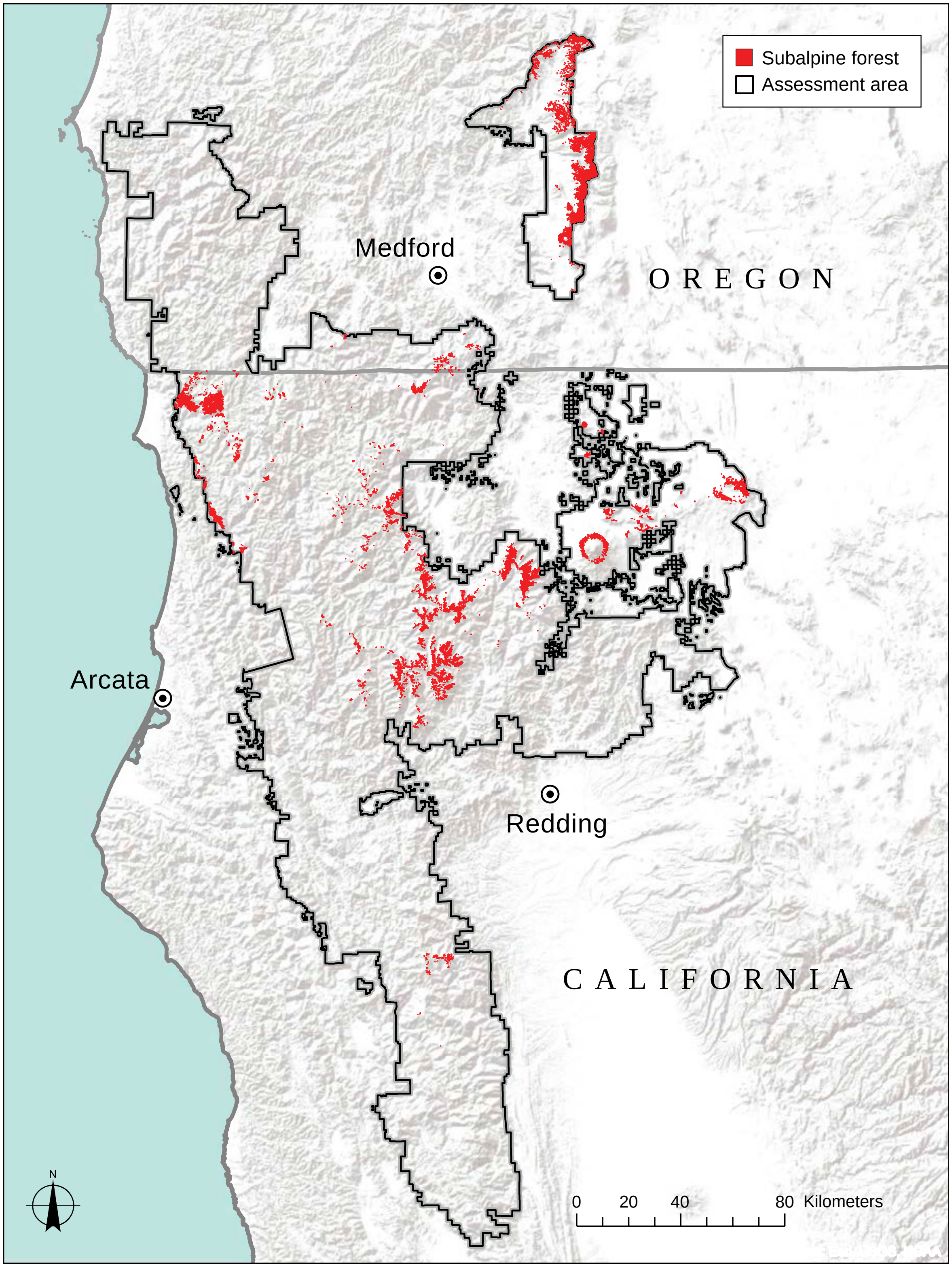

Figure 4.1—Map showing the distribution of subalpine forests in the assessment area. Data were obtained from U.S. Forest Service polygons of existing vegetation. Map displays forest types designated as subalpine conifer (California Wildlife Habitat Relationships type = SCN) in California and mountain hemlock (forest type contains mountain hemlock in combination with several other subalpine species) in Oregon.

forests, mountain hemlock co-occurs with red fir, lodgepole pine, western white pine, and noble fir (Sawyer et al. 2009). In his study of subalpine forest patterns in the Siskiyou Mountains, Whittaker (1960) observed that mountain hemlock and noble fir were often the only species represented in high-elevation stands, with noble fir tending to dominate lower, more xeric sites and mountain hemlock generally dominating higher, more mesic sites.

Several subalpine tree species have restricted or unique geographic distribution patterns in the assessment area (table 4.1). Foxtail pine is a California endemic with disjunct populations in the southern Sierra Nevada and the assessment area, where it occurs in the Klamath Mountains and isolated stands atop the South and north Yolla Bolly Mountains in the north Coast Range (Griffin and Critchfield 1972, Sawyer et al. 2009). Pacific silver fir, subalpine fir, and Alaska yellow-cedar (*Callitropsis nootkatensis*) reach the southern limit of their distribution in northwestern California where they are only found in isolated groves in the Klamath Mountains (Critchfield 1972 and Sawyer 2007, Griffin). Scattered stands of whitebark pine occur on exposed ridge crests and peaks in Shasta County in the southern Cascades and a few isolated localities in the Klamath Mountains (Griffin and Critchfield 1972). Brewer spruce is scattered throughout the montane and subalpine forests of the Klamath Mountains, but is generally restricted to local, disjunct populations of differing sizes (Sawyer 2007). Engelmann spruce is limited to a few stands in the Klamath Mountains and southern Cascades, where it occurs in glaciated valley bottoms at montane and subalpine elevations (Sawyer et al. 2009).

Climatic Associations

Variation in temperature and precipitation gradients shape the composition and structure of subalpine forests at both local and regional scales. Subalpine forests are characterized by prolonged winter snowpack (usually deeper than 2 m), a short growing season (6 to 9 weeks), and cool summer and cold winter temperatures, with frost possible any month of the year (Agee 1993). Although these forest types occur within a Mediterranean climate region, the high elevations greatly moderate this regional climate influence. Both temperature and evaporative demand decrease with increasing elevation, which reduces stress during the otherwise dry summer months (Millar and Rundel 2016). The assessment area's topography and proximity to the Pacific Ocean also create a significant amount of local and landscape-scale variation (Whittaker 1960). For example, in the Mount Jefferson Wilderness Area on the Deschutes National Forest in central Oregon, the mountain hemlock zone extends lower in elevation on north-facing slopes than south-facing slopes (1524 m compared to 1707 m, respectively); this pattern is related to moisture and snow

Subalpine forests are characterized by prolonged winter snowpack (usually deeper than 2 m), a short growing season (6 to 9 weeks), and cool summer and cold winter temperatures, with frost possible any month of the year.

Table 4.1—Classification and rarity of indicator species in the subalpine forests of northwestern California

California Forest Alliance[a]	CalVeg type[b]	CWHR type[c]	Rarity[d]	Range in bioregion[a]	Elevation range[a]
					meters
Abies amabilis (Pacific silver fir forest)	Klamath mixed conifer	Klamath mixed conifer	G5 S2 CNPS 2B.3	Klamath Mountains (Siskiyou County)	1700–2100
Abies lasiocarpa (subalpine fir forest)	Subalpine fir	Klamath mixed conifer	G5T5 S3 CNPS 2B.3	Klamath Mountains (Siskiyou County)	1700–2200
Callitropsis nootkatensis (Alaska yellow-cedar stands)	Alaska yellow-cedar	Klamath mixed conifer	G5 S4 CNPS 4.3	Klamath Mountains (Del Norte, Siskiyou Counties)	1500–2150
Picea breweriana (Brewer spruce forest)	Brewer spruce	Klamath mixed conifer	G3 S2	Klamath Mountains	1150–2700
Picea engelmannii (Engelmann spruce forest)	Engelmann spruce	Klamath mixed conifer, subalpine conifer	G5 S2 CNPS 2B.2	Klamath Mountains, southern Cascades (Shasta, Siskiyou Counties)	1200–2100
Pinus albicaulis (whitebark pine forest)	Whitebark pine	Subalpine conifer	G5 S4	Klamath Mountains, southern Cascades	2220–3660
Pinus balfouriana (foxtail pine woodland)	Foxtail pine	Subalpine conifer	G3 S3	Klamath Mountains, north Coast Range	2100–3700
Pinus monticola (western white pine forest)	Western white pine	Subalpine conifer	G5 S4	Klamath Mountains, southern Cascades	2000–2300
Tsuga mertensiana (mountain hemlock forest)	Mountain hemlock	Subalpine conifer	G5 S4	Klamath Mountains, north Coast Range, southern Cascades	1900–2700
Pinus contorta ssp. *murrayana* (lodgepole pine forest)	Lodgepole pine	Lodgepole pine	G5 S4	Klamath Mountains, southern Cascades	1500–3400

[a] A Manual of California Vegetation (Sawyer et al. 2009).

[b] U.S. Forest Service’s Classification and Assessment with Landsat of Visible Ecological Groupings system.

[c] California Wildlife Habitat Relationships System (Mayer and Laudenslayer 1988).

[d] Inventory of Rare and Endangered Plants (California Native Plant Society [CNPS] 2016): global (G) rankings: G3 = vulnerable to extirpation or extinction; G5 = secure, considering populations outside California; state (S) rankings: S2 = imperiled; S3 = vulnerable; S4 = apparently secure within California; CNPS rarity ranking: 2B.2 = endangered/rare and moderately threatened in California; 2B.3 = endangered/rare and not very threatened in California; CNPS 4.3 = uncommon with limited distribution and not very threatened in California.

depth, with more long-lasting snowpack limiting the upper distribution of the fir zone (Simon 1991). In the Siskiyou Mountains, mountain hemlock forests are generally confined to cool, north-facing, cirque-like topography with shallow soils (Atzet and Wheeler 1982). They can occasionally occur below 1600 m (5,400 ft) where snow drifts are found through July; however, the lower elevational limit is generally governed by temperatures and competition with red fir (Tesky 1992). Brewer spruce is absent from the drier, eastern portions of the assessment area because it is adapted to cool temperatures during the growing season (Ledig et al. 2012).

Subalpine forests receive most of their annual precipitation in the form of winter snow. In California subalpine habitats, winter snowfall is generally highest in the northern part of the state, including the Klamath Mountains and southern Cascades (Millar and Rundel 2016). Subalpine regions that are closest to the Pacific Ocean, such as those in the Marble Mountains and Yolla Bolly Mountains, generally receive more annual precipitation (including winter snowpack) than inland mountain ranges at the same latitudes (Millar and Rundel 2016).

Geology and Soils

Subalpine forests occur on soils derived from a variety of parent materials, including those of volcanic, sedimentary, metamorphic, and glacial origin (Means 1990). Soils in these forest types tend to be thin and of low quality (Verner and Purcell 1988). Mountain hemlock forests in the Siskiyou Mountains are best developed on loose, coarse-textured, well-drained soils with adequate moisture (Means 1990). In contrast to most subalpine forest types, which generally occur on dry, well-drained soils, subalpine fir and Engelmann spruce occur on rocky, moist moraines and are often considered to be indicators of high soil moisture (Verner and Purcell 1988). Ultramafic soils can support open subalpine woodlands with mixtures of several conifers, including whitebark, western white, and foxtail pine, which are found both on and off of ultramafic substrates (fig. 4.2) (Sawyer 2007, 2009). In one study of subalpine forests in the Klamath Mountains, Eckert and Sawyer (2002) noted that whitebark pine was primarily found in stands on high-elevation peaks with granitic or glacial till substrates.

Landscape Patterns

Subalpine landscapes consist of a mosaic of subalpine forests and woodlands, rock outcrops, scrub vegetation, meadows, and riparian ecosystems (fig. 4.3) Fites-Kaufman et al. 2007). Subalpine forests typically occupy less than half of this landscape, especially at higher elevations. Granitic and other rock outcrops can constitute a substantial proportion of subalpine forest landscapes, creating large patches of open and treeless areas that support sparse vegetation cover. This is particularly apparent at higher elevations and in the drier portions of the assessment area, east of the Cascades, where harsher

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Figure 4.2—Subalpine forests found on ultramafic soils within the assessment area. Clockwise from upper left: foxtail pine growing on the slopes above Deadfall Lake near Mount Eddy, Shasta-Trinity National Forest, elevation 2438 m; whitebark pine growing on ultramafic soils near China Mountain, Shasta-Trinity National Forest, elevation 2590 m; western white pine and foxtail pine along the contact zone between ultramafic and granitic soils near Stonewall Pass in the Trinity Alps Wilderness, Shasta-Trinity National Forest, elevation 2280 m; foxtail pine and western white pine (near ridgetops) on ultramafic soils near Mount Eddy on the Klamath National Forest, elevation 2438 m.

environmental conditions limit forest productivity and biomass. The underlying physical template and corresponding soil development and moisture patterns largely drive this variation within subalpine landscapes, resulting in a heterogeneous mixture of contiguous groves, open woodlands, individual tree clusters, shrub patches, wet and dry meadows, tree islands, and riparian corridors (Keane et al. 2012, Potter 1998).

The upper limit of the subalpine forest zone, often referred to as the treeline, is governed by average temperature during the growing season (Korner and Paulsen 2004, Millar and Rundel 2016). In very general terms, the upper limit of the subalpine forest zone is higher where local conditions are warmer (e.g., south-facing slopes) and varies by both latitude and regional climatic regimes (Millar and Rundel 2016). Unlike the treeline, the lower elevational limits of the subalpine forest are much less distinct and are generally influenced by snowpack dominance and shifts in disturbance regimes (Millar and Rundel 2016).

Julie A. Kierstead

Figure 4.3—Red fir, white fir, western white pine, and mountain hemlock near Kidd Creek, Trinity Alps Wilderness, Klamath National Forest, elevation 2042 m.

Ecological Setting

Indicator Species and Vegetation Classification

Several tree species can dominate the canopy of subalpine forests, either singly or in combinations of two or more, depending on the locality (Verner and Purcell 1988). These species include mountain hemlock, whitebark pine, western white pine, and lodgepole pine (Agee 2007). The most common subalpine species in the region is mountain hemlock. Because lodgepole pine is found in many other habitats that occur outside of the subalpine zone, its occurrence alone is not considered an indicator of subalpine habitat. Subalpine forests intergrade with red fir and Jeffrey pine at lower elevations, and noble fir in the northwestern portion of the assessment area. Less common associates include Alaska yellow-cedar, Brewer spruce, Engelmann spruce, foxtail pine, Pacific silver fir, and subalpine fir. In a limited number of places, these less common species can be indicators of subalpine habitat. For example, in the Trinity Alps and Marble Mountains of northwestern

California, several small stands of subalpine fir are indicative of the subalpine zone at these locations (Millar and Rundel 2016). In the Siskiyou Mountains, several tiny stands of Alaska yellow-cedar are also indicators of subalpine habitat.

Three vegetation types in the assessment area are classified as subalpine forest under the California Wildlife Habitat Relationships (CWHR) system: subalpine conifer, Klamath mixed conifer, and lodgepole pine (table 4.1) (Verner and Purcell 1988). The U.S. Forest Service's (USFS's) CalVeg (Classification and Assessment with Landsat of Visible Ecological Groupings) system identifies 10 subalpine vegetation types (table 4.1). Sawyer et al. (2009) also recognizes 10 vegetation alliances, with more than 40 plant associations. Subalpine alliances include subalpine forests (whitebark pine, lodgepole pine, western white pine, mountain hemlock, etc.), woodlands (foxtail pine), and unique stands (Alaska yellow-cedar) (see table 4.1).

Ecological Importance of Subalpine Forests

Subalpine forests provide a diverse array of ecosystem services, including watershed protection, soil formation, erosion control, carbon sequestration, and habitat for an array of species (Keane et al. 2012). One of the greatest ecosystem services provided by subalpine watersheds is water storage capacity (in the form of snowpack) and delivery of water to lower elevation habitats (Millar and Rundel 2016).

Subalpine ecosystems also provide habitat for many ecologically important and unique species. Tree species such as whitebark pine are considered keystone or foundation species in many high-elevation ecosystems throughout the Western United States (Tomback and Achuff 2010). Among its many keystone functions, whitebark pine provides an important food resource for numerous birds and mammals, plays an important role in enhancing and retaining snowpack and reducing soil erosion, and facilitates succession after disturbance by facilitating the growth of other establishing conifers and understory plants (Tomback et al. 2001, Farnes 1990). Of the more than 6,500 plant species that occur in California, an estimated 1,710 species are found in high elevations of the north Coast Range, and 1,860 species are found in high-elevation habitats of the southern Cascades (Millar and Rundel 2016). Although subalpine forests support a less diverse fauna than lower elevation terrestrial ecosystems, several wildlife species depend on subalpine forests for foraging, nesting, or denning (Verner and Purcell 1988). Subalpine forests are particularly important for uncommon species such as Pacific marten (*Martes caurina*), snowshoe hare (*Lepus americanus*), and in the eastern portion of the assessment area, heather vole (*Phenacomys intermedius*); (CDFW 2014). Clark's nutcracker (*Nucifraga columbiana*), Douglas' squirrel (*Tamiasciurus douglasii*), and other seed-caching wildlife species are important seed dispersers and predators of subalpine tree species in the assessment area (Tomback 1982, Vander Wall 2008).

Natural Range of Variation Descriptions

Function

Fire—

The fire history of subalpine systems in the assessment area is not well documented. Most trees that are characteristic of subalpine forests are not well adapted to resist fire, leaving few fire-scarred trees to reconstruct an accurate record of past fire events. The lack of specific regional studies required integration of information from areas adjacent to, but outside of, the assessment area (e.g., central Cascades of Oregon and southern Cascades of California), and occasionally from studies conducted in more distant regions (e.g., the Sierra Nevada, Washington, and Canada).

Fire type—Under presettlement conditions, fires in subalpine forests were relatively infrequent, of low to moderate intensity, and were usually limited in size (Agee 2007, Skinner 2003). Higher elevation subalpine forests, such as those dominated by whitebark pine, foxtail pine, and open stands of western white pine, typically experienced slow-moving surface fires owing to the presence of sparse surface and canopy fuels, natural terrain breaks, and harsh site conditions, which result in lower tree densities and biomass (Agee 2007, Keane et al. 2012). Fires in the higher elevation forests of the assessment area were small, often scarring only a single tree and probably of low or mixed intensity (Skinner 2003). Occasional crown fires did occur in these forest types, especially when foliar moistures were low and could have been aided by lichens draped within the canopy (Agee 1993). Lichens can be exceptionally flammable and their presence, combined with a high concentration of dead fuels around clumps of trees in subalpine forests, can encourage clump-to-clump torching, while the intervening shrublands are left unburned (Agee 1993, Franklin and Dyrness 1988).

Under presettlement conditions, fires in subalpine forests were relatively infrequent, of low to moderate intensity, and were usually limited in size.

In more mesic subalpine forests, such as those dominated by lodgepole pine or mountain hemlock, the fire regime was more likely characterized by a mixture of semifrequent surface fires and occasional crown fires, resulting in multiple fire types that were dependent on localized fire weather and fuel loading conditions (Agee 1993). The relatively higher frequency of crown fires in these forests was due to the presence of heavy and compact surface fuels, higher tree densities, and greater rates of litter and woody fuel deposition (van Wagtendonk et al. 2018). Sustained crown fires in these forests often only occurred under extreme dry and windy conditions (Agee 1990). In one study of subalpine fire regimes in the Mount Jefferson Wilderness in central Oregon, Simon (1991) found that variation in fire activity (between 1924 and 1991) was correlated with variation in annual precipitation. In the Lake Tahoe Basin, modeled fire behavior in presettlement subalpine stands dominated by mountain hemlock generally produced surface fires, with the potential for torching restricted to stands with high fuel loading under the most extreme weather conditions (98th percentile) (Maxwell et al. 2014).

There are few historical observations of fire in subalpine forests in the assessment area. During his surveys of the region in the late 1800s, William Brewer (1930: 481) made numerous references to fire:

> The whole of this wide landscape was bathed in smoky vapor, and the mountains faded in it at no great distance. It would be difficult to say where the smoky earth ceased and the smoky sky began.

In high-elevation lodgepole pine forests of Yosemite National Park, John Muir (1894: 225) remarked:

> During the calm, dry season of Indian summer the fire creeps quietly along the ground, feeding on the dry needles and burs; then arriving at the foot of a tree, the resiny bark is ignited, and the heated air ascends in a powerful current, increasing in velocity, and dragging the flames swiftly upward; then the leaves catch fire, and an immense column of flame beautifully spired on the edges, and tinted a rose-purple hue, rushes aloft thirty or forty feet above the top of the tree, forming a grand spectacle, especially on a dark night. It lasts, however, only a few seconds, vanishing with magical rapidity, to be succeeded by others along the fireline at irregular intervals for weeks at a time—tree after tree flashing and darkening, leaving the trunks and branches hardly scarred.

These observations support current studies in contemporary reference sites that these forests were characterized by surface fires interspersed with occasional wind-driven crown fires (e.g., Keifer 1991). Interestingly, fire type roughly parallels fire tolerance of subalpine tree species (Agee 1993). Most trees that occur in the subalpine forest type are not well adapted to survive fire. With the exception of lodgepole pine (evader) and whitebark pine (resister), most subalpine trees are fire avoiders (Agee 1993). For example, mountain hemlock, a common component of subalpine forests in the assessment area, is not well adapted to fire owing to its thin bark, low-hanging branches, highly flammable foliage, and tendency to grow in dense clusters (Sawyer et al. 2009). In observations from the early 1900s, Leiberg noted that when fires did occur in mountain hemlock stands, "most of the trees will be found to have dead and decaying streaks through the trunk" (Leiberg 1900).

Lightning is a common occurrence in high-elevation subalpine forests, providing ample opportunities for ignition (Atzet and Wheeler 1982); estimates from the Klamath Mountains suggest that lightning was common enough to produce

Table 4.2—Average historical fire return interval (FRI) estimates for subalpine forests in the assessment area[a]

Subalpine forest type (aggregation)[b]	Mean FRI[c]	Median FRI[c]	Minimum FRI[c]	Maximum FRI[c]	Number of studies	Subregions included
	Years					
Lodgepole pine	67	—	—	—	1	Southern Cascades (CA)
Mountain hemlock[d]	80	59	21	207	6	Cascades (CA, OR, WA), Klamath Mountains
Pacific silver fir[e]	138	—	—	—	1	Central Cascades (OR)
Subalpine fir[e]	113	—	—	—	1	Central Cascades (OR)
Western white pine[f]	50	42	15	370	1	Statewide (CA)
Whitebark pine	80	—	20	221	1	Cascades (OR and WA)

CA = California, OR = Oregon, WA = Washington. — = no estimate available.

[a] Individual FRI estimates and sources are presented in table A1.4 of the appendix.

[b] Averages for forest type aggregations were calculated using composite and single-tree samples.

[c] Values reported represent grand means (i.e., the mean of the means presented in the studies assessed).

[d] Includes forest types with Shasta red-fir, and whitebark pine.

[e] Value derived from analysis of stand ages rather than direct evidence of fire

[f] Values taken directly from summary provided by Van de Water and Safford (2011).

median fire return intervals (FRIs) of 10 to 15 years (Agee 2007). However, the cold, moist environment combined with a lack of dry litter, live understory, and discontinuous fuel greatly limits fire spread (Atzet and Wheeler 1982, Sawyer et al. 2009, Sugihara et al. 2018). The long FRI in high-elevation forests generally resulted in sufficient fuel buildup, making fire occurrence dependent mostly on whether climate or weather variables were suitable for ignition and spread. Taylor et al. (2008) examined the relationship between climate variability and fire extent in upper montane forests of the southern Cascades for the period that extended from 1700 through 1900. They determined that widespread burning was strongly associated with dry and warm conditions in the year of the fire, but that the strength of this relationship also varied over time in response to climatic variation over longer multidecadal time scales (Taylor et al. 2008).

Fire frequency—Historical FRI estimates for subalpine forests in the assessment are highly variable and largely dependent on forest type (tables 4.2 and A1.4). Although this variability suggests that a simple determination of mean or median FRI may not adequately (or accurately) describe the ecological process of fire in these high-elevation systems, some general trends are apparent. Mean and median composite FRI values were generally longest in forests dominated by Pacific silver fir (mean: 138 years) and subalpine fir (mean: 113 years), moderately long in whitebark pine (mean: 80 years) and mountain hemlock forests (mean: 80 years), and shortest in lodgepole pine (mean: 67 years) and western white pine forest types (mean: 50 years). Engelmann spruce has been noted to have fire-free intervals averaging more than 150 years (Sawyer et al. 2009). Limited fire scar and contemporary fire history data from foxtail pine stands in the southern Sierra Nevada suggest FRI estimates between 130 and 260 years (Keifer 1991, Rourke 1988). Some mean FRI estimates (e.g., mountain hemlock) differ by an order of magnitude (table A1.4), suggesting high levels of variability not only between forest types that occupy the subalpine forest zone, but also within specific subalpine forest associations. For example, a study of whitebark pine forests in the Cascade Range of Oregon and Washington documented FRIs that ranged from less than 10 to more than 170 years, with plots represented in every 10-year FRI range (Murray 2008).

The longer FRI in higher elevation forests is likely due to slower fuel accumulations and weather conditions that are less conducive to carrying fire.

Subalpine forests historically experienced fire at longer intervals than lower elevation forest types (Agee 1993, Habeck 1985). This longer FRI in higher elevation forests is likely due to slower fuel accumulations and weather conditions that are less conducive to carrying fire (Skinner 2003). In subalpine forests of the Pacific Northwest, forests composed of Engelmann spruce, mountain hemlock, subalpine fir, and whitebark pine show lower levels of mid-summer dryness than short fire interval vegetation types; these subalpine forests burn rarely, and when they do burn, fires often remain at low intensity, although they have the potential for high-intensity, stand-replacing fire under specific circumstances, such as during drought (Habeck 1985).

Accurate estimates of FRIs in subalpine forests are difficult to obtain because high mortality often leaves few fire-scarred trees (Agee 1993). As a result, there are few fire-scar-based estimates of return interval for subalpine forests that are specific to the assessment area. Only two studies, conducted in Crater Creek Basin in the Klamath Mountains of California, derived fire chronologies from fire-scarred tree specimens (Skinner 2003, Whitlock et al. 2004). These studies suggest that presettlement fires may not have been infrequent in some subalpine forest types in the assessment area; estimates of fire intervals in stands dominated by Shasta red fir, whitebark pine, foxtail pine, mountain hemlock, and other high-elevation species ranged from a mean single-tree (point) FRI estimate of 75.4 years (range: 12.5 to 256 years) to a mean composite FRI estimate of 39.5 years (range: 5 to 172 years). An unpublished study, which derived FRI from an analysis of stand ages rather than direct evidence of fire, documented markedly shorter mean FRIs in mountain hemlock plant associations (range: 11.5 to 36 years) in the Klamath Mountains on the Rogue River–Siskiyou National Forest in southwestern Oregon (White et al. 1997). Using a similar cohort methodology in the Klamath Mountains of southwestern Oregon, Atzet (1996) reported a much longer mean fire-free period of 115 years; however, they also noted that this estimate of periodicity was derived from observations rather than direct measurements. The complex topography of the Klamath Mountains, combined with the complex mosaic of subalpine vegetation, may result in more variable FRIs than those reported for subalpine ecosystems outside of the assessment area (Frost and Sweeney 2000).

A number of studies conducted outside of the assessment area also provide useful estimates of FRIs in subalpine systems. Siderius and Murray (2005) estimated a mean composite FRI of 73 years in whitebark pine stands in the southern Cascades of Oregon. In the Mount Jefferson Wilderness, located in the Cascade Range of central Oregon, Simon (1991) used age class chronologies to characterize historical FRIs and obtained estimates of 168 years for the mountain hemlock zone and 113 years for subalpine parklands (which included whitebark pine). Simon (1991) also noted that evidence of a historical fire, which occurred in 1724 in the mountain hemlock zone, was still apparent in the form of even-aged, 266-year-old stands, providing evidence that these areas burned very little (if at all) over that long period of time. Note that the use of stand age in this and other studies to derive FRI does not provide direct evidence of fire, but rather encompasses other disturbance types (e.g., wind, disease, avalanches, etc.) that may also contribute to stand initiation. Farther north in British Columbia, some subalpine mountain hemlock stands are thought to have been fire free for more than a millennium (Lertzman and Krebs 1991). Studies conducted closer to the assessment area, in the central and southern Cascades of Oregon area, also show very little evidence of recent fire in mountain hemlock stands, suggesting some stands have not burned for many centuries (Agee 1994, Foster 1998).

At a more local scale, longer FRIs are generally found in areas where fuel accumulations are low and landscape features, such as rock outcrops and meadows, are present to break up fuel continuity (Skinner and Chang 1996). Landscape position and context can also influence FRI patterns. For example, in his study of high-elevation forests in the southern Cascades, Foster (1998) found that fire frequency was negatively associated with elevation (i.e., less frequent at higher elevations) and positively associated with slope position (i.e., more frequent at higher slope positions). Simon (1991) noted that all of the major fires that occurred in subalpine forests in the Mount Jefferson Wilderness started within a zone defined by southerly aspects above 1524 m (5,000 ft) in elevation. In the southern Sierra Nevada, Caprio and Graber (2000) used reconstructions of annual area burned and found that mean and maximum FRI estimates for subalpine forests tended to be greater on relatively mesic north-facing slopes (mean FRI = 374; max FRI = 1,016 years) compared to xeric south-facing slopes (mean FRI = 187; max FRI = 508 years). In contrast, Taylor (2000) found that median FRI was similar across all slope aspects in red fir–mountain hemlock forests in Lassen Volcanic National Park. In this case, topography and landscape context (i.e., where the stand was in relation to other variables that influence fuel continuity and rate of fire import) likely played a strong role in determining the FRI.

Fire rotation is the time in years required for fire to burn an area that is equivalent to the study area's total extent and can be thought of as a landscape-level FRI (Agee 1993). In the Cascade Range of Oregon, fire rotation estimates for mountain hemlock stands varied from 450 to more than 1,000 years (Booth 1991, Dickman and Cook 1989, Simon 1991). Northeast of the assessment area, in the Mount Jefferson Wilderness in the Cascades of central Oregon, Simon (1991) reported a natural fire rotation of 472 years for subalpine fir stands. Studies of presettlement (pre-1905) fire regimes in the Caribou and Thousand Lakes wilderness areas on the Lassen National Forest estimated fire rotations that varied between 46 years for lodgepole pine forests and 147 years for red fir–mountain hemlock forests (Bekker and Taylor 2001). In foxtail pine stands of the southern Sierra Nevada, contemporary fire rotation estimates were much longer, ranging from 2,100 years (based on all fires) to 7,200 years (lightning fires only) (Rourke 1988). At a much broader scale, Agee (1990) conducted an analysis of vegetation and fire frequency data from forests in Oregon and adjacent states and estimated the fire cycle in subalpine forests to be 800 years, and the prehistorical average annual area burned was about 1336 ha, which is less than 1 percent of the area occupied by subalpine forests.

Current and future deviation from NRV—Analysis of data in the California Fire Return Interval Departure (FRID) database (Safford et al. 2015) indicates that about 79 percent of subalpine forests in California have experienced at least one fire over the past 108 years. Comparisons between the modern FRI (in this case based on the period 1908–2015) and estimates of the presettlement FRI (i.e., before 1850) (Safford and Van de Water 2014, Van de Water and Safford 2011) suggest that about 86 percent of subalpine forests in the assessment area exhibit low departure from their presettlement FRI. A small fraction (14 percent) of subalpine forest types may be burning slightly more frequently than they would have historically. Safford and Van de Water (2014) derived percent FRID (PFRID) to estimate the current percentage departure from the mean presettlement FRI. Currently, subalpine forests in the assessment area have a weighted average PFRID value of -8.7 percent and an average FRI of about 122 years. Within the California portion of the assessment area, this means that the average number of fires that have occurred in subalpine forests over the past 108 years is 0.9 (Safford et al. 2015). This analysis suggests that subalpine forests in the assessment area have likely missed only one or two fire cycles at most (i.e., mostly low FRID) since fire suppression became effective in the early 20th century.

In their analysis of contemporary (1985–2010) patterns of fire extent in Oregon and Washington, Reilly et al. (2017) calculated fire rotations of about 889 years and 176 years for mountain hemlock and subalpine forest, respectively. They concluded that fire rotation in the subalpine vegetation zone was within historical estimates (109 to 275 years); however, they also noted that estimates should be interpreted with caution owing to the relatively small extent of the subalpine zone and the disproportionate impact that large fire years can have on fire rotation.

Climate-driven increases in fire activity are already apparent in the Western United States (Westerling et al. 2006), and models specific to California and Oregon project increased fire activity to persist and possibly accelerate under most future climate scenarios (see "fire frequency" under red fir discussion on p. 24.). Compared to mid- or low-elevation forests, subalpine forests at the highest elevations may experience smaller increases in fire frequency owing to their greater available moisture (Westerling et al. 2011). However, projected increases in predicted fire probability suggest that fire frequency will likely increase, resulting in decreased FRI and fire rotation in subalpine forests of the assessment area.

Seasonality—Historically, most fires in subalpine forests occurred during the short dry season in late summer or fall (Agee 1993, Sawyer et al. 2009). Conditions for ignition and spread were most likely during this time period, which often coincides with higher densities of lightning strikes and low fuel and foliar moisture levels (Simon 1991). Skinner (2003) estimated that 95 percent of fires in montane and subalpine forests of the Klamath Mountains historically burned in late July through early October. In a study of fire regimes in the Mount Jefferson Wilderness in Oregon, Simon (1991) noted that contemporary fires typically occurred in subalpine forests during the summer months. Between 1918 and 1990, no lightning fires were recorded before mid-May or beyond the first week in September, and nearly two-thirds of documented fires occurred from mid-July to mid-August. Studies of high-elevation forests in the southern Cascades and Sierra Nevada document similar timing. For example, in lodgepole pine and red fir–western hemlock forests in the Thousand Lakes Wilderness, the position of fires on presettlement annual growth rings indicated that 99 to 100 percent of historical fires burned during the late summer and fall (Bekker and Taylor 2001, Taylor 2000). In the Lake Tahoe Basin, 92 percent of historical fires in red fir–western white pine forests burned during the late summer to fall, and 7 percent burned in the early to mid-summer (Taylor 2004). In high-elevation forests of Yosemite National Park, most wildfires and wildland fire use between 1974 and 2005 burned during the months of July, August, and September (van Wagtendonk and Lutz 2007).

Although more of an exception, some crown fires in the Cascade Range of Oregon and Washington have occurred in spring, while there was still snow on the ground; however, these fires each occurred after short periods (5 to 7 days) of exceptionally warm weather, which resulted in low foliar moistures (Agee 1993). Whitebark pine forests have also been shown to burn throughout the growing season, but most fires (especially large fires) historically occurred later in the growing season (Agee 1993).

Current and future deviation from NRV—The California Fire Perimeters Database (CDFFP 2015) contains information about fires (>1 ha) that occurred in subalpine forests of the assessment area over the past 105 years. Analysis of these data indicates that about 70 percent of the fires recorded during this time period occurred from late July to early October. Most of the remainder burned between June and early July (27 percent), while a few burned in winter (1 percent) or early spring (2 percent). Of those fires that occurred in early June, more than half occurred in a single year (2008) during a period of severe drought when one storm ignited numerous fires across the region. As seen in figure 4.4, this general pattern of fires occurring predominantly between late July and early October, with fewer fires occurring in early summer and late fall, has remained relatively consistent over the past 100 years.

These data, combined with the studies described above, indicate that fire season has not changed substantially in subalpine forests between historical and contemporary periods. However, warmer and drier summers, earlier snowmelt, and more rapid drying of fuels associated with climate change may result in longer fire seasons (i.e., increased frequency of fires in early summer and late fall) in the subalpine zone in the 21st century (Schwartz et al. 2015).

Extent—There is very limited information on the spatial extent of prehistoric fires in subalpine forests of the assessment area. Historically, fire size in these high-elevation systems would have been dependent on climate-related variables (e.g., fire weather), as well as the quantity and arrangement of vegetation and fuels (Agee 1993). Trees in the subalpine zone were often patchy, intergrading with herbaceous parkland, rock, or snow, which can hinder fire spread from patch to patch (Agee 1993). As a result, fires in the subalpine zone were likely small, commonly limited in extent to individual trees and clumps of individuals (Sawyer et al. 2009).

Fires in the subalpine zone were likely small, commonly limited in extent to individual trees and clumps of individuals.

In one of the few fire history studies of subalpine forests in the Klamath Mountains, Skinner (2003) found that 85 percent of fires detected occurred only on single trees, and none met the criteria for an extensive fire (defined as fires that scarred at least two trees, on two or more sites, in more than one basin). In

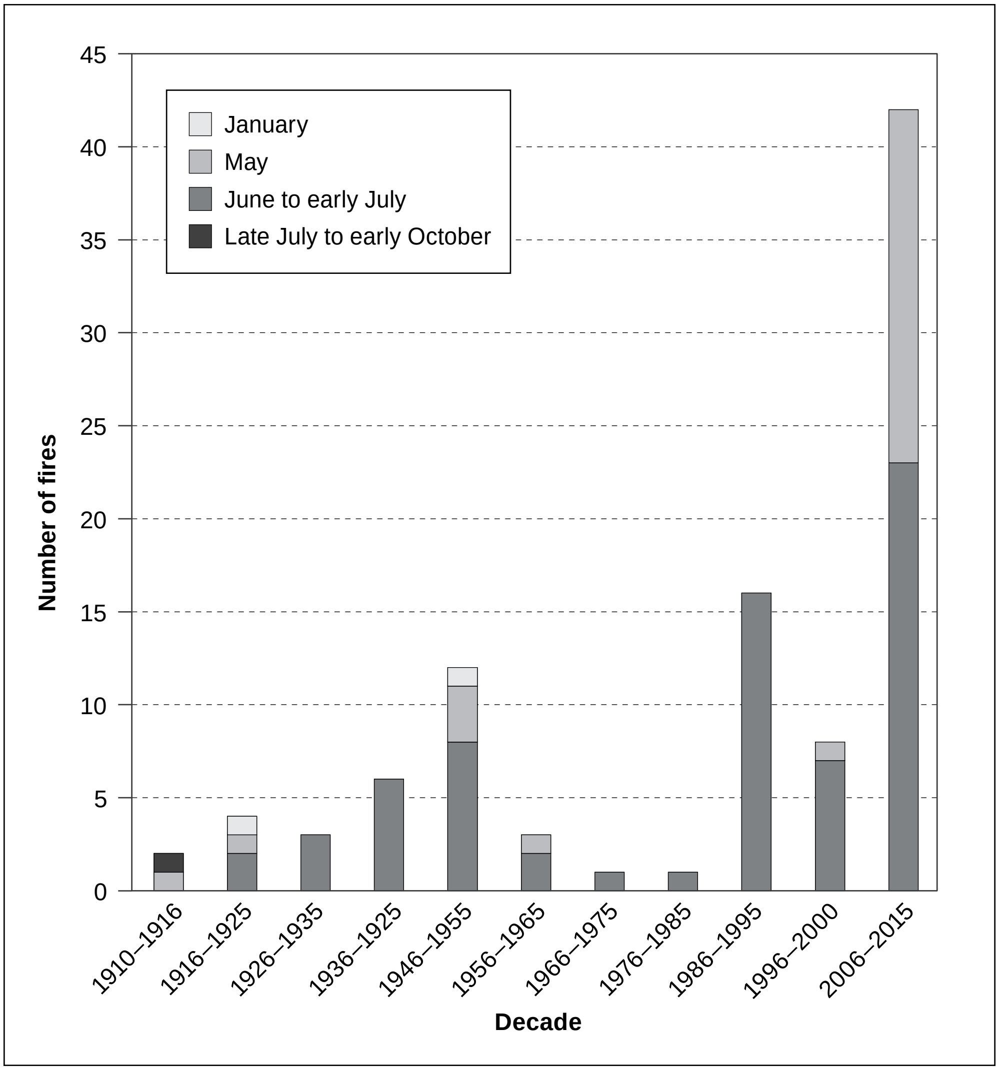


Figure 4.4—Seasonality of fires that have burned in subalpine forests in the California portion of the assessment area over the past 105 years. Fire data (>1 ha in size) were obtained from the California fire perimeter database (CDFFP 2015).

the southern Cascades of Oregon, Foster (1998) also found no indication of large fires in mountain hemlock forests (as indicated by evidence in contiguous plots). Presettlement (1729–1918) mean fire size in the Thousand Lakes Wilderness in the southern Cascades was estimated to be 140 ha (range: 124 to 155) in red fir–mountain hemlock forest (Bekker and Taylor 2001). In Lassen Volcanic National Park, historical mean fire size was 176 ha (median = 129 ha; range: 11 to 733 ha) in red fir–western white pine forest (Taylor 2000). Farther north in the subalpine fir forests of the Pasayten Wilderness in north-central Washington, Fahnestock (1976) determined that historical fires were generally small, with only about 15 percent exceeding 15 ha.

Although infrequent, fires in these systems were occasionally large, often coinciding with below-average regional precipitation (Taylor 2000). Fahnestock (1976) found evidence of two large historical fires in subalpine fir forests that each burned about 10 000 ha and accounted for more than 60 percent of the total area burned. In mountain hemlock forests in Oregon, Dickman and Cook (1989) documented historical fires greater than 3200 ha in size, although a majority were patches that were less than 500 ha. In subalpine forests in the Mount Jefferson Wilderness, Simon (1991) documented three fires, exceeding 2833 ha in size, before 1900 (years = 1724, 1814, 1871). In his early surveys of the Cascade Range, John Leiberg (1900) observed "extensive fires in the subalpine areas, which have destroyed large and dense growths of [mountain] hemlock and noble fir, and have induced soil conditions exceptionally favorable to reforestation by lodgepole pine." In one subalpine area, near the summit and high slopes of the southern Cascades, Leiberg (1900) also noted that fire had destroyed 75 percent of the timber.

Contemporary reference sites in the southern Sierra Nevada also provide evidence that the size of historical fires in subalpine forests may have been variable but were likely small. For example, in upper montane and subalpine forests of the Emigrant Basin Wilderness between 1951 and 1973, nearly 80 percent of lightning-caused fires were less than 0.1 ha and none were larger than 4 ha (Greenlee 1973 in Potter 1998). In Sequoia and Kings Canyon National Parks between 1968 and 1973, 80 percent of unsuppressed fires were smaller than 0.1 ha, and 87 percent were smaller than 4 ha (Potter 1998). In Yosemite National Park, 56 percent of unsuppressed fires in red fir and lodgepole pine forests between 1972 and 1993 were less than 0.1 ha, and 82 percent were smaller than 4 ha (van Wagtendonk 1994). In contrast to average fire size, the highest proportion of area burned (>70 percent) in red fir and lodgepole pine forests of Yosemite National Park tended to be from fires between 4 and 400 ha; an additional 28 percent of burned area is attributed to fires between approximately 400 and 2000 ha. (van Wagtendonk 1994).

Current and future deviation from NRV—Data from the California Fire Perimeters Database indicates that over the past 108 years, the average size of fires (>1 ha) in subalpine forests of the assessment area has been approximately 42 ha. Eighty-nine percent of these fires have been less than 100 ha (fig. 4.5). This is smaller than the average fire size (156 ha) estimated for presettlement high-elevation stands in the southern Cascades (Bekker and Taylor 2001, Taylor 2000); however, it should be noted that these estimates were from red fir–mountain hemlock and red fir–western white pine stands, which have shorter FRI and more contiguous fuels. As figure 4.6 suggests, the average size of fires has generally been

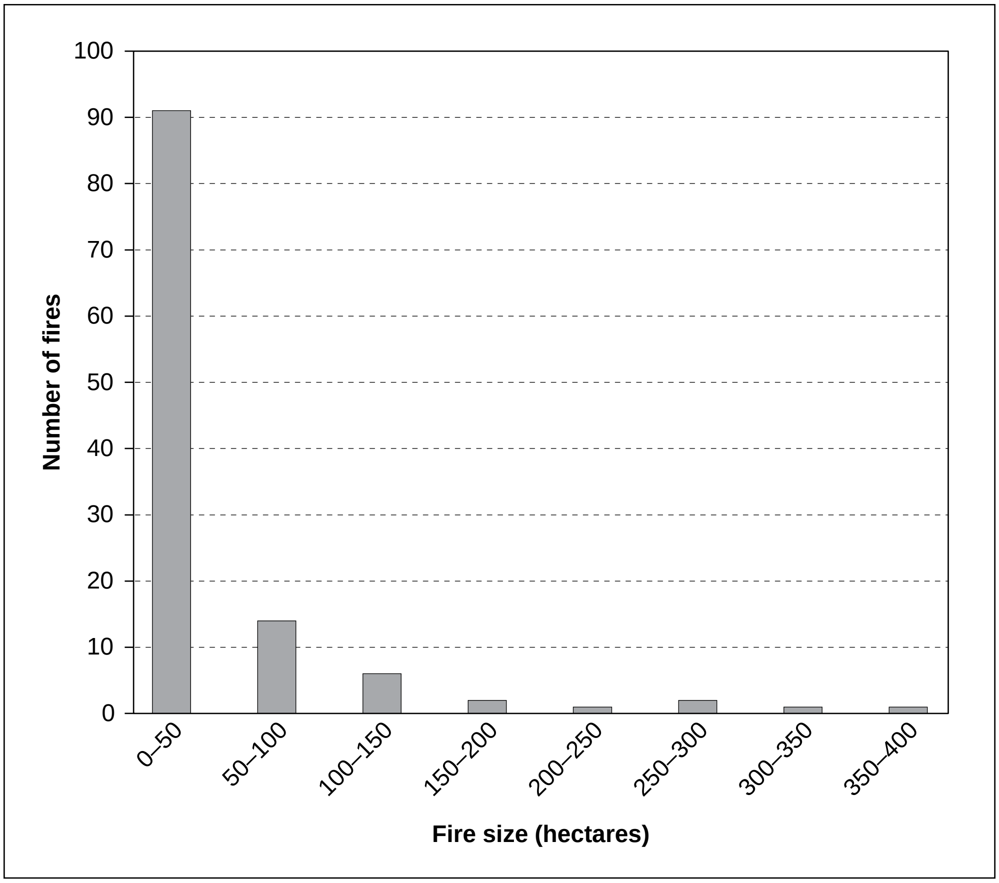


Figure 4.5—Distribution of fire size in subalpine forests of the assessment area. Data were obtained from the California fire perimeter database (CDFFP 2015) and include all fires greater than >1 ha in size that burned between 1910 and 2015.

smaller than estimates of presettlement fires; although presettlement estimates may fail to detect smaller sized fires and be biased toward higher average sizes. These data, together with the information summarized in the discussion above, suggests that fire size in subalpine forests is currently below or within the range of natural variation.

Severity—Fires in subalpine systems are erratic and often weather-driven, making it difficult to estimate fire behavior and subsequent fire effects (Agee 1993). In addition, variation in species assemblages and corresponding fire tolerance in subalpine forests can also complicate generalizations about fire severity. Although fires in the subalpine zone may burn at low to moderate intensity, the sensitivity of the vegetation to fire can result in high-severity fire effects (Taylor and Skinner 1998, 2003). Early forest survey reports conducted by the U.S. Geological Survey noted that high-severity, stand-replacing fires in subalpine fir forests were not uncommon (Gorman 1899 in Agee 1994, Leiberg 1900).

Although fires in the subalpine zone may burn at low to moderate intensity, the sensitivity of the vegetation to fire can result in high-severity fire effects.

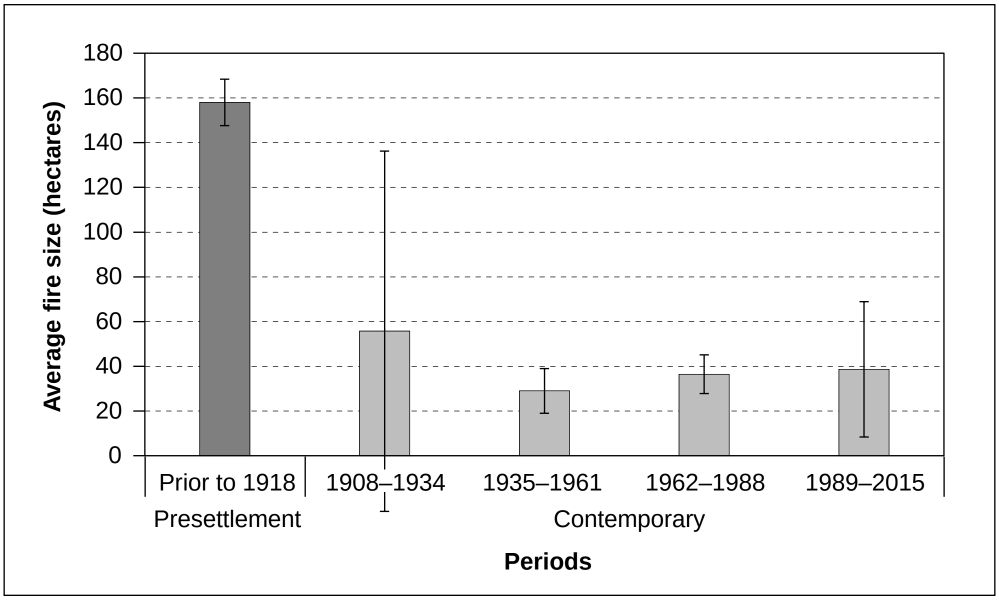


Figure 4.6—Comparison of mean fire size from estimates of presettlement fire regimes and wildfires that have burned within the assessment area since the institution of fire suppression in the early 20th century; the latter are divided into four equal time periods to show temporal trends. Only fires >1 ha are included. Presettlement estimates were taken from studies conducted in the Thousand Lakes Wilderness (Lassen National Forest) and Lassen Volcanic National Park (Bekker and Taylor 2001, Taylor 2000,). Contemporary fire sizes were estimated using the California fire perimeter database (CDFFP 2015). Error bars represent the standard error. Note that fire size for modern fires (since 1908) were calculated based on the area burned within the subalpine zone and does not reflect total fire size.

Subalpine forests in contemporary reference sites are often characterized as having low or mixed-severity fire regimes, the latter of which is generally characterized by multiple fire severity classes (Meyer and North 2019). Higher elevation subalpine forests (e.g., white pine forests) typically experience low-severity fire (often less than 25 percent tree mortality), whereas mesic lodgepole pine or mountain hemlock forests are generally characterized as mixed severity (Agee 1993, Arno 2000, Keane et al. 2012). In subalpine forests of the Sierra Nevada, the proportion of area burned at high severity (defined as more than 75 percent tree mortality) averaged 27 percent and 7 percent across studies in lodgepole pine and other subalpine forests, respectively (Meyer and North 2019).

In one study of subalpine mountain hemlock forests in Crater Lake National Park, Forrestel (2013) documented continuous establishment of mountain hemlock over a period of four centuries, which suggests that high- and even moderate-severity fires are rare in this forest type. Observations from the 2006 Bybee Fire in Crater Lake National Park also support the idea that fires in subalpine forests generally burn as low-severity surface fires, with small pockets of higher severity (Forrestel 2013).

Most of the time, a climate-limited situation for crown fire development exists in subalpine systems; however, in very unusual years, often separated by centuries, large-scale, high-severity crown fires can occur, especially in areas with heavy fuel buildup (Agee 1993, Sawyer et al. 2009). There have been contemporary examples of high-severity crown fires in the Olympic Mountains of the Pacific Northwest, including the 1978 Hoh Fire and 1981 Chimney Peak Fire (Agee 1993).

Current and future deviation from NRV—Averages for high-severity proportion and patch size are likely within NRV, but historical information is limited with respect to these variables. In their comparison of fire severity in presettlement and modern forest types, Mallek et al. (2013) estimated that modern rates of burning in Sierra Nevada subalpine forests for any severity class (i.e., low, moderate, high) was currently lower compared to the presettlement period (Mallek et al. 2013). Accordingly, current subalpine forests may be deficient in all fire severity classes at the bioregional scale. However, projections of future climate suggest that fire severity or intensity may increase in many parts of California during the mid-21st century, especially in subalpine forests (Lenihan et al. 2003, 2008).

Insects and pathogens—

Native insects and disease-causing organisms have been present in high-elevation forests of the assessment area for millennia, essentially for as long as their host trees (Hawksworth 1978, Mathiasen 2011). Several of these insect and disease species interact to affect the health of trees at higher elevations, especially during periods of stress associated with extended drought or following disturbance (Ferrell 1996). Unfortunately, very few quantitative data are available that describe the frequency and spatial extent of these species within the assessment area presettlement making a determination of the NRV challenging.

Although there is some evidence of large native insect outbreaks in presettlement subalpine forests (e.g., Brunelle et al. 2008), the historical structure and composition of subalpine forests generally ensured that these disturbances were brief and spatially confined (Hessburg et al. 1994). Insects and disease agents likely caused minor damage or localized mortality (i.e., at the scale of individual trees or small clumps of trees) and were generally limited by the cold temperatures, low humidity, rocky environments, and wide spacing of trees in high-elevation habitats (Ferrell 1996, Millar and Rundel 2016).

Insects and disease agents likely caused minor damage or localized mortality (i.e., at the scale of individual trees or small clumps of trees) and were generally limited by the cold temperatures, low humidity, rocky environments, and wide spacing of trees in high-elevation habitats.

Subalpine forests in the assessment area likely support a similar assemblage of insects and pathogens today as they did 100 years ago (Hessburg et al. 1994). One notable exception to this is white pine blister rust, which is caused by the nonnative fungus *Cronartium ribicola* and can infect whitebark and other five-needle pines

(Kliejunas and Adams 2003). The rust was inadvertently introduced into western North America around 1910; by 1929, detections had been recorded within the assessment area on the Six Rivers and Klamath National Forests (Kliejunas and Adams 2003).

The current level and extent of white pine blister rust infection within the assessment area varies widely among regions and between species. For example, in a study of whitebark pine forests in Idaho, Montana, and Oregon, Larson (2011) found the lowest infestation rates just north of the NRV assessment area in the Cascade Range (mean = 11 percent of trees per plot) and on Paulina Peak (mean = 5 percent); mean infection values in other portions of the study area ranged from 38 to 66 percent of surveyed trees. Eskelson and Monleon (2018) examined forest inventory data in northeastern California, Oregon, and Washington and found no evidence of white pine blister rust in any of the California plots measured. In contrast, they detected white pine blister rust in 17 percent of inventory plots in Oregon and 49 percent of inventory plots in Washington (Eskelson and Monleon 2018). Within the California portion of the assessment area, Maloney et al. (2011) conducted an assessment to determine the incidence and distribution of white pine blister rust in high-elevation forests and found that rust incidence was higher in the north Coast region (42 percent of trees surveyed), moderate and variable in the Klamath Mountains (12 percent), and low in the southern Cascades (1 percent). Western white pine had the highest mean incidence (42 percent in the Klamath Mountains), followed by whitebark pine (24 percent in the northern Sierra Nevada), while the geographically restricted foxtail pine had a mean incidence of 12 percent in the Klamath Mountains (Maloney 2011). In a study of eight whitebark pine stands in the Lake Tahoe Basin, Maloney et al. (2012) found that the mean incidence of white pine blister rust among populations was 35 percent (range: 1 to 65 percent). Goheen et al. (2002) surveyed whitebark pine stands along the Pacific Crest Trail on the Umpqua National Forest and reported wide variation in white pine blister rust occurrence along transects (range 0 to 100 percent), with an estimated 46 percent of all live trees infected (Goheen et al. 2002). In their monitoring of whitebark pine, Jules et al. (2017) also reported wide variability in blister rust infection within plots (range 0 to 100 percent), with average infection rates among plots of 51 percent in Crater Lake National Park and 54 percent in Lassen Volcanic National Park.

Since its introduction into high-elevation forests in the early 1900s, white pine blister rust and its interaction with other stressors (i.e., mountain pine beetle [*Dendroctonus ponderosae*], climate warming, altered fire regimes, and drought) has been identified as a major factor in the decline of whitebark pine across the West.

Although not considered in detail in this assessment, the presence of this and other exotic pathogens is well outside of the NRV for subalpine forests.

One of the most prevalent native insects in subalpine forests is the mountain pine beetle, which can affect lodgepole pine, whitebark pine, western white pine, and foxtail pine (Ferrell 1996, Millar and Rundel 2016). Paleoecological reconstructions and historical surveys from the Rocky Mountains provide some evidence that significant outbreaks of mountain pine beetle occurred at irregular intervals in the past. For example, in their study of whitebark pine forests in the northern Rocky Mountains, Brunelle et al. (2008) found evidence of outbreaks during the early Holocene (~8,000 YBP) in periods associated with rapid changes in climate. Historical records from this region also provide evidence of more recent historical (early 1900s) outbreaks of mountain pine beetle, generally in response to mild winters, warmer and drier summers, and highly susceptible stand conditions (Gibson et al. 2008). In contrast, Ferrell (1996) reviewed insect damage reports for the Sierra Nevada from 1917 to 1993 and found that mountain pine beetle outbreaks in subalpine forests were relatively infrequent and usually limited to small clumps of trees; although larger outbreaks in dense stands were not uncommon. Note that it is not always apparent whether historical insect damage surveys were effective at detecting mountain pine beetle outbreaks within high-elevation subalpine forests, owing to the relatively low economic importance and inaccessibility of these forest types (Gibson et al. 2008). Consequently, historical information is inconclusive with respect to the frequency and extent of mountain pine beetle outbreaks in subalpine forests in the assessment area.

There are several other insects and pathogens that affect high-elevation forests of the assessment area that are not discussed in detail here (see “insects and pathogens” under red fir section, p. 34). One important disease agent in mountain hemlock forests in the southern Cascades of Oregon is laminated root rot (*Phellinus sulphurascens*). This fungus, which can cause rings of mortality several hectares in size, historically played an important role in successional processes and subsequent fire behavior and intensity (Hessburg et al. 1994, McCauley and Cook 1980). Laminated root rot can persist across a range of topographic, climatic, and soil conditions and appears to be well adapted to environmental conditions that support an abundance of highly susceptible hosts (e.g., mountain hemlock) (Hadfield et al. 1986, Hessburg et al. 1994). Surveys of mountain hemlock forests in the Mount Thielsen Wilderness on the Umpqua National Forest documented levels of canopy reduction from laminated root rot that ranged from low (32 percent of plots) to moderate (51 percent of plots) to high (17 percent of plots)(Goheen et al. 2002). In their assessment of the Klamath Mountains, Hessburg et al. (2000) determined that the percentage of the landscape that is currently considered highly vulnerable to laminated root rot is not significantly different than recent historical time periods; although current vulnerability was slightly higher in the southern Cascades region (outside the assessment area) compared to historical estimates.

Current and future deviation from NRV—Despite the lack of quantitative historical evidence, some inferences can be made about potential changes in the scale and intensity of insect and disease infestations over time. Hessburg et al. (1999b, 2000) assessed changes in the vulnerability of the landscape to insect and pathogen disturbance in eastern Oregon and Washington, including a small portion of the assessment area. Using recent historical (1932–1966) and current (1981–1993) aerial photographs, they estimated landscape vulnerability to 17 disturbance agents (many of which affect subalpine forests) in the Klamath Mountain region. They determined that all but two showed no significant difference between historical and contemporary time periods. The exceptions to this were fir engraver beetle and Schweinitzii root and butt rot (*Phaeolus schweinitzii*), which showed a significant increase and decrease, respectively, between recent historical and current time periods (Hessburg et al. 2000). Fir engraver would have historically been a minor mortality agent in high-elevation forests, primarily affecting subalpine fir, grand fir (*Abies grandis*), or white fir; therefore, the increase in fir engraver mortality in contemporary subalpine forests is likely associated with periods of drought and the increased density of the host trees (Hessburg et al. 1994).

Recent modeling of mountain pine beetle activity throughout the Western United States (including the assessment area) provides additional insight into the NRV for beetle outbreaks. Hicke et al. (2006) modeled historical (1895–1960), current (1961–2005), and future (2006–2100) mountain pine beetle outbreaks in forests of the Western United States using climatic factors that are closely associated with adaptive seasonality (i.e., the synchronous emergence of adults from host trees at the appropriate time of year). They found that the adaptive seasonality of mountain pine beetle was similar in historical and current time periods for many parts of the Western United States, including the higher elevations of the Cascades. However, future projections of adaptive seasonality at high elevations (>3000 m) show that the total area susceptible to bark beetle attack would initially increase (during 2005–2050), then steadily decline but remain above current levels between 2060 and 2100 (Hicke et al. 2006). Bentz et al. (2010) found similar results comparing current and projected future adaptive seasonality with a similar population model. Weed et al. (2015) found evidence of geographic variation in the response of mountain pine beetle to warming minimum temperatures across the Western United States. In their study, beetle outbreaks between 1960 and 2011 could not be attributed to warming winters in areas like the Klamath Mountains and Cascade Range. These areas historically experienced winters that were not cold enough to drop below the predicted lower lethal temperature for mountain pine beetle; therefore, the authors suggest that increases in beetle survival in response to recent warming either did not occur or was relatively small (Weed et al. 2015).

Several recent studies and reports outside of the assessment area lend support to model projections for mountain pine beetle activity. Since 2006, mountain pine beetle activity in subalpine forests dominated by western white pine and lodgepole pine has increased substantially above background levels, especially on the Modoc National Forest and on the north slope of Mount Shasta (California Forest Pest Council 2011). Monitoring in high-elevation whitebark pine forests in Crater Lake National Park suggests that mountain pine beetle activity has increased markedly since about 2010, possibly as a result of extended dry seasons and warmer temperatures (Jules et al. 2016, 2017; Preisler et al. 2012; Smith et al. 2011). In their study of whitebark pine in Yellowstone National Park, Logan et al. (2010) concluded that mountain pine beetle mortality levels were well beyond the more episodic events of the past. Meigs et al. (2015) analyzed mountain pine beetle mortality between 1970 and 2012 across the Pacific Northwest; they found a significant increase in beetle-related mortality in lodgepole pine forests of the eastern Cascades, but not in the Coast Range or Klamath Mountains of Oregon. In the Sierra Nevada, mountain pine beetle outbreaks, combined with drought and other stressors, have caused significant mortality of whitebark pine (Millar et al. 2012). These outbreaks have led to changes in the structure, regeneration, and dominance of whitebark pine stands (Meyer et al. 2016). In many of these studies, increased mountain pine beetle activity has been clearly linked to increases in temperature and climatic water deficit, decreases in precipitation, and greater stand densities or tree diameters (Meyer et al. 2016; Millar et al. 2007, 2012; Preisler et al. 2012). This suggests that future outbreaks in high-elevation subalpine stands are likely in the near future.

In 2012, the U.S. Forest Service (USFS) conducted a nationwide strategic assessment of mortality risk owing to insects and diseases (Krist et al. 2014). They used a suite of variables, including host tree parameters, climatic data, physiographic variables, soil information, and other data to create individual pest and pathogen models. They used these models to project basal area mortality risk for individual tree species over a 15-year time period (2013 to 2027). Within the assessment area, their models projected an average basal area loss of 44 percent for whitebark pine and 12 percent for western white pine, primarily owing to mountain pine beetle and white pine blister rust. Mountain hemlock was projected to decline by an estimated 10 percent as a result of root diseases (Krist et al. 2014).

Over the 20th and 21st centuries, tree mortality related to insect and disease outbreaks has greatly increased in subalpine forests across western North America (Millar and Rundel 2016). In times of drought, when soil moisture stress is high, insect outbreaks can reach record levels (Millar and Rundel 2016). In other cases, the duration, extent, and severity of an outbreak has been linked to increased uniformity and continuity of the host tree species (Gibson et al. 2008).

Collectively, these studies and reports suggest that outbreaks associated with native insects and pathogens are within NRV for subalpine forests in the assessment area, at least prior to the more recent (2006–2012) outbreak events in high-elevation white pine stands in the southern Sierra Nevada and northeastern California. However, changes in species composition, stand density, or canopy structure in subalpine forests (see "Structure" in chapter 4 on p. 97) can have strong effects on insect and pathogen responses and associated ecosystem impacts (Hessburg et al. 1994). Nonnative pathogens such as white pine blister rust, which were not present before European-American settlement and present a significant threat to high-elevation pines, will continue to affect subalpine forests in the assessment area. Near-future (to 2050) projections for high-elevation white pine forests suggest that an increase in the frequency and extent of mountain pine beetle outbreaks may greatly exceed NRV by the early to mid-21st century. This projected increase could have substantial cascading impacts on subalpine forest ecosystems of the assessment area, similar to those observed recently in the central and northern Rocky Mountains (Edburg et al. 2012).

Near-future (to 2050) projections for high-elevation white pine forests suggest that an increase in the frequency and extent of mountain pine beetle outbreaks may greatly exceed NRV by the early to mid-21st century.

Wind and avalanche—

Although high windspeeds are not uncommon in subalpine habitats, windthrow is relatively rare owing to the ability of many subalpine tree species to accommodate or resist wind (Millar and Rundel 2016). Wind can play a more indirect role in influencing stand structure, however, with areas of deeper wind-influenced snowdrifts often defining areas that retain soil moisture later in the growing season (Millar and Rundel 2016).

Avalanches exert important local controls on subalpine forests, by influencing spatial patterns, species diversity, as well as tree size, form, and persistence. For example, in whitebark pine-mountain hemlock forests within the Antelope Creek Lakes Research Natural Area on the Klamath National Forest, reproduction of mountain hemlock in the understory mainly occurs in areas of disturbance such as avalanche sites or heavy snow accumulation (Cheng 2004). In a study of high-elevation meadows in the Mount Jefferson Wilderness in the southern Cascades of Oregon, Zald et al. (2012) found rapid colonization by mountain hemlock on debris flows, which they attributed to lower snow depth (i.e., as a result of lower overstory canopy and increased wind distribution) and longer snow-free periods.

Current and future deviation from NRV—Historical avalanche and wind events are difficult to compare to current rates because they are highly infrequent and unpredictable. Nevertheless, current rates of wind and avalanche in subalpine forests of the assessment area are broadly considered within NRV. Interestingly, projected declines in snowpack in the 21st century could result in reduced probability of avalanche and increased landslide activity in the coming decades (Huggel et al. 2012). However, reductions in snowpack could be offset in the near future by potential increases in precipitation variability and the frequency of heavy, unpredictable snowfall and rain-on-snow events, which can elevate avalanche probability, especially at higher elevations (Bebi et al. 2009).

Structure

The structure of subalpine forests in the assessment area prior to settlement was likely a mosaic of openings, dense tree clusters, and single trees, which were associated with variation in disturbance intensity and tree species ecology (Agee 1993). Subalpine forests in the assessment area can generally be divided into two subzones with different structural attributes: (1) the forest zone, which is characterized by relatively continuous forest cover (i.e., greater than 75 percent); and (2) the parkland zone, which occurs at higher elevations and is characterized by more widely spaced groups of trees (Gedalof and Smith 2001). John Muir (1918: 32) described this variation in subalpine forest zones during his travels around Mount Shasta in 1918:

> The next higher is the Fir Zone, made up almost exclusively of two species of silver fir. It is from two to three miles wide, has an average elevation above the sea of some six thousand feet on its lower edge and eight thousand on its upper, and is the most regular and best defined of the three.
>
> The Alpine Zone has a rugged, straggling growth of storm-beaten dwarf [whitebark] pines, which forms the upper edge of the timberline. This species reaches an elevation of about nine thousand feet, but at this height the tops of the trees rise only a few feet into the thin frosty air, and are closely pressed and shorn by wind and snow; yet they hold on bravely and put forth an abundance of beautiful purple flowers and produce cones and seeds. Down towards the edge of the fir belt they stand erect, forming small, well-formed trunks, and are associated with the taller two-leafed and mountain pines and the beautiful Williamson spruce [mountain hemlock].
>
> The woods differ but little from those that clothe the mountains to the southward, the trees being slightly closer together and generally not quite so large, marking the incipient change from the open sunny forests of the Sierra to the dense damp forests of the northern coast, where a squirrel

> may travel in the branches of the thick-set trees hundreds of miles without touching the ground. Around the upper belt of the forest you may see gaps where the ground has been cleared by avalanches of snow, thousands of tons in weight, which, descending with grand rush and roar, brush the trees from their paths like so many fragile shrubs or grasses.

There is very little quantitative data that describe the structure of subalpine forests in the assessment area presettlement. Therefore, this NRV assessment relies heavily on studies conducted in undisturbed (i.e., unlogged) contemporary reference landscapes, both within and outside of the assessment area, as well as historical accounts of early surveyors.

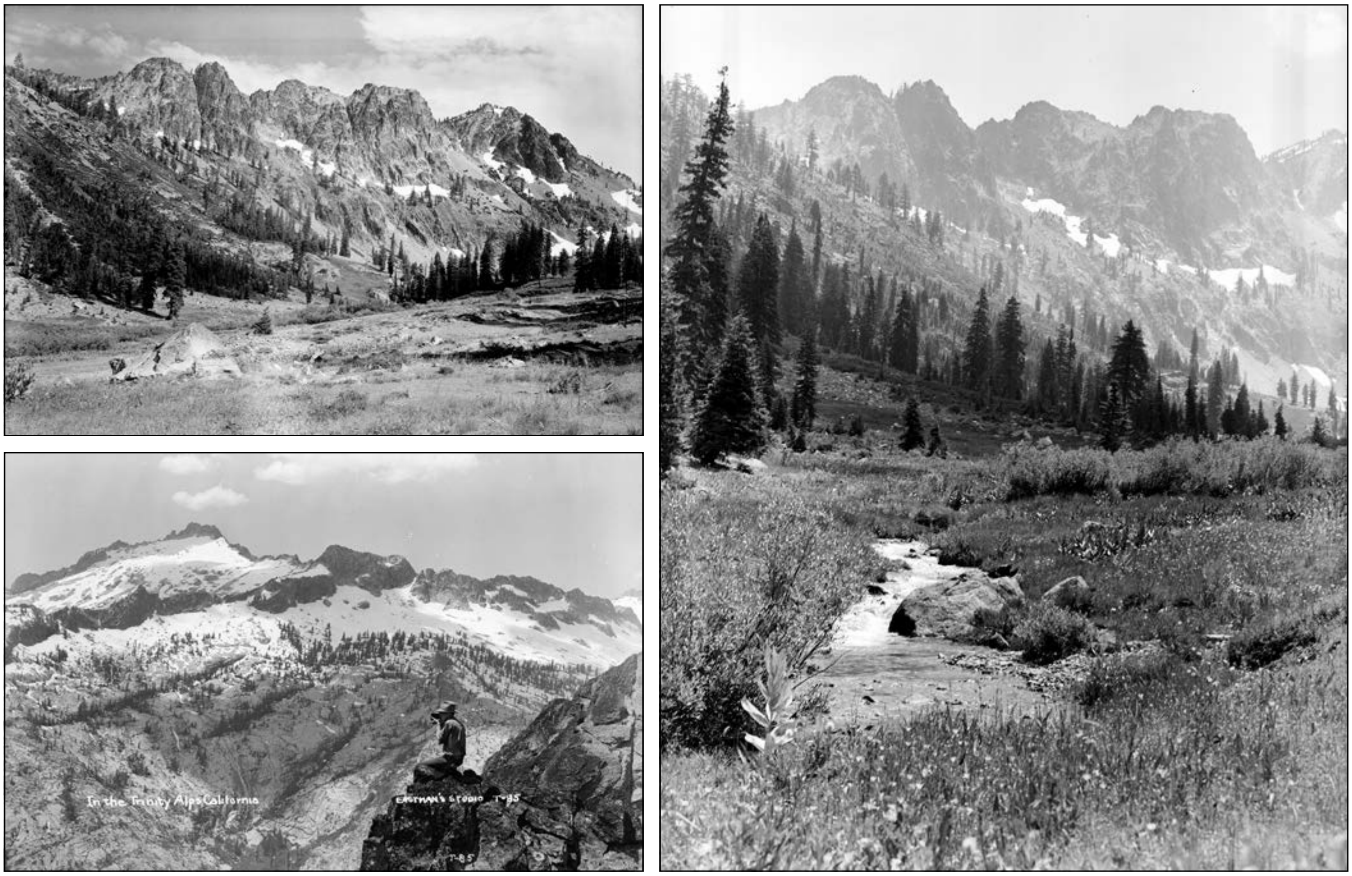
Tree density—

In general, tree density is greatest at the lower elevational limits of the subalpine zone, in forest types dominated by mountain hemlock and noble fir, and lowest in higher elevation woodlands, where harsh growing conditions result in widely spaced, stunted trees.

Tree density is highly variable, both within and among subalpine forest stands, owing to the complex topography and diverse species composition found within these high-elevation habitats (table A1.5) (Fites-Kaufman et al. 2007). In general, tree density is highest at the lower elevational limits of the subalpine zone, in forest types dominated by mountain hemlock and noble fir, and lowest in higher elevation woodlands, where harsh growing conditions result in widely spaced, stunted trees (Cheng 2004, Verner and Purcell 1988). This variability was noted by early surveyors like John Leiberg (1900: 262) who made the following observation of subalpine habitats in the southern Cascades (near what is now the Rogue River–Siskiyou National Forest):

> The aspect and density of the alpine-hemlock type of forest vary exceedingly according to age and altitude, far more so than either the red-fir or the yellow-pine types. At high elevations a mature forest of the type consists of low, stunted individuals, set well apart. At middle altitudes the type is usually open and park-like when well advanced in age, the alpine-hemlock component being most frequently collected in groups with individuals of white pine and noble fir scattered throughout. At its lower altitudinal limits well-preserved examples of the type often present magnificent examples of forest growth but little inferior in timber volume to the best stands of the red-fir type. In such cases the type is almost wholly composed of alpine hemlock [mountain hemlock] and noble fir; the trees stand 20 to 30 feet apart with long columnar trunks, and run from 100 to 200 mature trees to the acre. Stands of this kind occur on the northern slopes of the Umpqua Rogue River divide, and in the Cascades at the head of the Middle Fork of Rogue River. They are never of any very large extent, most of them having been ruined by fire long since.

Photographs taken by Jervie Henry Eastman in the 1930s in the Trinity Alps (fig. 4.7) and Marble Mountains (fig. 4.8) also illustrate the variability in subalpine



Courtesy Special Collections, University of California, Davis Library. From Calisphere (https://calisphere.org).

Figure 4.7—Photographs from the Eastman (J.H.) Collection showing high-elevation habitat in the Trinity Alps in 1938.

Courtesy Special Collections, University of California, Davis Library. Calisphere (https://calisphere.org).

Figure 4.8—Photograph from the Eastman (J.H.) Collection showing high-elevation habitat in the Marble Mountain "Primitive Area" in 1929.

tree density within these elevational bands, with dense stands and clumps of trees adjacent to very open stands of subalpine trees.

The following observations of the southern Cascades by John Leiberg also suggest that forests at higher elevations were predominantly open (i.e., low-density) stands with widely spaced groups of trees:

> Much of the region under examination is composed of high subalpine regions which naturally carry light stands of timber…..it is in the alpine-hemlock [mountain hemlock] type of forest that the stands of timber become thin and light [Leiberg 1990: 269].
>
> The forest along the crest is thin. Much of it exists as small groups separated by bare pumice flats or as scattered trees, in which case such areas might perhaps be more properly classed as wooded than as forested. Elevations above 7,500 feet along the crest are rocky and nearly devoid of arborescent growth, stunted individuals of the white-bark pine being almost the only denizens of such tracts [Leiberg 1990: 316].

Journal entries from William H. Brewer that describe his surveys along the west slope of Mount Shasta in the 1860s (Brewer 1930: 473) also provide evidence that forests at higher elevations were historically open: "We came on to Shasta Valley, over the pass on the east side of Shasta Peak. This pass is about six thousand feet high and very gradual. The views of the peak were the most sublime we have yet had. We were up to within two thousand feet of the lower edge of the snow, in the sparse timber and pure air of this height." In the passage below, Brewer (1930: 312) describes his camp near Mount Shasta, which was situated above a forest dominated by white fir, Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*), incense cedar (*Calocedrus decurrens*), yellow pine, and sugar pine (*Pinus lambertiana*), and below a timberline dominated by whitebark pine.

> Camp 99 was here, at the upper edge of the timber, streaks of snow coming down below this level. It is about 7,400 feet altitude. Here the trees are still numerous, although scarcely forming a "forest." They are of only one species, a grand fir, *picea nobilis*,[1] many of the trees over four feet in diameter—one near camp is six feet—yet they cease entirely a few hundred feet higher.

Brewer (1930) also made the following observations about high elevations in the Siskiyou Mountains and southern Oregon:

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[1] The "grand fir" described here is actually red fir (*Abies magnifica* var. *shastensis*).

> Just north of this ranch are several high peaks of the Siskiyou Mountains…. the whole region a mountainous one—the peaks five thousand to seven thousand feet high, some indeed much higher—and all furrowed into deep canyons and sharp ridges, many of the former over two thousand feet deep. The hills are covered with scattered timber, not dense enough to be called forests, or in places with shrubby chaparral [Brewer 1930: 481].

> We followed along on the crest of ridges for several miles, with deep canyons on every side, the soil barren but supporting a growth of low bushes, scarcely dense enough to be called chaparral, with here and there a small pine or cedar. One species of pine bears cones when but two feet high, and little trees ten feet high were fruitful with them [Brewer 1930: 484].

Published estimates of tree density in historical subalpine forests of the assessment area are not available. However, there are some studies conducted outside of the assessment area that suggest that average tree density may have been lower in presettlement subalpine forests. In one study of mountain hemlock forests in Lassen Volcanic National Park, Taylor (1995) documented an increase in tree density (>5 cm diameter at breast height [d.b.h.]) over a period of 150 years in response to warm mesic conditions that were common at the close of the Little Ice Age (1850–1880). A similar pattern of increasing tree density between presettlement and contemporary time periods was found in high-elevation secondary-growth lodgepole pine forests of the Lake Tahoe Basin by Taylor et al. (Taylor 2004, Taylor et al. 2014). In the central Sierra Nevada, Dolanc et al. (2013a) compared historical inventory data (Wieslander Vegetation Type Mapping, 1929–1934) with modern survey data (2007–2009) and found increases in tree density in subalpine forest types. They found a net increase in tree density of 30 percent, including a 63 percent (range: 44 to 91) increase in small-tree (<30.4 cm d.b.h.) density for whitebark pine, lodgepole pine, mountain hemlock, and red fir (Dolanc et al. 2013a).

In subalpine forests that included foxtail pine and lodgepole pine stands, Vankat and Major (1978) documented an apparent increase in tree density and cover in their comparison of historical photographs (taken prior to 1920) with more modern photographs (1978) taken in Sequoia National Park. This pattern of increasing tree densities and cover was also documented by Gruell (2001) in his comparison of historical (1867–1900) and contemporary (1990s) photographs taken in subalpine forests throughout the Sierra Nevada. Analysis of repeat photography by Skovlin et al. (2001) revealed a similar trend in high-elevation habitats of northeastern Oregon; in many of these stands, the density of subalpine fir had increased, and individuals had encroached into areas that had little or no fir in 1925.

Current and future deviation from NRV—Quantitative data describing tree density presettlement is lacking for the assessment area, making it difficult to evaluate change in this attribute over the past century. Stand reconstruction studies, which generally provide our best estimates of presettlement stand structure, are also limited in their ability to accurately account for very small trees in the historical reference period (Taylor 2004). Comparisons between contemporary (unlogged) reference areas and contemporary (managed) stands can also offer insight into potential deviation from NRV; however, we were unable to find these comparative studies for subalpine forests in the assessment area.

Based on historical observations, as well as studies conducted in subalpine habitats outside of the assessment area, it is likely that subalpine forests in the assessment have experienced some increase in tree density. However, the magnitude of this increase is likely variable, both within the assessment area and within the subalpine forest zone, and dependent on elevation, soil type, topography, and species composition of the stand. Within the subalpine forest zone, tree density will generally be highest at the lower elevational limits, in forest types dominated by mountain hemlock and noble fir, and lowest in higher elevation woodlands, where harsh growing conditions result in widely spaced, stunted trees (Cheng 2004, Verner and Purcell 1988). Future climate projections, coupled with continued fire exclusion, could lead to increases in tree density in the future; however, note that wildfire and mortality trends (e.g., from native insects and nonnative pathogens) may also counteract this trend to some extent.

Tree density is generally greatest at the lower elevational limits and lowest in higher elevation woodlands, where harsh growing conditions result in widely spaced, stunted trees.

Tree size and size class distribution—

Published studies that describe historical tree size and size class distributions for subalpine forests in the assessment area are not available. Historical survey accounts occasionally noted the large size of the trees encountered in subalpine habitats (Brewer 1930, Leiberg 1900); however, it is difficult to determine how frequent these large trees were presettlement.

Although outside of the assessment area, a few studies conducted in subalpine forests of the southern Cascades and Sierra Nevada suggest that the size class distribution of subalpine forests has likely shifted to smaller size classes over time, with an overall decline in large-diameter trees. For example, Dolanc et al. (2013a) documented a 63 percent increase in small-tree density and a corresponding decrease of 20 percent in large-tree density for subalpine species such as western white pine, lodgepole pine, and red fir over a period of 80 years. Secondary-growth high-elevation lodgepole pine forests in the Lake Tahoe Basin showed a similar pattern, with a decrease in average tree diameter between presettlement and contemporary time periods (Taylor 2004, Taylor et al. 2014). Maxwell et al. (2014) also documented a shift away from larger diameter trees in reference forests of the Lake Tahoe Basin to smaller diameter trees in contemporary unlogged forests.

In the interior Columbia River basin, and portions of the Klamath and Great Basins, Hessburg et al. (2000) constructed historical and current vegetation maps from aerial photographs taken in 1932–1966 and 1981–1993, respectively. Across their large study area, which included but was not limited to subalpine forest types in the assessment area, they found that large (>63.5 cm d.b.h.) and medium (40.5 to 63.5 cm d.b.h.) trees were historically more widespread than they are currently (Hessburg et al. 2000).

The distribution of size classes in contemporary high-elevation forest types of the assessment area shows similar patterns to the Sierra Nevada and interior Northwest, specifically a greater abundance of trees in smaller size classes. For example, Eckert (2006) studied 25 undisturbed foxtail pine stands in the Klamath Mountains and found that all sampled stands assumed a reverse J-shaped distribution with the first age class representing 34 to 69 percent of the sampled trees. The USFS Forest Inventory and Analysis program also collected data from 37 subalpine stands across the assessment area between 2001 and 2014, in areas that were both managed (e.g., logged, burned, etc.) and unmanaged; these data suggest that contemporary subalpine forests have a greater proportion of trees in the smaller size classes and fewer trees in the larger diameter (>76.2 cm d.b.h.) size classes (fig. 4.9).

Current and future deviation from NRV—Based on historical observations, as well as studies conducted in subalpine habitats outside of the assessment area, it is likely that contemporary subalpine forests in the assessment area have experienced a shift in tree size class distribution toward smaller size classes. However, a lack of quantitative historical data within the assessment area also makes it difficult to confidently identify this trend. Changes in size class distribution within contemporary subalpine forests may be the result of increasing daily minimum temperature and precipitation over the past several decades, which can increase tree regeneration, recruitment, and large-tree mortality rates in subalpine tree species (Dolanc et al. 2013a). Projected changes in climate, which will likely include warmer temperatures in subalpine habitats, could lead to further increases in tree density (especially at higher elevations) and will likely intensify the tendency toward a reverse J-shaped size class distribution.

Basal area—

No historical data were found that provide estimates of basal area in presettlement subalpine forests in the assessment area. There are a few studies that provide

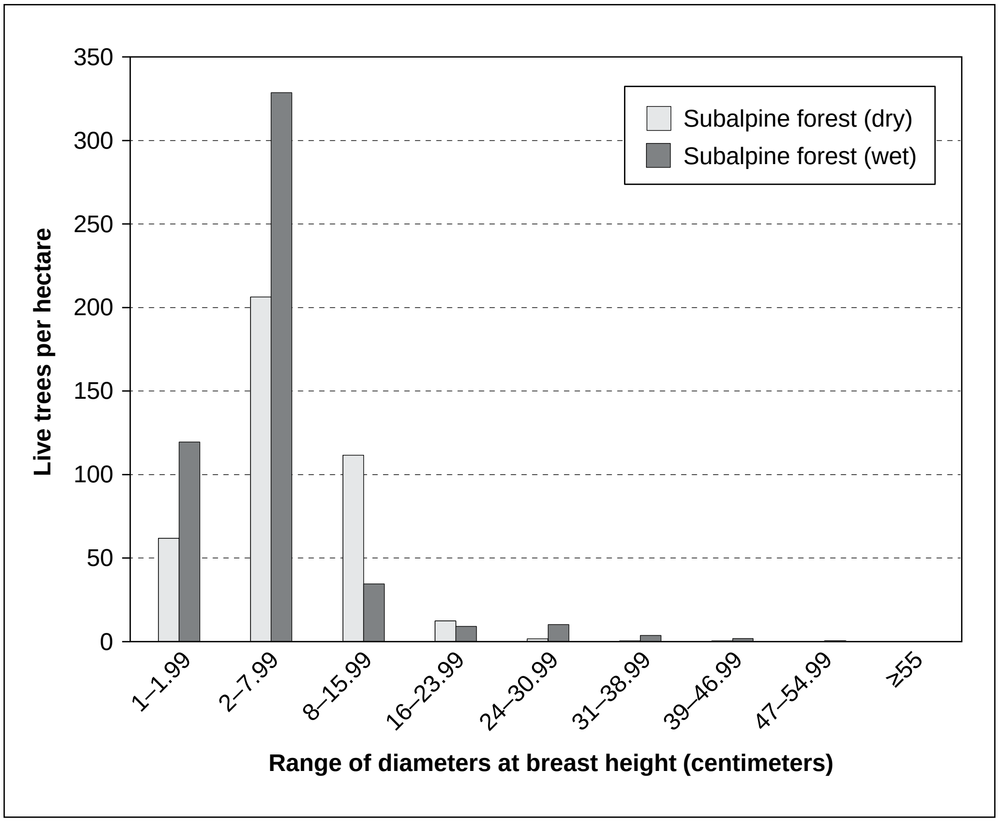

Figure 4.9—Tree size class distributions of subalpine stands in the assessment area based on a total of 37 U.S. Forest Service Forest Inventory and Analysis plots (2001– 2014) in subalpine forests (wet and dry types) covering the entire assessment area.

basal area estimates in unmanaged subalpine stands (i.e., potential reference sites); however, the limited number of forest types represented by these data generally do not have analogous estimates for managed stands, making comparisons difficult. A limited number of studies conducted in the Sierra Nevada suggest that the basal area in contemporary forests may not be outside the NRV. For example, stand- and landscape-scale comparisons between presettlement and contemporary lodgepole pine stands in the Lake Tahoe Basin found no evidence for significant changes in basal area (Maxwell et al. 2014, Taylor 2004, Taylor et al. 2014). In contrast, an unpublished monitoring report from whitebark pine forests in the Cascade Range of Oregon and Washington suggest that there have been substantial changes in basal area over a period of 200 years (time period: 1804–2004) (Siderius and Murray 2005). Using stand reconstructions, Siderius and Murray (2005) documented dramatic (more than 200 percent) increases in the average basal area for all subalpine tree species, with pronounced increases in shade-tolerant species such as subalpine fir and Shasta red fir.

Current and future deviation from NRV—Modern basal area estimates for subalpine forests in the assessment are likely within or slightly higher than basal areas in presettlement times. Variation in basal area often depends on the relative effects of fire exclusion (which may increase basal area) and logging or other forest mortality agents (which decrease basal area). Future predicted changes in basal area are also variable. For example, Lenihan et al. (2008) modeled vegetation distribution and productivity in California under three future climate change scenarios. Under the moderate warming scenario, which also featured similar precipitation to today, the extent of tree-dominated vegetation in the assessment area increased substantially (although some loss of conifer forest occurred). In contrast, under the two drier and warmer scenarios, significant loss of conifer forest occurred (much of it to fire). A recent study by Young et al. (2017) found that tree mortality across California, including high-elevation forest types, was highest in areas that were both dry and dense. This suggests that subalpine areas with higher basal area may be at higher risk of mortality in a hotter and drier climate.

Canopy cover—

Canopy cover in subalpine forest types is generally low (e.g., less than 50 percent) and much more open than in lower elevation montane forest types (Whittaker 1960). Contemporary reference sites (i.e., those that have been undisturbed by past management) can provide some insight into NRV for canopy cover within the assessment area. In the Red Butte–Red Fir Ridge Research Natural Area on the south side of Mount Shasta, whitebark pine woodlands are very open with canopy cover that is generally less than 20 percent, with the highest cover at lower elevations of the zone (Cheng 2004). Monitoring in the Mount Eddy Research Natural Area on the Shasta-Trinity National Forest estimated canopy cover of western white pine and foxtail pine forests of 20 percent and 18 percent, respectively (Cheng 2004, Keeler-Wolf 1990). Lower elevation mountain hemlock forests generally have higher canopy cover than the more open, higher elevation forest types. For example, in the Antelope Creek Lakes Research Natural Area on the Klamath National Forest, mountain hemlock forests were estimated to have more than 59 percent canopy cover (Cheng 2004, Keeler-Wolf 1990). In mountain hemlock forests in the Klamath Mountains, canopy cover can average 70 percent (Atzet and Wheeler 1982). Canopy gaps were historically an important component in these lower elevation zones of subalpine forests. In British Columbia, a study conducted by Lertzman and Krebs (1991) in the lower edge of the mountain hemlock zone documented canopy gaps that ranged from 5 to 525 m^2 (mean: 77 m^2; median: 41 m^2). They found that many small gaps were created from the mortality of single trees, which were then slowly filled with shade-tolerant subalpine species such as Pacific silver fir. In general, it takes a very long time to develop closed conditions in subalpine forests; however, once closed, the forest canopy may remain closed for centuries (Agee 1993).

The USFS Forest Inventory and Analysis program collected data from 37 subalpine stands across the assessment area. Analysis of this more recent data documented a mean canopy cover for contemporary subalpine forests of 19 percent ± 20 percent (SD). This low value may be due to the method used to calculate canopy cover in plots (see Safford and Stevens 2017 for discussion) or the fact that plots represent an unbiased sample that includes managed (e.g., logged) and unmanaged stands; areas that have burned, including those affected by high-severity wildfire; and low-productivity sites.

Current and future deviation from NRV—Canopy cover comparisons between presettlement and contemporary stands in the assessment area are limited by the scarcity of quantitative information from historical or contemporary reference stands and the high degree of variability among stands. Nevertheless, based on contemporary reference stand information, canopy cover of subalpine stands is likely within NRV. Exceptions to this include contemporary forests that have experienced an increase in tree density (see discussion above), which will likely have concurrent increases in canopy cover.

Tree regeneration—

Tree establishment in subalpine forests is dependent on several factors, including seed source and dispersal, disturbance regime, and climate. Warmer temperatures have been shown to have a positive effect on recruitment in some subalpine forest types, particularly for those stands situated at or near treeline. For example, in his study of mountain hemlock in Lassen Volcanic National Park, Taylor (1995) found that seedling establishment was highest during two historical time periods that were characterized by warm temperatures and low April snowpack. Another study from Mount Rainier found a positive association between a warm spring and subalpine fir establishment, owing to an extended establishment period when snowpack melted early (Little 1992 in Agee 1993). At Lake Mountain, in the Klamath Mountains, foxtail pine (the northern most stand) has expanded its range and established on northeastern slopes during drought years with less snowpack and a longer growing season (Tomback et al. 2011). Zald et al. (2012) investigated patterns of mountain hemlock invasion over time in subalpine meadows of the Mount Jefferson Wilderness in the central Cascades of Oregon. They found a large increase in mountain hemlock regeneration over a period of 50 years, which they attributed to a number of interacting factors, including lower snow depth (Zald et al. 2012). Collectively, these findings suggest that warmer temperatures, combined with longer snow-free periods, can exert a strong influence on seedling establishment and recruitment for subalpine species.

Note that the influence of climatic variables on regeneration patterns varies both locally (i.e., within stands) and regionally. On cold, wet sites, increases in temperature or decreases in precipitation can create favorable conditions for seedling establishment (Franklin et al. 1971); whereas in warmer, drier sites, these same trends can create conditions unsuitable for tree regeneration (Kuramoto and Bliss 1970). For example, in his study of treeline populations of foxtail pine in the southern Sierra Nevada, Lloyd (1997) found a negative relationship between summer temperatures and seedling recruitment, which he attributed to climatic water deficit. In areas like these, which may be more arid or windy, snow can facilitate tree establishment by increasing soil moisture during the growing season, reducing desiccation, and moderating temperature extremes (Holtmeier and Broll 1992, Lloyd 1997).

On cold, wet sites, increases in temperature or decreases in precipitation can create favorable conditions for seedling establishment whereas in warmer, drier sites, these same trends can create conditions unsuitable for tree regeneration.

Disturbances, like fire, avalanches, and insect-related mortality, can also affect regeneration patterns in subalpine habitats. Forest development is often very slow because of the harsh conditions found in these high-elevation habitats; therefore, disturbances that kill all or most of a stand can create almost permanent meadows or open parklands that will persist for long periods of time (Agee 1993, Franklin and Dyrness 1988). Lodgepole pine is often present in subalpine stands as a result of fire. Dickman and Cook (1989) found that shorter FRIs in mountain hemlock stands in the southern Cascades of Oregon favor dominance of lodgepole pine. In his surveys of the Cascades, Leiberg made several observations about lodgepole pine regeneration and succession in mountain hemlock forests:

> At the present time the type [mountain hemlock] is losing ground, in so far as the preponderance of the alpine-hemlock is in question. This is wholly due to forest fires which, wherever they burn in the subalpine elevations below 7,800 feet, are followed by reforestations in which the lodge pole-pine element prevails from a ratio as high as 60 per cent to total. The suppression of the hemlock through this cause is doubtless only temporary, although a century or two may elapse before it completely re-covers the lost ground [Leiberg 1900: 261].
>
> Reforestations of lodgepole pine are of a transient character at high elevations in the southern Cascades. After a longer or shorter period of time the soil in stands of lodgepole pine regains its former ratio of moisture. This is proved by the occurrence of reforestations of lodgepole pine, which are giving way to species requiring a higher ratio of soil humidity. The more common species which replace lodgepole pine stands at the higher elevations are alpine hemlock [mountain hemlock] and noble fir; at lower levels white and red fir, while in the upper canyons of the South Fork of Rogue River, Engelmann spruce is sometimes found replacing old, mature, and decaying growths [Leiberg 1900: 284].

Current and future deviation from NRV—The limited number of studies and historical observations suggest that tree regeneration in subalpine forest is generally within or above NRV. This is particularly true when comparing recent 20th century trends with longer historical time periods that include both the Little Ice Age and the Medieval Warm Period (table 1.1). Although recruitment patterns in subalpine forests can be complex and variable, overall there is a general trend of increasing recruitment, particularly in higher elevations that are at or near tree line.

Understory vegetation cover—

There are very few descriptions of understory cover in subalpine forests that predate the period of extensive sheep grazing in these high-elevation habitats (i.e., before the late 1800s). In his extensive travels of montane forests in the Sierra Nevada, William Brewer (1930: 520) noted that “the ground under the [subalpine] tree is generally nearly bare. There is but little grass or undergrowth of either herbs or bushes.” Studies conducted in contemporary reference areas within the assessment area provide some insight into the NRV for understory cover in subalpine forest types. For example, in the Antelope Creek Lakes Research Natural Area on the Klamath National Forest, Cheng (2004) noted that whitebark pine stands are characterized by an open understory with scattered shrubs that include bush chinquapin (*Chrysolepis sempervirens*) and greenleaf manzanita (*Arctostaphylos patula*). Cheng also describes mountain hemlock forests in the research natural area as having very little understory development and a lighter litter and duff layer than lower elevation red fir forests. Atzet and Wheeler (1982) noted that shrubs and herbs in the understory of mountain hemlock forests in the Klamath Mountains quite often total less than 10 percent cover. The dominant understory shrub and herbaceous species in their study area was deer oak (*Quercus sadleriana*) and one-sided wintergreen (*Orthilia secunda*). In western white pine forests in the Mount Eddy Research Natural Area on the Shasta-Trinity National Forest, the understory is sparse (i.e., <15 percent) owing to the presence of serpentine soils and is dominated by pine-mat manzanita (*Arctostaphylos nevadensis*) and huckleberry oak (*Quercus vacciniifolia*) (Cheng 2004). On more xeric, higher elevation sites, understory cover can reach up to 65 percent, while shrub cover in foxtail pine forests of this research natural area ranges from close to zero up to about 25 percent (Cheng 2004).

Whittaker (1960) did elevational transects in the Siskiyou Mountains and noted a wide range of herbaceous coverage (1 to 90 percent) in subalpine forests, ranging from relatively open stands with lush herbaceous understory to denser stands with almost bare understory. He noted that understory cover decreased along a moisture gradient, from north- to south-facing slopes, and from lower to higher elevations. He also noted that understory cover was inversely correlated with tree density (Whittaker 1960).

Current and future deviation from NRV—Taken collectively, these historical accounts and contemporary studies suggest that presettlement understory plant cover was spatially variable and relatively sparse in many subalpine stands, especially outside of meadows, riparian areas, and mesic microhabitats. Studies of subalpine forests in the Sierra Nevada (i.e., Potter 1998, Vankat and Major 1978) suggest that understory cover in contemporary subalpine forests largely mirror this general pattern and are driven to a large extent by variability in the underlying substrate and soil moisture. For example, an analysis of data collected between 2009 and 2014 from subalpine stands across the assessment area by the USFS Forest Inventory and Analysis program provides an estimated mean cover of 16 percent (± 19 percent [SD]) for shrubs and 7 percent (± 11 percent [SD]) for herbaceous species. This general consistency between presettlement and current conditions implies that understory cover in contemporary subalpine forests is within NRV.

Presettlement understory plant cover was spatially variable and relatively sparse in many subalpine stands, especially outside of meadows, riparian areas, and mesic microhabitats.

Spatial patterns—

The spatial distribution of trees in subalpine forests presettlement likely varied in response to physical site conditions, disturbance intensity, and the ecological attributes (e.g., seed dispersal, shade tolerance, etc.) of different subalpine tree species (Agee 1993). At higher elevations of the subalpine zone, where conditions were the harshest, the forest was characterized by widely spaced individual trees or clumps of trees; whereas at lower elevations, the forest was a more continuous mosaic of single trees, tree clusters, and canopy openings (Gedalof and Smith 2001). Historical photographs, like those taken by Jervie Henry Eastman in the 1930s (figs. 4.7 and 4.8), provide insight into these patterns. They show relatively dense stands of trees in the subalpine zone of the Trinity Alps and Marble Mountains that were patchy and interspersed with herbaceous parkland, rock, or snow.

In early forest surveys of the Cascades, John Leiberg (1900: 264) observed spatial clumping in mountain hemlock (which he referred to as alpine hemlock) and other subalpine species:

> The pure growths of alpine hemlock [mountain hemlock] are distinguished also by another peculiarity, that of forming associations of ten to twenty-five individuals which stand so close that apparently they all spring from a common root. These small aggregations are extremely characteristic of the habit of the species at the middle and upper altitudinal limits of its growths. Numbers of these associations, more or less closely grouped together, form a shade so dense that the other component species of the alpine-hemlock type stand little chance of growth in such places. We have,

> therefore, pure growths of the alpine hemlock [mountain hemlock]. At lower elevations, or in reforestations after fires, the alpine hemlock rarely forms these small associations, and in consequence the growth is much mixed with other species.

John Muir (1894) made a similar observation for western white pine in the subalpine zone of the Sierra Nevada, where he noted that trees “grow in clusters of from three to six or seven.” Historical accounts from the Sierra Nevada by Muir (1894), Leiberg (1902), and Harris (1939) also include observations about “scattered” trees in stands of foxtail pine and whitebark pine. Many current unlogged subalpine stands have a comparable spatial distribution of trees (Meyer and North 2019).

Research conducted outside of the assessment area suggests that high-elevation forests that were logged during the late 19th century can differ substantially from presettlement stands with respect to tree spatial patterns and structural diversity. In his study of high-elevation lodgepole pine stands in the Lake Tahoe Basin, Taylor (2004) found that presettlement forests had greater structural complexity than contemporary stands that had been logged in the late 19th century. He used Shannon’s diversity index to examine the number of different tree size classes (measure of richness), as well as the number of trees within each of these size classes (measure of evenness), and found that presettlement lodgepole pine stands had significantly greater structural diversity than contemporary stands. In these historical stands, the largest trees (≥40 cm at stump height) were clumped at all spatial scales and small-diameter (10 to 40 cm) trees were randomly distributed. This spatial random pattern is consistent with the process of thinning by both low-severity fire and competition (Taylor 2004). Contemporary stands displayed the opposite pattern, with small-diameter trees that were clumped at all spatial scales, and large trees that were randomly distributed. These results suggest that, although forest structure at both time periods was highly heterogeneous, the underlying spatial pattern has changed. Taylor (2004) suggests that the lower structural diversity in contemporary lodgepole pine forests of the Lake Tahoe Basin was the result of logging rather than the exclusion of fire.

Current and future deviation from NRV—Based on historical observations and contemporary stand information, tree spatial patterns in unlogged subalpine forests are likely within NRV. Exceptions to this may include contemporary stands that were logged in the past or areas where fire has been excluded for an extended period of time; these areas may be slightly less heterogeneous in terms of spatial structure and exhibit higher small tree density and continuous canopy cover than unlogged, reference landscapes.

Seral class proportions—

The interagency LANDFIRE Program developed a nationwide map of potential vegetation (referred to as Biophysical Settings [Bps]) representing the distributions of vegetation types as they are hypothesized to have existed presettlement. LANDFIRE BpS modeling suggests that historical reference conditions in subalpine forests of the assessment area were dominated by mid- and late-seral classes (fig. 4.10). An exception to this occurs in the upper elevations of the subalpine zone, where forest transitions to alpine. This vegetation type is represented as mesic parkland in the North Pacific Maritime Mesic Subalpine Parkland BpS model (dominated by mountain hemlock, subalpine fir, and Pacific silver fir) and is characterized by early-seral classes (fig. 4.10). BpS models predict that within the mid- and late-seral stages, subalpine forests of the assessment area had higher proportions of open than closed-canopy conditions (LANDFIRE 2017).

Hessburg et al. (1999a) used aerial photographs taken in 1932–1966 and 1981–1993 to compare recent historical and current conditions and develop an estimate of the range of variation for vegetation types in the Cascade Range of Washington state. They found that the total area occupied by different forest structural classes in whitebark pine–subalpine larch (*Larix lyallii*) and subalpine fir–Englemann spruce forest types did not differ significantly between these two time periods and were within the range of variation (fig. 4.11). Subalpine forests within the watershed they examined were historically dominated by mid-seral structural classes, and this pattern was similar in current periods, with the exception of lodgepole pine, which had an increase in early-seral classes in the current period. Although the percentage of the landscape occupied by subalpine seral stages was within the range of variation, Hessburg et al. (1999a) did find that the mean patch size for some structural classes was outside of the range of variation. For example, the mean patch size of whitebark pine–subalpine larch forests in two mid-seral classes (understory reinitiation and stem exclusion, open canopy) were well above the historical estimate and outside of the estimated range of variation.

Current and future deviation from NRV—The current proportion of seral stages in subalpine forests of the assessment area is generally within NRV. However, recent historical (1932–1993) trends suggest that the patch size of some mid-seral structural classes may be approaching or possibly exceeding the upper limit of this NRV.

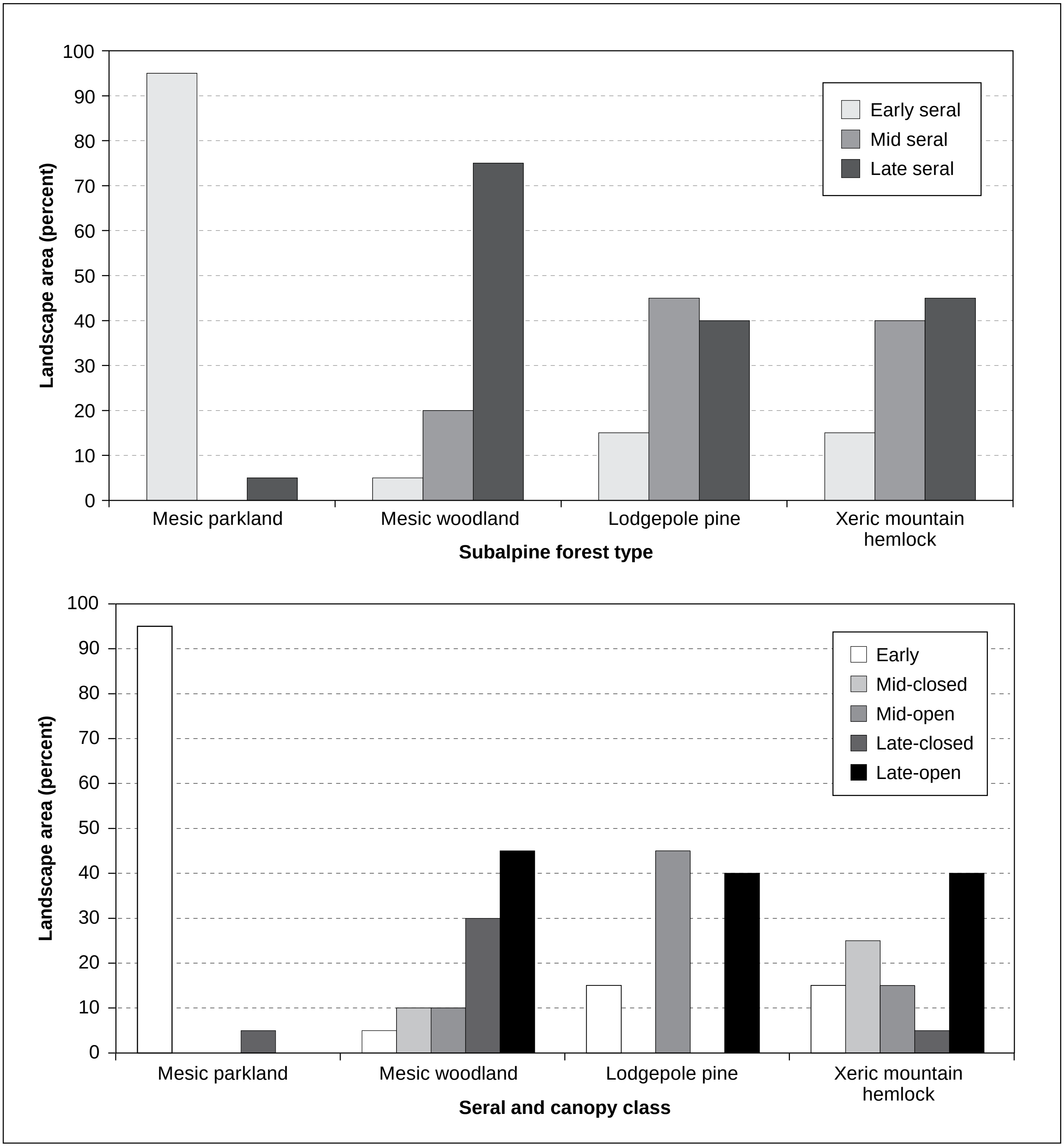

Figure 4.10—Percentage of the landscape occupied by subalpine forests in different seral stages based on LANDFIRE biophysical setting models for (1) mesic parkland (clumps of trees interspersed with low shrublands and meadows); (2) mesic woodlands (dominated by mountain hemlock, lodgepole pine, and red fir, with a significant western white pine component); (3) lodgepole pine forest and woodland (dominated by lodgepole pine, but can include Douglas-fir and white fir in warmer areas and western white pine and Shasta red fir in cooler areas); and (4) xeric mountain hemlock (dominated by mountain hemlock with a mixture of other subalpine tree species). The bottom figure displays the open and closed canopy subclasses within mid- and late-seral classes (LANDFIRE 2017).

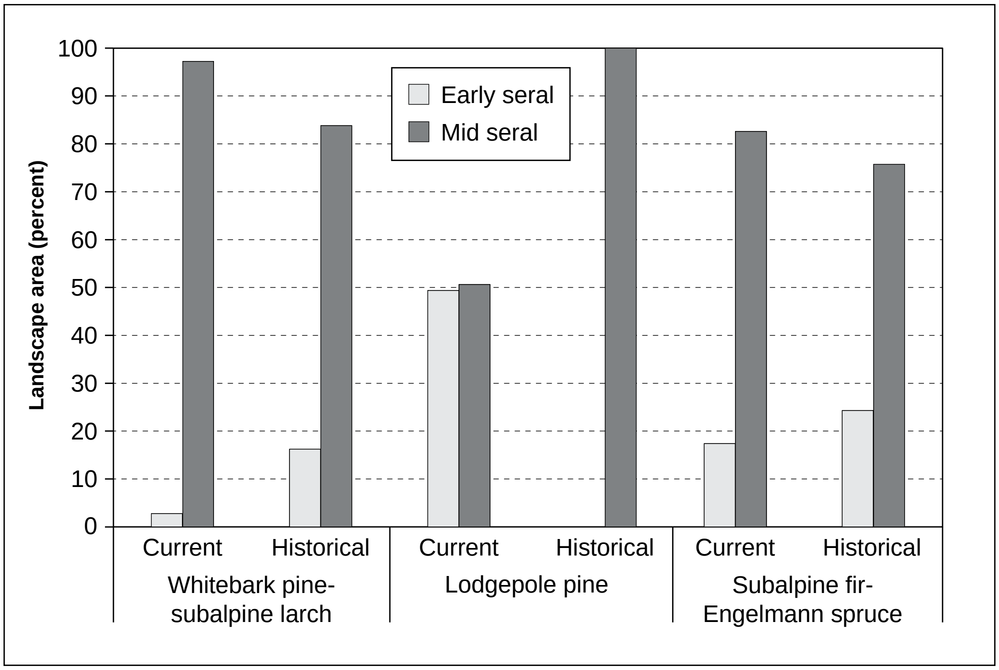

Figure 4.11—Re-created using data presented in Hessburg et al. (1999b) summarizing their analysis of stand structure in current (1981–1993) and historical (1932–1966) subalpine forest types in the Cascades of eastern Washington. Early-seral stand structure includes the stand initiation class; mid-seral includes the following classes: stem exclusion–closed canopy, stem exclusion–open canopy, understory reinitiation, and young forest multistory. The percentage of the landscape was determined by calculating the percentage of area occupied by that forest type in the study area (i.e., watershed).

Composition

Overstory species composition in presettlement subalpine forests varied across the assessment area in response to underlying environmental conditions, elevational gradients, disturbance processes, and the ecology of individual tree species. In general, subalpine forests were (and are today) dominated by conifers. Although less common, several deciduous, broadleaf species (e.g., aspen [*Populus tremuloides*]) were also found in moist subalpine environments such as riparian corridors, meadow fringes, and areas with abundant soil moisture (Millar and Rundel 2016).

Leiberg (1900) made several observations about the composition of subalpine forests during his early surveys of the southern Cascade Range of Oregon. His insights provide information about the diversity of conifer species that historically occurred in subalpine habitats of the assessment area, as well as their relative abundance.

> The timber-line tree is the white-bark pine. A hundred feet or so below its upper limit the alpine hemlock [mountain hemlock] comes in, soon joined by the noble fir and alpine fir [subalpine fir], with lodgepole pine and an Engelmann spruce here and there [Leiberg 1900: 244].

> The species which form the integral parts of the alpine-hemlock type of forest are: lodgepole, white, and white-bark pine; noble and alpine fir; alpine hemlock;

> and Engelmann spruce. With the exception of the white and white-bark pine and Engelmann spruce, the ratio in which the species exist in the type is largely a matter of chance…..What the relative ratio of the species belonging to the type would be on any large area undisturbed by fire for a century or more, there are no means of knowing, although it is hardly to be doubted that the alpine hemlock would show a preponderating ratio [Leiberg 1900: 259–260].
>
> White pine, alpine fir, and Engelmann spruce mostly occur as scattered individuals or small aggregations in mixed stands of alpine hemlock and noble fir. Occasionally, in the high country between Mount Pitt[2] and Klamath Point, the spruce in some of the canyon bottoms is present in as high ratio as 75 per cent [Leiberg 1900: 265].
>
> On small areas the alpine-hemlock type is frequently pure—that is to say, composed entirely of alpine hemlock. On larger areas the type forms stands in which the same species prevails to the extent of 75 to 85 per cent. The few species of conifers which thrive at the usual altitudes of the type make the chances for stands of pure growths far more numerous and certain than is the case among the lower altitude types with their more complex composition [Leiberg 1900: 259].
>
> The general ratio of alpine hemlock in the composition of the type is approximately 50 per cent as regards numbers of individuals, and 60 per cent if reference be had to the timber volume. The species which crowds it most closely as to numbers is the lodgepole pine, but it is much inferior to the hemlock in timber volume. Typical alpine-hemlock forests contain 60 to 80 per cent of the species [Leiberg 1900: 260].

In one instance, Leiberg (1900) estimated the composition of forest at various elevations on what is present-day Mount McLoughlin in Oregon:

Location	Species	Percentage
Timberline	Whitebark pine	100
91 m (300 ft) below timber line	Whitebark pine	50
	Mountain hemlock	50
244 m (800 ft) below timber line	Whitebark pine	2
	Mountain hemlock	88
	Noble fir	10

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[2] This is present-day Mount McLoughlin, situated in the northeastern part of the assessment area.

In the Klamath Mountains, foxtail pine can be the dominant tree in extensive stands at high elevations on isolated peaks and ridges (Eckert and Sawyer 2002). These northern stands of foxtail pine are generally more diverse than foxtail pine stands in the southern Sierra Nevada (Mastrogiuseppe and Mastrogiuseppe 1980), with the highest diversity in the Trinity and Marble Mountains in the northernmost portion of the range (Eckert and Sawyer 2002). Stands of exceptional species diversity are also found in some subalpine forests of the Klamath Mountains (e.g., Crater Creek and Sugar Creek Research Natural Areas); in these areas, as many as six conifer species can be found in a mixed subalpine forest type (Cheng 2004).

Studies from the Sierra Nevada bioregion suggest that the composition of contemporary subalpine forests is comparable to historical stand composition (Meyer and North 2019). For example, Dolanc et al. (2013a) resampled historical vegetation plots (Wieslander Vegetation Type Mapping, 1929–1934) in the central Sierra Nevada and found that changes in the relative frequency of high-elevation tree species was relatively minor, leading to the conclusion that species composition has remained relatively unchanged over the past 70 to 140 years. Another set of studies conducted in the Lake Tahoe Basin determined that lodgepole pine forests at the lower elevational limit of subalpine forests did not change significantly in terms of tree species composition between presettlement and current periods (Taylor 2004, Taylor et al. 2014).

Other studies suggest that the species composition of subalpine forests has likely shifted over the past century. Skovlin et al. (2001) analyzed repeat photography in high-elevation habitats of northeastern Oregon and found that subalpine fir trees had encroached into areas that formerly had little or no fir. Smithers et al. (2018) documents a compositional shift in subalpine regeneration in the White Mountains from Great basin bristlecone (*Pinus longaeva*) to limber pine (*P. flexilis*). Siderius and Murray (2005) used stand reconstructions to evaluate changes in whitebark pine forests in the Cascade Range between 1804 and 2004 and found a dramatic increase in late-seral tree species (i.e., subalpine fir and Shasta red fir) over the past 80 years. Monitoring of whitebark pine stands in Crater Lake National Park has also documented encroachment of whitebark pine stands, in this case by mountain hemlock, which Jules et al. (2017) have suggested is the result of fire suppression. In the Cascade-Klamath region and Warner Mountains, Slaton et al. (2019) observed high densities of non-whitebark pine regeneration in whitebark pine dominated stands, which could also be attributed to the absence of fire.

Current and future deviation from NRV—Historical observations indicate that presettlement subalpine forests supported a diverse mix of tree species in relative abundances that are likely comparable to present-day subalpine forests of the assessment area. Stand reconstruction studies from outside of the assessment area indicate that tree species composition in subalpine forests is likely within NRV. However, observations and monitoring also suggest that altered disturbance regimes (e.g., long-term exclusion of fire) may result in species composition shifts, with future increases in late-seral, shade-tolerant species.

Chapter 5: Projected Future Conditions and Trends

Background

Future climatic change is often projected from statistical or dynamical downscaled global climate models (GCMs). Assumptions inherent to each alternative greenhouse gas emission scenario and GCM (based on the type of atmospheric general circulation model) influence model projections. The use of multiple GCMs or emission scenarios provides a more comprehensive outlook of the future effects of climate change on a region, biome, or species of interest. For example, the National Center for Atmospheric Research Parallel Climate Model (PCM) projects warmer and similar (no significant change in) precipitation conditions in California; whereas the National Oceanic and Atmospheric Administration Geophysical Fluid Dynamics Laboratory (GFDL) model projects hotter and drier conditions for the state (Cayan et al. 2006). The spatial resolution of these models usually varies from 160 to 800 km^2 for GCMs to 800 m^2 through 50 km^2 for downscaled models, although much higher resolutions are available. The relatively lower resolution of GCMs necessitates analysis at regional or large landscape scales. Temporally, model projections are typically presented in 10-, 20-, or 30-year intervals.

In addition to projections in future climate, ecological response models may assess the response of ecological variables to climate change. These models vary from qualitative conceptual models to quantitative niche-based (e.g., Maximum Entropy or Maxent) and dynamic vegetation models (e.g., MC1). Model outputs may project changes in the climatic envelope of an individual species (e.g., red fir), vegetation type (e.g., red fir forest), or biome (e.g., evergreen conifer forest).

Model Projections

Temperature and precipitation—

Although most future-climate modeling efforts examine areas that are much broader than northwestern California and southwestern Oregon, there are a few studies that have examined landscapes within the assessment area (e.g., Barr et al. 2010, 2011; Doppelt et al. 2008; Lenihan et al. 2003, 2008; Halofsky et al. 2019). Collectively, these studies are in general agreement that mean annual and monthly temperatures are projected to increase within the assessment area by the end of the century. For example, Barr et al. (2010) used three GCMs and an MC1 to project future changes in temperature for the Klamath Basin; they projected an increase in annual average temperature of 1.1 to 2.0 °C by mid-century and 2.5 to 4.6 °C by late century. Doppelt et al. (2008) used a similar modeling approach for the Rogue River Basin and projected annual average temperature increases of 0.5 to 1.6 °C by around 2040 and 2.2 to 4.4 °C by around 2080. Comparable increases were projected from an analysis of 31 GCMs for south-central Oregon (Halofsky et

Mean annual and monthly temperatures are projected to increase within the assessment area by the end of the century.

al. 2019) and three GCMs for the Deschutes Basin in Oregon (Barr et al. 2011). In south-central Oregon, mean temperatures in the coldest and warmest months were projected to increase, with most GCM models projecting mean temperature in the coldest months rising to above freezing by the end of the century (Halofsky et al. 2019). Modeling completed for the Pacific Northwest by Mote and Salathé (2010) project rates of warming that range from 0.1 to 0.6 °C per decade.

Precipitation projections are generally less certain than those for temperature. GCM projections of precipitation over California tend to disagree on the sign of change (e.g., Berg et al. 2015, Das et al. 2013, Neelin et al. 2013) and projected trends throughout the 21st century are generally insignificant (Berg and Hall 2015). Despite global projections for increases in precipitation extremes (Kharin et al. 2007, 2013; Sun et al. 2007), Cayan et al. (2008) found only modest increases in the number and magnitude of large precipitation events in California. However, Berg and Hall's (2015) analysis of 34 GCMs project increases in the frequency of interannual precipitation extremes over California, with extremely dry wet seasons becoming roughly 1.5 to 2 times more common and wet extremes generally tripling in frequency by the end of the century. Despite only small changes in mean precipitation, Swain et al. (2018) found evidence for an approximately 25 percent increase in rapid transitions between dry and wet conditions in northern California by the end of the 21st century. The north and south coasts of California are also projected to experience the largest increase in humid nighttime heatwave events (Gershunov and Guirguis 2012). Recent modeling conducted by Cvijanovic et al. (2017) also suggests that expected Arctic sea-ice loss in the next few decades could substantially affect California's precipitation patterns, resulting in increased incidence and severity of future droughts.

Although climate models diverge with respect to future trends in precipitation over northwestern California, there is widespread agreement that the trend toward lower snow-water equivalent and earlier snowmelt will continue. Results from a downscaling of three climate models for the Rogue River Basin in southwest Oregon suggest that rising temperatures will increase the percentage of precipitation falling as rain and decrease snowpack considerably, particularly at lower elevations; one model projected that snowpack will be reduced 75 percent from the baseline by 2040 to an insignificant amount by 2080 (Doppelt et al. 2008). Hayhoe et al. (2004) used two contrasting GCMs (much warmer and drier vs. somewhat warmer and drier) under low and high greenhouse gas emissions scenarios to make projections of climate change impact for California over the next century. By 2100, under all GCM emissions scenarios, April 1 snowfall was down by -22 percent to -93 percent in the 2042- to 3048-m elevation belt, and the date of peak snowmelt was projected to occur from 3 to 24 days earlier in the season.

Thorne et al. (2015) used the Basin Characterization Model (BCM), a regional water balance model, to assess the response of snowpack, climatic water deficit, recharge, and runoff to changing future (2070–2099) climate across the 5,135 watersheds in California. Under both the PCM (warmer and similar) and GFDL (hotter and drier) scenarios, climatic water deficit increased by an estimated 110 to 122 percent in northwestern California and 115 to 129 percent in the southern Cascades. Snowpack, runoff, and recharge also continued to decrease and were amplified under future projections for California. April 1 snowpack was projected to decrease in northwestern California from a current mean of 52 to 9 mm (PCM) and 3 mm (GFDL) and in the southern Cascades from a mean of 181 to 52 mm (PCM) and 20 mm (GFDL). Watersheds west of Mount Shasta, where relatively little change has occurred historically, are predicted to experience high levels of change under both future projections (Thorne et al. 2015).

Tree growth in high-elevation forests is often limited by cold temperatures; therefore, projected increases in temperature could increase the growth and productivity of red fir and subalpine forests in areas where there is adequate moisture (Chmura et al. 2011, Salzer et al. 2009). For example, Graumlich et al. (1989) found that the productivity of four high-elevation stands in Washington increased by 60 percent over the past century; they related this increase most strongly to increasing summer temperatures and associated changes in absorbed solar radiation. Monitoring of mountain hemlock stands at Crater Lake National Park found evidence of a strong inverse relationship between growth and total cool season precipitation, and a positive (albeit weaker) relationship with mean summer temperature (Appleton and St. George 2014).

Aside from these potential positive responses, projected changes in climate, which include decreased snowpack, earlier snowmelt, increased summer evapotranspiration, and increases in the frequency and severity of drought will pose a significant challenge for many high-elevation tree species, particularly in the assessment area where the highest elevations generally reach less than 2800 m. In their study of mountain hemlock stands in southern Oregon, Peterson and Peterson (2001) found that earlier snowmelt, higher summer temperatures and lower summer precipitation negatively affected tree growth by reducing water availability. These climatic changes will be especially challenging for narrowly distributed species, such as Brewer spruce, which are often limited to small, geographically disjunct populations (Ledig et al. 2012).

Projected changes in climate, which include decreased snowpack, earlier snowmelt, increased summer evapotranspiration, and increases in the frequency and severity of drought will pose a significant challenge for many high-elevation tree species.

A limited number of ecological response models have focused on red fir or subalpine forests at a broad, regional scale; projected changes in the distribution of these forest types are summarized in the sections below. These ecological response models provide many insights into the potential broad-scale impacts of climate change to tree species (McKenney et al. 2007, Shafer et al. 2001), but results from these models should be interpreted with caution because of the many assumptions and limitations associated with them (Clark et al. 2011, Rowland et al. 2011).

Red fir distribution—

Projected changes in the distribution of red forests are summarized in table 5.1. Pooling all studies, we found that climate models projected an average range reduction of 85 percent in red fir across a range of geographic scales (subregional to entire species geographic range) by the end of the 21st century (table 5.1). All studies listed in table 5.1 used the A2 emissions scenario (regionally oriented economic development), with the exception that McKenney et al. (2007) used a combination of the A2 (regional development) and B2 (local environmental sustainability) emissions scenarios. Ecological response models included species distribution models (BioMove, ANUCLIM, Maxent, Bioclim) in four studies but also included the MC1 vegetation dynamic model for biome projections in Lenihan et al. (2003, 2008).

Thorne et al. (2016) conducted a climate vulnerability analysis for several vegetation types in California using two GCMs and two emission scenarios, selected to represent a range of future conditions in the state by the end of the 21st century. Vulnerability scores were derived from a combination of climate exposure, climate sensitivity, adaptive capacity, and spatial disruption (i.e., the need for vegetation to shift its location). They assessed vulnerability within the area currently occupied by Pacific Northwest subalpine forest (Thorne et al. 2016). This forest type is entirely within the assessment area and includes red fir, mountain hemlock, western white pine, and in the Klamath Mountains, isolated stands of Pacific silver fir, subalpine fir, and Alaska yellow-cedar. Thorne et al. (2016) determined that 60 to 93 percent of the area currently occupied by this forest type would be considered no longer suitable at the end of the century. The results of this assessment indicate that red fir forests in the assessment area are highly to moderately vulnerable to future projected climatic changes.

Most red fir forests in the assessment area will be outside the historical and contemporary climate envelope by the end of the century.

Most red fir forests in the assessment area will be outside the historical and contemporary climate envelope by the end of the century. Projected changes in the distribution of red fir forests consistently show a pronounced reduction in their geographic extent within the assessment area by 2070–2100 (table 5.1). These results are consistent with several ecological response model projections for the Pacific Northwest that show a substantial decline in the suitability or viability of red fir and other high-elevation forests in the assessment area (fig. 5.1) (Crookston et al. 2010, Michalak et al. 2017). Ultimately, the degree of climate vulnerability and adaptive capacity in red fir will be contingent on several factors not covered by

Table 5.1—Model projections describing future changes in the distribution of red fir forests within the assessment area

Unit of analysis	Geographic scope	GCM (model type) and trends[a]	Emissions scenario	Decrease[b]	Stable[b]	Increase[b]	Time period	Reference
				- - - - - - - -	*Percentage*	- - - - - - - -		
Species (red fir)	California	CCSM (BioMove)—warmer and wetter	A2	77	23	1	2080	CDFFP (2015)
Species (red fir)	California	HCM (BioMove)—hotter and drier	A2	99.9	0.1	<0.1	2080	CDFFP (2015)
Species (red fir)[c]	Species range	Ensemble of three models—full dispersal	A2 and B2	76.8	23.2	—	2071–2100	McKenney et al. (2007)
Species (red fir)[c]	Species range	Ensemble of three models—no dispersal	A2 and B2	87.5	12.5	—	2071–2100	McKenney et al. (2007)
Biome[d]	California (conifer forest)	PCM (MC1)—warmer and possibly wetter	A2	5	—	—	2071–2100	Lenihan et al. (2008)
Biome[d]	California	GFDL (MC1)—hotter and drier	A2	52	—	—	2071–2100	Lenihan et al. (2008)
Species (red fir)[e]	California	Ensemble of 11 models	A2	56	27	10	2040–2065	Southern Sierra Partnership (2010)
Species (red fir)[f]	Northwest California and Oregon	GFDL (FVS) – hotter and drier	A2	>99	<1	—	2090	Crookston et al. (2010)
Average across studies[g]:				85.3	14.7	0.5	2071–2100	

— = no estimate available.

[a] CCSM = Community Climate System Model-National Center for Atmospheric Research; HCM = Hadley Centre Model; PCM = Parallel Climate Model; GFDL = Geophysical Fluid Dynamics Laboratory; MC1 = dynamic vegetation model; and FVS = Forest Vegetation Simulator.

[b] Percentage decrease, increase, or stability indicates the percentage change in the area covered by red fir within the geographic scope and time period of each study.

[c] Estimates for percentage of stability and increase ("percentage remaining") are pooled. Includes models that assume full dispersal or no dispersal.

[d] Projections are for conifer forest biome, which includes mixed-conifer forest, red fir forest, and other conifer-dominated forest types.

[e] Decrease is defined as percentage of red fir distribution that is "stressed." Projected estimates also include an uncertain category defined as areas lacking model agreement (range: 6 to 7 percent).

[f] Decrease is defined as percentage of red fir distribution that has low viability (i.e., viability score <0.5) using the FVS model; values were estimated from visual inspection of spatial data, which are available from Data Basin (https://databasin.org/).

[g] Includes CDFFP (2015) and McKenney et al. (2007).

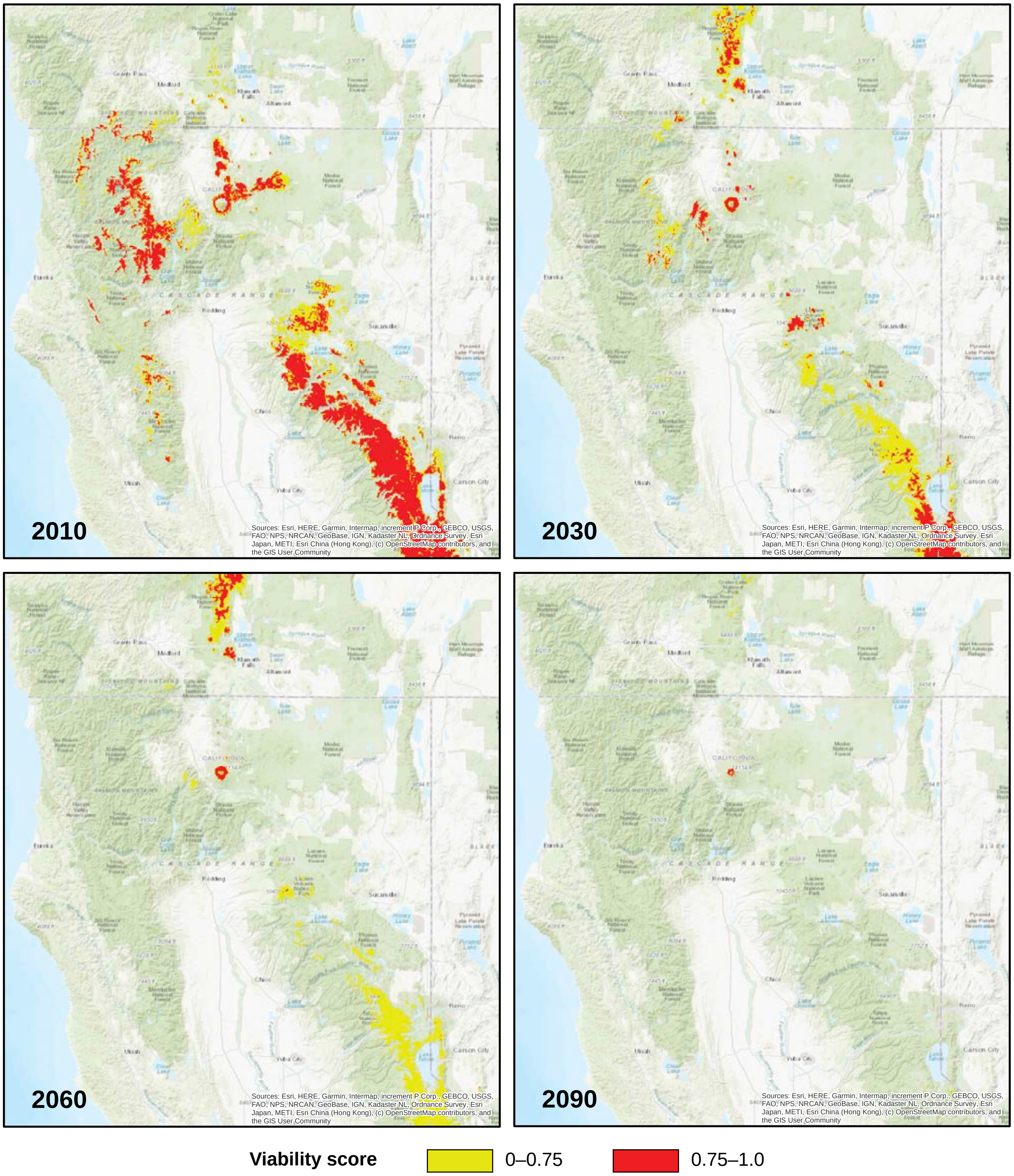

Figure 5.1—Examination of the data from Crookston et al. (2010) in Data Basin, a science-based mapping and analysis platform (https://databasin.org/), suggests a significant decline in the area considered viable for red fir by as early as 2030; viable habitat is predicted to be entirely or nearly (less than 1 percent) absent from the assessment area by 2090. Viability scores (scale of 0 to 1.0) were derived using the Geophysical Fluid Dynamics Laboratory—Princeton University, National Oceanic and Atmospheric Administration Research (GFDLCM21) general circulation model and A2 emission scenario (high emissions, regionally diverse world, rapid growth) to model future climate. Viability scores close to zero indicate a low suitability, whereas those near 1.0 indicate a suitability so high that the species is nearly always present in that climate. Red and yellow polygons on the map show areas of predicted high to moderate suitability for red fir in the future.

most species distribution models, including dispersal rates, biotic interactions, evolutionary processes (e.g., adaptation, genetic drift), physiological tolerances, edaphic constraints, interacting stressors, and forest management actions (Clark et al. 2011, Rowland et al. 2011). Consequently, red fir forests may persist in or adapt to areas of high climate exposure despite ominous model projections for red fir forests in the assessment area.

Subalpine forest distribution—

Projected changes in the distribution of subalpine forests (tables 5.2 and A1.6) consistently show a pronounced reduction in their geographic extent within the assessment area by 2070–2100. Pooling all studies, we found that climate models run under higher emission scenarios (A1, A2, and RCP 8.5) predict an average decline in subalpine forest types of 66 percent; lower emission scenarios (B1, B2, and RCP 4.5) had an average decline of 63 percent (table 5.2). In their analysis of low and high greenhouse gas emissions scenarios for California, Hayhoe et al. (2004) projected that subalpine forests will be reduced by 50 to 75 percent over the next century. Halofsky et al. (2013) analyzed shifts in vegetation in response to projected climate change in central Oregon and found that the area currently occupied by subalpine forests will decline from 2 percent (current) to less than 1 percent by the end of the century. Franklin et al. (1991) hypothesized that a mean annual temperature increase of 2.5 °C in the mountain hemlock zone of Oregon could result in a decrease in area from 9 to 2 percent; whereas an increase of 5.0 °C could effectively eliminate the species. Using a dynamic vegetation model and three GCMs, Rogers (2009) and Rogers et al. (2011) projected a 60 to 90 percent decline in subalpine forests in western Oregon and Washington by the end of the century.

Thorne et al. (2016, 2017) conducted a climate vulnerability analysis for two subalpine forest types in the assessment area: (1) Pacific Northwest subalpine forest, which is entirely within the assessment area and includes red fir, mountain

Table 5.2—Average projected change in the area occupied by subalpine forest types at the end of the century (2100). Model projections were obtained from downscaled global climate models (GCMs), climate envelopes (species distribution), and dynamic vegetation models[a]

Emissions scenario	Decrease	Stable	Number of studies
	- - - *Percentage* - - -		
Higher (A1, A2, RCP 8.5)	66	25	5
Lower (B1, B2, RCP 4.5)	62	37	3
Total	66	27	6

[a] Individual projection estimates and sources are presented in table A1.6 of the appendix.

Subalpine forests in the assessment are highly to moderately vulnerable to future projected climatic changes.

hemlock, western white pine, and in the Klamath Mountains; isolated stands of Pacific silver fir, subalpine fir, and yellow cedar; and (2) subalpine aspen forests and pine woodlands, which extend outside of the assessment area to include the Sierra Nevada and includes foxtail, whitebark, limber, bristlecone, and lodgepole pine as well as aspen. Vulnerability scores were derived from a combination of climate exposure, climate sensitivity, adaptive capacity, and spatial disruption (i.e., the need for vegetation to shift its location). Among the vegetation types they analyzed, subalpine vegetation was one of the most sensitive groups, with the lowest adaptive capacity. Thorne et al. (2016) determined that within the area currently occupied by these subalpine forest types, the area that would be no longer suitable at the end of the century ranged from 34 percent (lowest emission scenario) to 95 percent (highest emission scenario) for subalpine aspen forests and pine woodlands and 60 to 93 percent for Pacific Northwest subalpine forest (Thorne et al. 2016). Collectively, the results of this assessment indicate that subalpine forests in the assessment are highly to moderately vulnerable to future projected climatic changes.

While climate change poses a significant challenge to subalpine forests in general, it poses an ever greater threat to narrowly distributed species, like Brewer spruce that have been fragmented into small and widely separated populations (Ledig et al. 2012). A study conducted by Ledig et al. (2012) used spline climatic models to predict future suitable habitats for Brewer spruce, which is endemic to the Klamath Mountains of California and Oregon. Their projections showed a significant decline, and eventual extinction, of Brewer spruce between 2030 and 2090 owing to the disappearance of contemporary climatic conditions and a northward displacement of the climate niche by hundreds of kilometers (Ledig et al. 2012).

Crookston et al. (2010) used Forest Vegetation Simulator (FVS) to examine changes in forest composition in response to projected changes in climate. Their data, which include viability scores for several individual species, indicate the potential for a dramatic decrease in suitable habitat for many subalpine species over the next century (see fig. 5.2). Examination of the data from Crookston et al. (2010) in Data Basin, a science-based mapping and analysis platform (https://databasin.org/), suggests a significant decline in the area considered viable for western white pine, Pacific silver fir, mountain hemlock, subalpine fir, lodgepole pine, Engelmann spruce, and whitebark pine by as early as 2030; viable habitat is predicted to be entirely or nearly (less than 1 percent) absent from the assessment area by the end of the century.

Based on these collective modeling results, most subalpine forests in the assessment area will be outside their historical and contemporary climate envelope by the end of the century. Projected changes in the distribution of subalpine forests

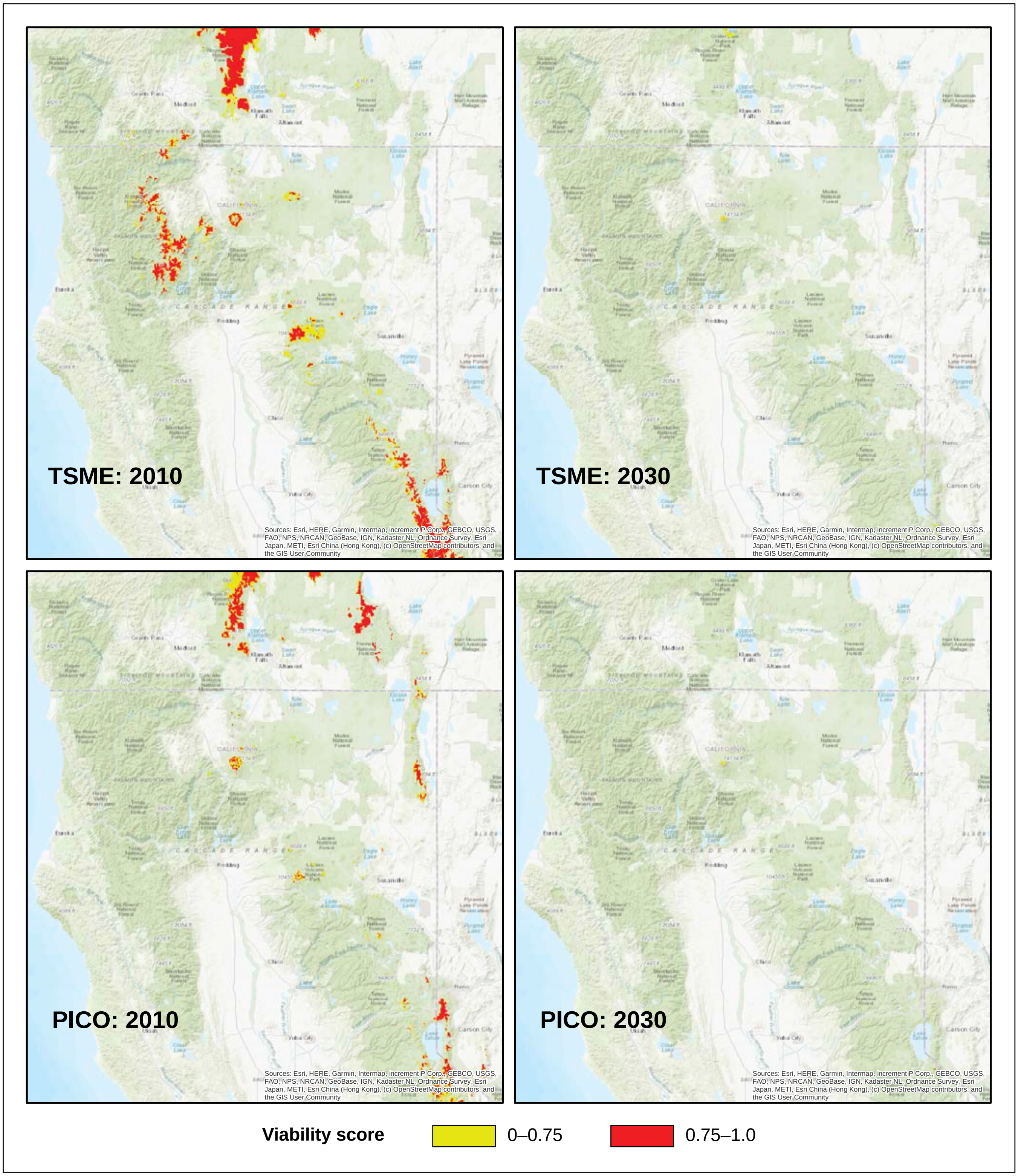

Figure 5.2—Examination of the data from Crookston et al. (2010) in Data Basin, a science-based mapping and analysis platform (https://databasin.org/), suggests a significant decline in the area considered viable for mountain hemlock (TSME) and whitebark pine (PICO), as well as other subalpine species, by as early as 2030; viable habitat is predicted to be entirely or nearly (less than 1 percent) absent from the assessment area by 2090. Viability scores (scale of 0 to 1.0) were derived using the Geophysical Fluid Dynamics Laboratory—Princeton University, National Aeronautics and Space Administration (GFDLCM21) general circulation model and A2 emission scenario (high emissions, regionally diverse world, rapid growth) to model future climate. Viability scores close to zero indicate a low suitability, whereas those near 1.0 indicate a suitability so high that the species is nearly always present in that climate. Red and yellow polygons on the map show areas of predicted high to moderate suitability for subalpine forests in the future.

consistently show a pronounced reduction in their current geographic extent within the assessment area by 2070–2100. Several models also project a relatively high degree of climate vulnerability for subalpine forests in the assessment area. These projections support theoretical models that predict greater loss of populations at geographic range margins and low-latitude limits (Hampe and Petit 2005).

Ultimately, the degree of climate vulnerability and adaptive capacity in these forest types will be contingent on several factors not covered by most species distribution models, including dispersal rates, biotic interactions, evolutionary processes (e.g., adaptation, genetic drift), physiological tolerances, edaphic constraints, interacting stressors, and forest management actions (Clark et al. 2011, Rowland et al. 2011). Generalized projections often suggest that subalpine taxa will shift continuously upslope until they run out of habitat at mountain summits; however, others have suggested that the heterogeneous terrain within subalpine ecosystems may also provide escape avenues other than upward, providing areas (e.g., with cooler and wetter microclimates) of possible refugia under future climatic conditions (Dobrowski et al. 2011, Millar and Rundel 2016). Consequently, subalpine forests may persist in or adapt to areas of high climate exposure despite ominous model projections for these forest types in the assessment area.

Red Fir and Subalpine Forests Summary

Based on our assessment of historical observations and datasets, as well as studies conducted in contemporary reference landscapes, the following general conclusions can be made about the natural range of variation (NRV) for red fir and subalpine forests in northwestern California and southwestern Oregon:

- Comparisons between historical and current conditions in the assessment area suggest that many of the ecological attributes of red fir and subalpine forests are currently within the NRV (tables 5.3 and 5.4); however, some conditions and processes are also approaching or are projected to be outside of the estimated limits of NRV in the near future.
- In contemporary red fir forests, the absence of fire for much of the 20th century has substantially lengthened the fire return interval (FRI) and fire rotation period to values outside of the NRV. In contrast, departure from the natural fire regime is less apparent in subalpine forests of the assessment area, which are largely within NRV primarily because of longer presettlement FRIs and fewer missed fire cycles.
- Future climate and wildfire projections indicate an extension of the fire season and an increase in fire size and severity in both red fir and subalpine forest types.

- Recent increases in native insect- and disease-related tree mortality in red fir and subalpine forests suggest that the scale and intensity of infestations may be increasing to levels outside of NRV in the near future. These changes are most likely driven by changing climatic factors, such as warming temperatures, decreased snowpack, and extended drought, as well as increases in stand density and changing successional patterns owing to fire suppression.
- Red fir and subalpine forests have both experienced an increase in tree densities, particularly in the smaller size classes, and concurrent decreases in the density of large-diameter trees. These alterations have likely resulted from logging, long periods of fire exclusion, and changes in climate.
- In red fir forests, simplification of forest structure has occurred at both the stand and landscape scales, with shifts away from a heterogeneous, partially open-canopy structure to one characterized by more continuous closed-canopy conditions.
- Climate envelope models consistently project a substantial loss in the area currently occupied by red fir and subalpine forests in the assessment area by the end of the 21st century. This suggests that the greatest changes to these high-elevation forest types in the coming decades will occur as a direct consequence of climate change and its indirect impact on disturbance intensity and frequency.

Table 5.3—Deviations from the natural range of variation (NRV) based on comparisons between historical/contemporary reference sites and current red fir forests in the assessment area[a]

Variable(s)	Historical reference period	Modern reference site	Currently within NRV	Confidence	Current trend (including future trends)	Notes
Fire return interval	1561-1938	No	No	High	Currently longer than it was historically	Low to moderate departure from NRV; future projections may be outside NRV
Fire rotation	1735–1905	No	No	Moderate	Currently longer than it was historically	Low to moderate departure from NRV; future projections may be outside NRV
Fire seasonality	1561–1938	No	Yes	Moderate	Marginally lengthening	Future climate projections suggest that more fires may occur later and earlier in the season
Fire size	1561–1938	Yes	Yes	Moderate	Marginally increasing	Recent trends suggest that fire size may soon exceed NRV
Fire severity	1735–1899	Yes	Yes/no	Moderate	Marginally increasing	Recent trends suggest increases in fire severity; future projections indicate that this trend is likely to continue
High-severity fire patch size	None	Yes	Yes	Low	Marginally increasing	Recent trends suggest marginal increase in the patch size of high-severity fire effects
Insects and pathogens	Mid-1800s–1950	Yes	Yes/no	Low	Increasing incidence of insects and pathogens	Generally within NRV, but recent trends suggest that values may soon exceed NRV
Wind	1874–1960	No	Yes	Low	—	
Avalanche	None	No	Yes	Low	—	
Tree density	1825–1920	Yes	No	High	Currently higher than expected under NRV	Increase in tree density, particularly in the smallest size classes
Average tree size	1870–1920	Yes	No	Moderate	Currently decreasing with respect to NRV	Increase in small-diameter trees and decline in large-diameter trees
Tree size class distribution	1825–1936	No	No	High	Current shifts to smaller size classes evident in many stands	
Basal area	1825–1920	Yes	Yes	Moderate	—	
Canopy cover	None	Yes	Yes/no	Moderate	Currently higher than expected under NRV	May be increasing with fire exclusion, especially in lower canopy strata; however, may also be decreasing in landscapes with increasing proportion of high-severity fire
Tree regeneration	None	Yes	Yes	Moderate	—	

Table 5.3—Deviations from the natural range of variation (NRV) based on comparisons between historical/contemporary reference sites and current red fir forests in the assessment area[a] (continued)

Variable(s)	Historical reference period	Modern reference site	Currently within NRV	Confidence	Current trend (including future trends)	Notes
Understory cover	None	Yes	No	Low	Currently lower than expected under NRV	Early-seral shrubs and herbaceous species may increase in the future in response to increased scale and severity of disturbance events (e.g., fire, insect outbreaks, etc.)
Structural complexity (horizontal and vertical)	None	Yes	No	Low	Decrease in fine-scale complexity within current stands	Decreasing proportion of canopy gaps and shifts to shorter (and denser) and multistory structural classes in fire-excluded stands; future increases in high-severity patch size could increase the proportion of sparse, open stands.
Seral class proportions	1600–1850	No	No	Low	Decreasing proportion of open canopy late-seral class in contemporary stands	Based on LANDFIRE Biophysical Settings Modeling for the Northwestern Bioregion
Overstory species composition	1825–1920	Yes	Yes	Moderate	—	
Understory species composition	None	Yes	Yes	Low	—	Difficult to assess owing to very little presettlement data. Based on relative abundance of shrub and herbaceous plant species in contemporary reference sites.

— = not enough information available to determine trend.

[a] The historical reference period was estimated based on the dates provided in the studies referenced in the text. The availability and use of modern reference sites to determine NRV is indicated; these were located within or outside of (but comparable to) the assessment area.

Table 5.4—Deviations from the natural range of variation (NRV) based on historical and modern reference information in subalpine forests of the assessment area[a]

Variable(s)	Historical reference period	Modern reference site	Currently within NRV	Confidence	Current trend (including future trends)	Notes
Fire return interval	1518–1985	No	Yes	Moderate	Marginal shortening	Currently within NRV; future climate projections may result in shorter fire return intervals (outside of NRV)
Fire rotation	Prior to 1933	No	Yes	Moderate	Marginal shortening	Currently within NRV; future climate projections may result in shorter fire rotations (outside of NRV)
Fire seasonality	1568–1920	No	Yes	High	Marginal lengthening	Future projected climate conditions may result in longer fire seasons
Fire size	1525–1925	Yes	Yes	Moderate		
Fire severity	Pre-1900	Yes	Yes	Low	Marginal increase	Future projections suggest an increase in high-severity fire effects
Native insects and pathogens	1895–1966	Yes	Yes	Moderate	Future increase in mortality rates associated with insects and pathogens	Generally within NRV, but approaching values that may soon exceed NRV; current period excludes recent outbreaks and nonnative pathogens (e.g., white pine blister rust)
Wind and avalanche	None	Yes	Yes	Low	—	
Tree density	1841–1930s	Yes	No	Moderate	Increasing	The magnitude of departure will be variable depending on elevation, topography, species composition, etc.
Average tree size	1870–1966	Yes	No	Low	Decreasing	
Tree size class distribution	1825–1936	Yes	No	Low	Shifting to smaller size classes	

Table 5.4—Deviations from the natural range of variation (NRV) based on historical and modern reference information in subalpine forests of the assessment area[a] (continued)

Variable(s)	Historical reference period	Modern reference site	Currently within NRV	Confidence	Current trend (including future trends)	Notes
Basal area	1600–1870	No	Yes	Moderate	—	
Canopy cover	None	Yes	Yes	Moderate		May be outside of NRV in areas that have experienced increased tree density
Tree regeneration	1841–1945	Yes	Yes	Low	—	General trend of increasing recruitment, particularly in higher elevations that are at or near tree line
Understory cover	None	Yes	Yes	Low	—	
Tree spatial patterns	1873–1930s	No	Yes	Moderate	—	Stands that were logged in the past may have lower structural diversity than expected under NRV
Seral class proportions	1932–1966	No	Yes	Low	—	The patch size of some mid-seral structural classes may be approaching or possibly exceeding the upper limit of NRV
Overstory species composition	1804–1936	Yes	Yes	Moderate	—	

— = not enough information available to determine trend.

[a] The historical reference period was estimated based on the dates provided in the studies referenced in the text. The availability and use of modern reference sites to determine NRV is indicated; these were located within or outside of (but comparable to) the assessment area.

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U.S. Equivalents

When you have:	Multiply by:	To get:
Millimeters (mm)	0.0394	Inches
Centimeters (cm)	2.54	Inches
Meters (m)	3.28	Feet
Kilometers (km)	.621	Miles
Hectares (ha)	2.47	Acres
Square kilometers (km^2)	0.386	Square miles
Square meters per hectare (m^2/ha)	4.37	Square feet per acre
Degrees Celsius (°C)	1.8 (°C + 32)	Degrees Fahrenheit

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Appendix 1: Supplemental Tables

Table A1.1—Historical fire return interval (FRI) estimates for red fir forests in the assessment area listed by mountain range and aggregated by forest type[a]

Vegetation type	Subregion	Mean FRI	Median FRI	Min. FRI	Max. FRI	Years sampled	Sample type[b]	Reference
Red fir	California	40	33	15	130	—	—	Van de Water and Safford (2011)
Red fir–white fir	Southern Cascades	28	—	—	—	1700–1900	Single tree	Sensenig (2002)
Red fir–white fir	Southern Cascades	11	—	2	66	1700–1900	Composite	Sensenig (2002)
Red fir–white fir	Southern Cascades	23	9.5	3	37	1749–1849	Composite	Bekker and Taylor (2001)
Red fir–white fir	Southern Cascades	—	24	4	55	1694–1899	Single tree	Bekker and Taylor (2001)
Red fir–white fir	Southern Cascades	10	8	—	—	1650–1918	Composite	Bekker and Taylor (2010)
Red fir–white fir	Southern Cascades	27	17	—	—	1650–1918	Single tree	Bekker and Taylor (2010)
Red fir–white fir	Southern Cascades	41	—	—	—	1735–1874	Composite	Taylor and Solem (2001)
Red fir–white fir	Southern Cascades	21.4	—	—	—	1740–1850	Composite	Taylor (1993)
Red fir–white fir	Southern Cascades	41	—	5	65	1830–1930	Composite	Taylor and Halpern (1991)
Red fir	Southern Cascades	—	20	8	35	1748–1846	Composite	McNeil and Zobel (1980), Skinner and Chang (1996)
Red fir	Southern Cascades	39	—	15	71	1628–1902	Composite	Chappell and Agee (1996)
Red fir	Southern Cascades	71	—	33	175	1629–1850	Composite	Foster (1998)
Red fir	Southern Cascades	63	—	—	—	1700–1900	Single tree	Sensenig (2002)
Red fir	Southern Cascades	25	—	6	89	1700–1900	Composite	Sensenig (2002)
Red fir–western white pine	Southern Cascades	74	70	26	109	1685–1937	Single tree	Taylor (2000)

Table A1.1—Historical fire return interval (FRI) estimates for red fir forests in the assessment area listed by mountain range and aggregated by forest type[a] (continued)

Vegetation type	Subregion	Mean FRI	Median FRI	Min. FRI	Max. FRI	Years sampled	Sample type[b]	Reference
Red fir- western white pine	Southern Cascades	15.3	—	—	—	1751–1849	Composite	Taylor (2000)
Red fir–western white pine	Southern Cascades	66	—	—	—	1768–1874	Composite	Taylor and Solem (2001)
Red fir–mountain hemlock	Southern Cascades	—	20	9	91	1783–1849	Composite	Bekker and Taylor (2001)
Red fir–mountain hemlock	Southern Cascades	—	20	9	91	1783–1849	Composite	Bekker and Taylor (2001)
Red fir–mountain hemlock	Southern Cascades	100	100	—	—	1650–1918	Composite	Bekker and Taylor (2010)
Red fir–mountain hemlock	Southern Cascades	100	100	—	—	1650–1918	Single tree	Bekker and Taylor (2010)
Red fir	Klamath Mountains	40	—	—	—	—	Stand age	Atzet and Martin (1992)
Red fir–mixed conifer	Klamath Mountains	75.5	—	7	194	—	Single tree	Skinner (2003)
Red fir–mixed conifer	Klamath Mountains	23.1	13	3	123	1561–1908	Composite	Skinner (2003)
Red fir–mixed conifer	Klamath Mountains	75.4	—	12.5	256	—	Single tree	Skinner (2003)
Red fir–mixed conifer	Klamath Mountains	58.7	39.5	5	172	1568–1920	Composite	Skinner (2003), Whitlock et al. (2003)
Red fir–mixed conifer	Klamath Mountains	25.4	—	6	52	—	Single tree	Skinner (2003)
Red fir–mixed conifer	Klamath Mountains	18.3	14	3	52	1693–1894	Composite	Skinner (2003)
Red fir–mixed conifer	Klamath Mountains	44.5	—	5.8	88	—	Single tree	Skinner (2003)
Red fir–mixed conifer	Klamath Mountains	23.4	15	4	72	1587–1938	Composite	Skinner (2003)

= No estimate available

[a] Summary values for red fir forest types based on elevation (low, mid, high) and geographic location are provided in table 3.1.

[b] Refers to whether estimates were derived from a single tree or a composite of samples (fires scarring two or more trees) or from analysis of stand age.

Table A1.2—Proportion of fire severity classes in red fir forests of the assessment area based on historical and contemporary reference site information[a]

Forest type	Location	Fire severity			Reference
		Low	Moderate	High	
Contemporary reference sites and stand reconstructions:					
Red fir	Southern Cascades (Crater Lake National Park [NP]): Crater Peak Fire	29	32	34	Chappell (1991), Chappell and Agee (1996)
Red fir	Southern Cascades (Crater Lake NP): Desert Cone Fire	20	69	7	Chappell (1991), Chappell and Agee (1996)
Red fir	Southern Cascades (Crater Lake NP): Castle Point Fire	66	28	7	Chappell and Agee (1996)
Red fir–white fir	Southern Cascades (Caribou Wilderness)	13	21	33	Bekker and Taylor (2001)
Red fir–mountain hemlock	Southern Cascades (Caribou Wilderness)	15	30	28	Bekker and Taylor (2001)
Red fir–white fir	Southern Cascades (Caribou Wilderness)	43	44	13	Taylor and Solem (2001)
Red fir–western white pine	Southern Cascades (Caribou Wilderness)	33	48	19	Taylor and Solem (2001)
LANDFIRE Biophysical Setting Model[b]					
Red fir	Southern Cascades	58	19	23	LANDFIRE (2017)
Red fir	North Coast Range of Oregon and northern California, Cascade Range	52	29	19	LANDFIRE (2017)
Red fir	Eastern Cascades of Oregon	45	41	13	Haugo et al. (2019)
Red fir	Klamath Mountains of Oregon	42	45	13	Haugo et al. (2019)
Contemporary studies:					
Red fir	eastern Cascades of Oregon	6	41	53	Haugo et al. (2019)
Red fir	Klamath Mountains of Oregon	20	42	38	Haugo et al. (2019)
Fir–high-elevation conifer	Four National Forests in the Northwestern Bioregion (California only)			19	Miller et al. (2012b)[c]

[a] Summary fire severity proportions for aggregated red fir forest types are provided in table 3.3.

[b] Based on LANDFIRE Biophysical Setting Model estimates of historical reference conditions.

[c] Percentage calculated for long-unburned forests (i.e., those that burned between 1987 and 2008, but had no previously recorded fire since 1910).

Table A1.3—Average total and relative red fir tree densities and basal area within historical (pre-1940) stands, contemporary reference stands in active fire regime landscapes, contemporary unlogged sites in fire-excluded landscapes, and contemporary logged and unlogged stands that were measured as part of the U.S. Forest Service (USFS) Forest Inventory and Analysis (FIA) program

Subregion	Tree density			Basal area			No. of stands	Min d.b.h.[a]	Forest type[b]	Fire history[c]	Reference
	Total	Red fir	Red fir	Total	Red fir	Red fir					
	- - - *Trees/ha* - - -		*Percentage*	- - - *m²/ha* - - -		*Percentage*		*cm*			
Historical (pre-1940) estimates:											
California	318	310	97	—	—	—	6		RF	—	Schumacher (1928)[d]
Southern Cascades	—	—	62	—	—	—	6	>10	RF	—	Leiberg (1900)
Southern Cascades	36	—	—	11	—	—	1	>20	RF	—	Dubrasich (2010)
Southern Cascades	372	306	83	—	—	—	13	>10	RF		Kelly et al. (2005)
Klamath Mountains	262	177	62	—	—	—	13	>10	RF		Kelly et al. (2005)
Contemporary unlogged reference stands in active fire regime landscapes:											
Southern Cascades	725	564	78	85	70	82	2	>5	RF-WF	<1.5 psFRI	Taylor and Halpern (1991)
Southern Cascades	653	635	97	85	84	98	48		RF	<1 psFRI	Chappell (1991)[e]
Southern Cascades	191	138	68	58	43	75	11	>5	RF	<1 psFRI	USDA FS (n.d.)[f]
Klamath Mountains	388	232	61	46	33	69	11	>5	RF	<1 psFRI	USDA FS (n.d.)[f]
Southern Cascades	624	282	45	—	—	—	2	>10	RF	—	McNeil and Zobel (1980)
Contemporary unlogged reference stands in fire-excluded landscapes:											
Southern Cascades	1,285	740	58	69	38	55	24	>4	RF-WF	—	Bekker and Taylor (2001)
Southern Cascades	1,319	625	47	110	54	49	16	>4	RF-WF	—	Taylor and Solem (2001)
southern Cascades	1,225	580	47	97	36	37	5	>5	RF-WF	>2 psFRI	Beaty and Taylor (2001)
Southern Cascades	481	—	—	53	16	—	9	>10	RF-WF	—	Parker (1991)
Southern Cascades	—	1110	—	—	90	—	4	—	RF-WWP-WF	>2psFRI	Cheng (2004); Keeler-Wolf (1990)
Southern Cascades	1,810	1,497	83	61	55	91	20	—	RF	>2psFRI	Chappell (1991)[g]
Southern Cascades	604	409	74	79	53	71	4	>5	RF	>1.5psFRI	USDA FS (n.d.)[f]
Southern Cascades	644	497	77	17	14	82	28	>5	RF	—	Pierce et al. (2012)
Southern Cascades	986	536	54	136	105	77	22	>10	RF	—	Forrestel (2013
Southern Cascades	190	103	54	50	27	76	1	>20	RF	>2 psFRI	Dubrasich (2010)
Southern Cascades	868	736	85	81	64	79	35	>4	RF-WWP	—	Taylor (2000)
Southern Cascades	1,404	1,088	77	106	74	70	31	>4	RF-WWP	—	Taylor and Solem (2001)

Table A1.3—Average total and relative red fir tree densities and basal area within historical (pre-1940) stands, contemporary reference stands in active fire regime landscapes, contemporary unlogged sites in fire-excluded landscapes, and contemporary logged and unlogged stands that were measured as part of the U.S. Forest Service (USFS) Forest Inventory and Analysis (FIA) program (continued)

Subregion	Tree density			Basal area			No. of stands	Min d.b.h.[a]	Forest type[b]	Fire history[c]	Reference
	Total	Red fir	Red fir	Total	Red fir	Red fir					
	- - - *Trees/ha* - - -		*Percentage*	- - - *m²/ha* - - -		*Percentage*		*cm*			
Southern Cascades	433	—	—	66	47	—	16	>10	RF-WWP	—	Parker (1991)
Southern Cascades	1,190	613	52	62	25	40	24	>4	RF-MH	—	Bekker and Taylor (2001)
Southern Cascades	709	313	44	94	44	47	35	>4	RF-MH	—	Taylor (1990)
Klamath Mountains	1,883	1,469	78	69	—	—	35	>5	RF-WF	—	Regalia (1978)
Klamath Mountains	810	339	54	58	43	66	6	>5	RF	>1.5 psFRI	USDA FS (n.d.)[f]
Klamath Mountains	—	—	—	56	—	—	7	—	RF	>1.5 psFRI	Cheng (2004); Keeler-Wolf (1990)
Klamath Mountains	643	641	100	—	—	—	35	>5	RF	>1.5 psFRI	Regalia (1978)
North Coast Mountains	—	248	90.5	—	58	98	2	>3	RF	—	Barbour and Woodward (1985)
North Coast Mountains	—	360	100	—	68	100	3	>3	RF	—	Barbour and Woodward (1985)
North Coast Mountains	—	384	88	—	75	95	3	>3	RF	—	Barbour and Woodward (1985)
North Coast Mountains	—	496	89	—	47	98	2	>3	RF	—	Barbour and Woodward (1985)
North Coast Mountains	—	369	91	—	42	94	5	>3	RF	—	Barbour and Woodward (1985)
North Coast Mountains	—	303	90	—	45	93	1	>3	RF	—	Barbour and Woodward (1985)
Contemporary logged and unlogged stands (FIA plots 2001–2014):											
Assessment area	288	139	48	—	—	—	98	>5	RF-WF	—	USFS Forest Inventory and Analysis (2017)

— = no estimate/information available.

[a] Minimum diameter at breast height (d.b.h.) specifies the minimum tree size used in the density and basal area calculations.

[b] Forest types include red fir–white fir (RF-WF); red fir–western white pine-white fir (RF-WWP-WF); red fir (RF); red fir–western white pine (RF-WWP); and red fir–mountain hemlock (RF-MH).

[c] Fire history was estimated from individual studies and compared to the mean fire return interval (FRI) for mid-elevation red fir (presettlement fire return interval [psFRI] = 43 years; presented in table 3.1). Summary values are provided in table 3.4 and for comparison, used only those studies where the minimum d.b.h. values were between 3 and 5 cm d.b.h.

[d] Estimates are based on the oldest (≥160 years) red fir stands with a diameter distribution that most closely approximated presettlement conditions.

[e] Estimates include all trees and saplings >1.4 m height; data are for the Crater Peak site in Crater Lake National Park, which had at least four fires within the 100 years prior to measurement (within 1 FRI).

[f] USDA FS (n.d.) includes unlogged red fir stands from Lassen Volcanic National Park and the Marble Mountain Wilderness, respectively, that are located in active fire regime landscapes (i.e., burned; no departure from historical FRI) and fire-excluded stands (i.e., unburned; missed two or more historical FRIs).

[g] Estimates include all trees and saplings >1.4 m height; data are for the Desert Cone site where no obvious evidence of fire was evident for at least 190 years prior.

Table A1.4—Historical Fire Return Interval (FRI) estimates for subalpine forests in the assessment area listed by mountain range and aggregated by forest type

Vegetation type	Subregion	Mean FRI	Median FRI	Min. FRI	Max. FRI	Years sampled	Sample type[a]	Reference
		Years						
Subalpine forest	State of California	133	132	100	420	—	—	Van de Water and Safford (2011)
Western white pine	State of California	50	42	15	370	—	—	Van de Water and Safford (2011)
Lodgepole pine	Southern Cascades	67	—	—	—	1735–1929	Composite	Taylor and Solem (2001)
Pacific silver fir	Northern Cascades (WA)[b]	192	—	—	—	1573–1985	Composite	Agee et al. (1990)
Pacific silver fir	Central Cascades (OR)	138	—	—	—	—	Stand age	Simon (1991)
Subalpine fir	Northern Cascades (WA)[b]	154	—	—	—	1573–1985	Composite	Agee et al. (1990)
Mountain hemlock	Northern Cascades (WA)[b]	137	—	—	—	1573–1985	Composite	Agee et al. (1990)
Mountain hemlock–subalpine fir	Central Cascades (OR)	113	—	—	—	—	Stand age	Simon (1991)
Mountain hemlock	Central Cascades (OR)	168	—	—	—	—	Stand age	Simon (1991)
Mountain hemlock	Southern Cascades	143	—	44	192	1629–1850	Composite	Foster (1998)
Mountain hemlock	Klamath Mountains	115	—	—	—	—	Stand age	Atzet and Martin (1992)
Mountain hemlock	Klamath Mountains	11.5	—	—	—	1544 (first fire)	Stand-age	White et al. (1997)
Mountain hemlock	Klamath Mountains	20.3	—	—	—	1734 (first fire)	Stand age	White et al. (1997)
Mountain hemlock	Klamath Mountains	36.2	—	—	—	1522 (first fire)	Stand age	White et al. (1997)
Mountain hemlock[c]	Klamath Mountains	75.4	—	12.5	256	—	Single tree	Skinner (2003)
Mountain hemlock[c]	Klamath Mountains	39.5	58.7	5	172	1568–1920	Composite	Skinner (2003), Whitlock et al. (2003)
Whitebark pine–Lodgepole pine	Central Cascades (OR and WA)	73	—	12	314	1800–1950	Composite	Siderius and Murray (2005)
Whitebark pine–Engelmann spruce	Central Cascades (OR and WA)	72	—	9	196	1800–1950	Composite	Siderius and Murray (2005)
Whitebark pine	Central Cascades (OR and WA)	96	—	38	153	1800–1950	Composite	Siderius and Murray (2005)

— = no estimate available; WA = Washington; OR = Oregon.

[a] Refers to whether estimates were derived from a single tree or a composite of samples (fires scarring two or more trees) or from analysis of stand age.

[b] Not included in average FRI presented in table 4.2 because study site is outside of assessment area.

[c] Includes Shasta red-fir and whitebark pine.

Table A1.5—Summary of studies reporting average total tree densities and basal area within subalpine forests that are representative of contemporary unlogged reference sites (C-UL) and contemporary logged and unlogged stands (C) that include U.S. Forest Service (USFS) Forest Inventory and Analysis (FIA) data (collected 2001 to 2014)[a]

Forest type	Location[b]	Tree density	Basal area	No. of stands	Type	Reference
		Trees/ha	*m²/ha*			
Contemporary unlogged reference sites:						
Noble fir	Siskiyou—KM (CA and OR)	663	—	1	C (UL)	Whittaker (1960)[c]
Noble fir	Siskiyou—KM (CA and OR)	748	—	1	C (UL)	Whittaker (1960)[c]
Mountain hemlock	Mount Theilsen Wilderness—SC (OR)	—	52.2	20	C (UL)	Goheen (2002)[d]
Mountain hemlock	Crater Lake National Park—SC (OR)	591	87.3	16	C (UL)	Forrestel (2013)[e]
Mountain hemlock	Lassen Volcanic National Park—SC (CA)	483	17.6	12	C (UL)	Pierce et al. (2012)[f]
Mountain hemlock	Lassen Volcanic National Park—SC (CA)	1,992	—	24	C (UL)	Taylor (1995)[f]
Mountain hemlock	Lassen Volcanic National Park—SC (CA)	298	61	7	C (UL)	Parker (1991)[e]
Mountain hemlock-whitebark	Lassen Volcanic National Park—SC (CA)	843	—	10	C (UL)	Smith and Chung-MacCoubrey (2016)[g]
Mountain hemlock-whitebark	Crater Lake National Park—SC (OR)	415	—	10	C (UL)	Smith and Chung-MacCoubrey (2016)[g]
Western white pine	Mount Eddy RNA—KM (CA)	109	31.8	38	C (UL)	Cheng (2004); Keeler-Wolf (1990)
Foxtail pine	Crater Creek Candidate RNA—KM (CA)	1,150	80	10	C (UL)	Cheng (2004); Keeler-Wolf (1990)
Foxtail pine	Mount Eddy RNA— KM (CA)	108	42.5	38	C (UL)	Cheng (2004); Keeler-Wolf (1990)
Foxtail pine	Yolla Bolly Mountains—NCR(CA)	125	23	3	C (UL)	Eckert and Sawyer (2002)[g]
Foxtail pine	Trinity Alps—KM (CA)	177	30.6	5	C (UL)	Eckert and Sawyer (2002)[g]
Foxtail pine	Trinity Mountains—KM (CA)	233	24.8	4	C (UL)	Eckert and Sawyer (2002)[g]
Foxtail pine	Russian Peak—KM (CA)	315	24	1	C (UL)	Eckert and Sawyer (2002)[g]
Foxtail pine	Marble Mountains—KM (CA)	185	17.5	2	C (UL)	Eckert and Sawyer (2002)[g]
Whitebark pine	Crater Lake National Park—SC (OR)	123	21	20[f]	C (UL)	Smith and Sarr (2014)[g]

Table A1.5—Summary of studies reporting average total tree densities and basal area within subalpine forests that are representative of contemporary unlogged reference sites (C-UL) and contemporary logged and unlogged stands (C) that include U.S. Forest Service (USFS) Forest Inventory and Analysis (FIA) data (collected 2001 to 2014)[a] (continued)

Forest type	Location[b]	Tree density	Basal area	No. of stands	Type	Reference
		Trees/ha	*m^2/ha*			
Contemporary logged and unlogged stands:						
White fir	Assessment area (dry)	320	12.0	6	C (FIA)	USFS Forest Inventory and Analysis (2017)[f]
Lodgepole pine	Assessment area (wet)	378	14.0	31	C (FIA)	USFS Forest Inventory and Analysis (2017)[f]
Lodgepole pine	Assessment area (all)	368	14.0	37	C (FIA)	USFS Forest Inventory and Analysis (2017)[f]
Mountain hemlock-whitebark	Red Butte–Red Fir Ridge RNA—SC (CA)	—	25.2	1	C	Cheng (2004)
Whitebark pine	Red Butte–Red Fir Ridge RNA—SC (CA)	—	16.1	1	C	Cheng (2004)

— = No estimate available

[a] FIA data represent an unbiased sample of current conditions in subalpine forests. Therefore, inclusion of plots in managed (e.g., logged, burned, etc.) and unmanaged stands may have resulted in lower relative density and basal area estimates.

[b] Subalpine forest types are presented in general order of increasing elevation; however, it is important to note that there is considerable overlap between forest types (e.g., lodgepole pine, white fir). Abbreviations: RNA = research natural area, KM = Klamath Mountains, NCR = north Coast Range, SC = southern Cascades, CA = California, OR = Oregon.

[c] Measured trees >1 cm diameter at breast height (d.b.h.).

[d] Tree density is based on trees >12.7 cm d.b.h.

[e] Tree density is based on trees >10 cm d.b.h.

[f] Tree density estimates were based on trees >5 cm d.b.h.

[g] Includes all trees >1.37 m height.

[h] Includes the 10 whitebark pine plots reported by Smith and Chung-MacCoubrey (2016).

Table A1.6—Model projections describing future changes in the distribution of subalpine forests and tree species within the assessment area

Unit of analysis	Geographic scope	GCM (model type)[a]	Emissions scenario	Decrease[b]	Stable[b]	Time period	Reference
				- - - - *Percentage* - - - -			
Lodgepole pine	California	Ensemble	A2	58	34	2040–2065	SSP (2010)
Mountain hemlock	California	Ensemble	A2	40	38	2040–2065	SSP (2010)
Subalpine	California	PCM (MC1)	A2	48	—	2071–2100	Lenihan et al. (2008)
Subalpine	California	GFDL (MC1)	B2	60	—	2071–2100	Lenihan et al. (2008)
Subalpine	California	GFDL (MC1)	A2	77	—	2071–2100	Lenihan et al. (2008)
Subalpine	California	PCM (MC1)	A1	87	—	2070–2099	Hayhoe et al. (2004)
Subalpine	California	PCM (MC1)	B1	75	—	2070–2099	Hayhoe et al. (2004)
Subalpine	California	Hadley CM3 (MC1)	A1	75	—	2070–2099	Hayhoe et al. (2004)
Subalpine	California	Hadley CM3 (MC1)	B1	50	—	2070–2099	Hayhoe et al. (2004)
Subalpine	California (NE/SN)	CNRM CM5 (BCM)	RCP 4.5	34	66	2070–2099	Thorne et al. (2016)[c]
Subalpine	California (NE/SN)	CNRM CM5 (BCM)	RCP 8.5	67	33	2070–2099	Thorne et al. (2016)[c]
Subalpine	California (NE/SN	MICROC ESM (BCM)	RCP 4.5	72	28	2070–2099	Thorne et al. (2016)[c]
Subalpine	California (NE/SN	MICROC ESM (BCM)	RCP 8.5	95	5	2070–2099	Thorne et al. (2016)[c]
Subalpine	California (NW)	CNRM CM5 (BCM)	RCP 4.5	60	40	2070–2099	Thorne et al. (2016)[c]
Subalpine	California (NW)	CNRM CM5 (BCM)	RCP 8.5	68	32	2070–2099	Thorne et al. (2016)[c]
Subalpine	California (NW)	MIROC ESM (BCM)	RCP 4.5	85	15	2070–2099	Thorne et al. (2016)[c]
Subalpine	California (NW)	MIROC ESM (BCM)	RCP 8.5	93	7	2070–2099	Thorne et al. (2016)[c]
Subalpine	central Oregon	Hadley CM3 (MC1)	A2	50	—	2010–2100	Halofsky et al. (2013)
Subalpine	central Oregon	MIROC 3.2 (MC1)	A2	50	—	2010–2100	Halofsky et al. (2013)
Subalpine	central Oregon	CSIRO Mk3 (MC1)	A2	50	—	2010–2100	Halofsky et al. (2013)
Whitebark pine	Entire species range	Hadley and CCMA (RF)	Unknown	97	3	2090	Warwell et al. (2007)

— = no estimate available. NW = northwest, SN = Sierra Nevada, NE = northeast.

[a] Acronym definitions: PCM (Parallel Climate Model); GFDL (Geophysical Fluid Dynamics Laboratory); MC1 (dynamic vegetation model); Hadley CM3 (Hadley Centre Coupled Model, version 3), CNRM (Centre National de Recherches Météorologiques), CCMA (Canadian General Circulation Model); BCM (Basin Characterization Model); RF (Random Forest).

[b] Unless otherwise noted, the percentage of decrease or stability represent the percentage of change in the area covered by subalpine forest types within the geographic scope and time period of each study.

[c] These values represent modeled percentage of unsuitable and suitable habitat.

Appendix 2: Common and Scientific Names

Common name	**Scientific name**
Trees:	
Alaska yellow-cedar	*Callitropsis nootkatensis* (D. Don) Oerst. ex D.P. Little
Brewer spruce	*Picea breweriana* S. Watson
California red fir	*Abies magnifica* A. Murray bisb var. *magnifica*
Douglas-fir	*Pseudotsuga menziesii* (Mirb.) Franco var. *menziesii*
Engelmann spruce	*Picea engelmannii* Parry ex Engelm.
Foxtail pine	*Pinus balfouriana* Balf.
Grand fir	*Abies grandis* (Douglas ex D. Don) Lindl.
Great Basin bristlecone pine	*Pinus longaeva* D.K. Bailey
Incense cedar	*Calocedrus decurrens* (Torr.) Florin
Jeffrey pine	*Pinus jeffreyi* Balf.
Limber pine	*Pinus flexilis* James
Lodgepole pine	*Pinus contorta* Douglas ex Loudon
Mountain hemlock	*Tsuga mertensiana* (Bong.) Carrière
Noble fir	*Abies procera* Rehder
Oregon white oak	*Quercus garryana* Douglas ex Hook.
Pacific silver fir	*Abies amabilis* (Douglas ex Loudon) Douglas ex J. Forbes
Quaking aspen	*Populus tremuloides* Michx.
Red fir	*Abies magnifica* A. Murray bisb
Shasta red fir	*Abies magnifica* var. *shastensis* Lemmon
Subalpine fir	*Abies lasiocarpa* (Hook.) Nutt. var. *lasiocarpa*
Subalpine larch	*Larix lyallii* Parl.
Sugar pine	*Pinus lambertiana* Douglas
Western juniper	*Juniperus occidentalis* Hook.
Western white pine	*Pinus monticola* Douglas ex D. Don
White fir	*Abies concolor* (Gordon & Glend.) Lindl. ex Hildebr.
Whitebark pine	*Pinus albicaulis* Engelm.
Shrubs/subshrubs:	
Blackberry	*Rubus* L.
Bush chinquapin	*Chrysolepis sempervirens* (Kellogg) Hjelmqvist
Deer oak	*Quercus sadleriana* R. Br. Ter
Greenleaf manzanita	*Arctostaphylos patula* Greene
Huckleberry oak	*Quercus vacciniifolia* Kellogg
Devil's-club	*Oplopanax horridus* (Sm.) Miq.
Huckleberry	*Vaccinium* sp.
Mountain dryas	*Dryas drummondii* Richardson ex Hook.
One-sided wintergreen	*Orthilia secunda* (L.) House
Pinemat manzanita	*Arctostaphylos nevadensis* A. Gray
Tobacco brush	*Ceanothus velutinus* Douglas ex. Hook.
White veined wintergreen	*Pyrola picta* Sm.

Common name	Scientific name
Herbs:	
Canby’s licorice-root	*Ligusticum canbyi* (J.M. Coult. & Rose) J.M. Coult. & Rose
False hellebore	*Veratrum viride* Aiton
Red fir dwarf mistletoe	*Arceuthobium abietinum* Engelm. ex Munz f. sp. *magnificae* Hawksworth & Wiens
Sitka valerian	*Valeriana sitchensis* Bong.
Wiens’ dwarf mistletoe	*Arceuthobium abietinum* Engelm. ex Munz subsp. *wiensii* (Mathiasen & C. Daugherty)
Yarrow	*Achillea millefolium* L.
Lichen:	
wı´la; black tree lichen	*Bryoria fremontii* (Tuck.) Brodo & D. Hawksw.
Insects:	
Fir engraver beetle	*Scolytus ventralis*
Flatheaded fir borer	*Melanophila drummondii*
Roundheaded fir borer	*Tetropium abietis*
Mountain pine beetle	*Dendroctonus ponderosae*
Fungi and disease:	
Cytospora canker	*Cytospora abietis*
Yellow cap fungus	*Pholiota limonella*
Heterobasidion root disease	*Heterobasidion occidentale*
Laminated root rot	*Phellinus sulphurascens*
Armillaria root disease	*Armillaria ostoyae*
White pine blister rust	*Cronartium ribicola*
Schweinitzii root and butt rot	*Phaeolus schweinitzii*
Animals:	
Clark’s nutcracker	*Nucifraga columbiana*
Douglas’ squirrel	*Tamiasciurus douglasii*
Heather vole	*Phenacomys intermedius*
Pacific marten	*Martes caurina*
Snowshoe hare	*Lepus americanus*

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