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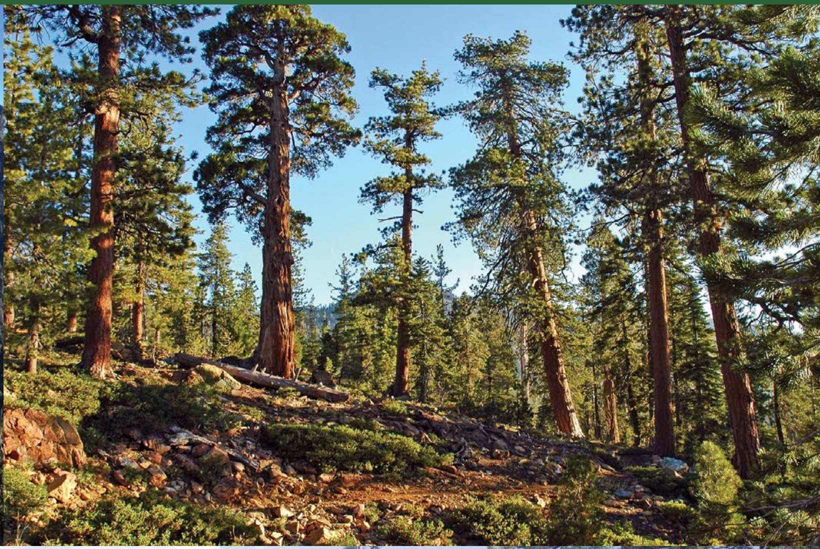
Natural Range of Variation for Yellow Pine and Mixed-Conifer Forests in Northwestern California and Southwestern Oregon

Gabrielle N. Bohlman, Hugh D. Safford, and Carl N. Skinner



Authors

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Cover: (top left) mixed-conifer forest on the Shasta-Trinity National Forest within the 2018 Hirz Fire footprint, photo by Gabrielle Bohlman, USDA Forest Service; (top right) upper montane incense cedar and Jeffrey pine forest on serpentine soils with huckleberry oak and pinemat manzanita in the understory; photo by Carl Skinner, USDA Forest Service; (bottom) mixed-conifer forest on the Rogue River-Siskiyou National Forest near the California-Oregon border; photo by Gabrielle Bohlman, USDA Forest Service.

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Abstract

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Yellow pine and mixed-conifer (YPMC) forests are common forest types in northwestern California and southwestern Oregon (the "assessment area"). YPMC forests occur above oak woodland and mixed-evergreen forests and below red fir forests, and are dominated by diverse combinations of ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson), Jeffrey pine (*P. jeffreyi* Balf.), incense cedar (*Calocedrus decurrens* (Torr.) Florin), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), and sugar pine (*P. lambertiana* Douglas) with a notable component of hardwood species, including a number of oaks (e.g., California black oak (*Quercus kelloggii* Newberry), canyon live oak (*Q. chrysolepis* Liebm.), Oregon white oak (*Q. garryana* Douglas ex Hook.), and various other species. We conducted an indepth assessment of the natural range of variation (NRV) of YPMC forests for the assessment area, focusing on ecosystem processes and forest structure from historical data sources from pre-Euro-American colonial settlement (presettlement) times (16th through mid-19th centuries) and current reference forests (YPMC forests that have retained frequent fire and have suffered little human degradation), and then compared current conditions to the NRV. The Mediterranean climate of the assessment area, modified by strong latitudinal and elevational gradients, combined with high geological and topographic complexity in the assessment area, strongly influence the distribution of forest types and the structure and composition of YPMC forests. Historically, fire was a key functional process in YPMC forests that helped maintain relatively open canopies, limit fuel accumulation, decompose biomass, recycle nutrients, promote the dominance of mostly shade-intolerant species, and create structural heterogeneity on the landscape. Forest structure in presettlement YPMC forests was highly variable at larger spatial scales, but was characterized by relatively low tree densities, large tree sizes, high seedling mortality as a result of recurrent fire, and highly heterogeneous understory structure. Following Euro-American colonial settlement, widespread changes occurred in YPMC forests in the assessment area, principally because of extensive logging accompanied by a century of highly effective, ubiquitously applied fire suppression. Modern YPMC forests have departed from NRV conditions for a wide range of ecosystem processes and structural attributes. There is strong consensus among published studies that, on average, modern YPMC stands have much higher densities dominated by smaller trees (often of shade-tolerant species) and much longer fire return intervals compared to reference YPMC forests. Additionally, fires that escape initial attack can be much larger and generally have larger proportions of high-severity areas than typical pre-Euro-American settlement fires. There is more moderate consensus among published studies that the average modern YPMC stand in the assessment area supports greater fuels and deeper forest litter, higher canopy cover and fewer canopy gaps, higher tree basal area, more coarse woody debris, a higher density of snags, lower grass and forb cover, less area burned across the landscape, and experiences a longer fire season compared to reference YPMC forests. Among the variables assessed, overall plant species richness and total percent cover of shrubs appear to be within or near the NRV.

Keywords: Yellow pine forests, mixed-conifer forest, ecosystem function, fire regime, forest structure, historical range of variation, HRV, natural range of variation, NRV, species diversity and composition, succession.

Preface

In 1976, President Gerald Ford signed the National Forest Management Act (NFMA), which—along with the Forest and Rangeland Renewable Resources Planning Act of 1974 (FRRRPA)—committed the Forest Service to developing and periodically updating land and resource management plans (LRMPs) at the national forest or national grassland level. The principal purpose of LRMPs is to provide for "multiple use" and "sustained yield" of natural resources in the National Forest System.

The NFMA and FRRRPA required the Forest Service to develop regulations to guide the LRMP revision process. These guidelines came to be known as the "planning rule," and were first published in 1982. Various inadequacies of the original rule became apparent over time, and multiple abortive efforts were made to modify or "modernize" it. In 2012, a new rule was finally adopted (36 CFR 219) (USDA FS 2012), and new forest plans began following the revised process in 2013.

According to 36 CFR 219.1(c), the purpose of the USDA Forest Service 2012 Planning Rule is "to guide the collaborative and science-based development, amendment, and revision of land management plans that promote the ecological integrity of national forests and grasslands…" The rule is focused on maintaining biological diversity on Forest Service units and ensuring the "integrity of the compositional, structural, and functional components comprising…ecosystems."

The 2012 planning rule places heavy emphasis on the concepts of "sustainability" and "ecological integrity." In the rule, sustainability is defined as "the capability of ecosystems to maintain ecological integrity," and ecological integrity is defined as follows:

> The quality or condition of an ecosystem when its dominant ecological characteristics (for example, composition, structure, function, connectivity, and species composition and diversity) occur within the natural range of variation and can withstand and recover from most perturbations imposed by natural environmental dynamics or human influence [36 CFR 219.19: 21271].

The definition of ecological integrity in the 2012 planning rule thus inherently requires the determination of the "natural range of variation" (NRV) for a suite of important ecosystem variables, organized into three main categories: composition, structure, and function. Physical connectivity derives from structure, and composition encompasses species connectivity, composition, and diversity. NRV is defined by the Forest Service Handbook 1909-12 in Chapter Zero Code:

> The variation of ecological characteristics and processes over scales of time and space that are appropriate for a given management application…

> NRV represents an explicit effort to incorporate a past perspective into management and conservation decisions.…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 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.

The period of colonial settlement by nonnative people primarily of European decent is often referred to as the ‘settlement’ period. This period took place mostly in the mid- and late-1800s in this region and was characterized by dramatic declines in native populations and disruption of cultures through spread of disease, dislocation, mistreatment, and state-sanctioned violence (Madley 2016).

Summary

The Northwestern California and Southwestern Oregon Assessment Area (assessment area) includes portions of the Klamath Mountains, the North Coast Range of California, and the southern Cascade Range of Oregon and California. The assessment area falls within the Mediterranean climate zone of North America and supports wet, cool winters and dry, warm summers. The area's topography and proximity to the Pacific Ocean create a notable level of geographic variation in climate. Mean annual precipitation ranges from more than 200 cm on northwestern coastal mountains to below 50 cm in interior valleys in the eastern portion of the assessment area. Yellow pine and mixed-conifer (YPMC) forests are an important component of this landscape; they occur throughout the low- and mid-montane zones in the assessment area with a higher prevalence across the eastern portion where maritime climatic influences are less apparent. Cold conditions tend to limit distributions of the dominant YPMC canopy species at upper elevations, and dry conditions limit distributions at lower elevations in the east. To the west, the YPMC pines are limited by wetter conditions that support less frequent fire and a component of shade-tolerant hardwoods and conifers that are strong resource competitors. Geology and topography in the assessment area are very diverse, with the Klamath Mountains being the most geologically complex portion of the assessment area. Geological and topographic complexity strongly influence the distribution of forest types as well as the structure and productivity of vegetation throughout the area.

The yellow pine component of YPMC forests in the assessment area is represented by ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) and Jeffrey pine (*P. jeffreyi* Balf.). Of the two, ponderosa pine is the most widespread geographically, occurring throughout western North America. Jeffrey pine is limited mostly to California, with some presence in westernmost Nevada, southwestern Oregon, and northern Baja California. Mixed-conifer forests in the assessment area support many tree species, but they are dominated by diverse combinations of ponderosa pine, Jeffrey pine, incense cedar (*Calocedrus decurrens* (Torr.) Florin), sugar pine (*P. lambertiana* Douglas), western white pine (*P. monticola* Douglas ex D. Don), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), and white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.) with a notable component of hardwood species, including California black oak (*Quercus kelloggii* Newberry), Oregon white oak (*Q. garryana* Douglas ex Hook.), canyon live oak (*Q. chrysolepis* Liebm.), Pacific madrone (*Arbutus menziesii* Pursh), golden chinquapin (*Chrysolepis chrysophylla* (Douglas ex Hook.) Hjelmqvist), tanoak (*Notholithocarpus densiflorus* (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh), and bigleaf maple (*Acer macrophyllum* Pursh).

Important ecological differences among the main canopy species in YPMC forests (e.g., tolerance of shade, fire, and drought) have influenced their distribution and abundance across climatic and topographic gradients which, in concert with frequent fires, have led to high levels of heterogeneity within this general forest type. Despite climatic fluctuations during the Holocene Epoch, the assessment area has been at least partially forested for 10,000 years, with an increase in the mixed-conifer component over the past 4,000 years when fire also has been very frequent, at least partially due to burning by American Indians. The past 150 years have seen extensive changes to assessment area YPMC forests, which experienced large-scale logging and nearly ubiquitous fire exclusion that have dramatically altered contemporary forest structure and ecological processes.

Fire is a key functional process in YPMC forests that helps maintain a relatively open canopy, limits fuel accumulation, decomposes biomass, recycles nutrients, promotes the dominance of mostly shade-intolerant and fire-tolerant species, and creates structural heterogeneity on the landscape. YPMC forests in the assessment area prior to Euro-American colonial settlement (pre-1850) mostly supported frequent (median fire return interval [FRI] <20 years), low- to moderate-severity fires. A low- to moderate-severity fire regime is dominated by fires with greater proportions of low-severity (<25 percent overstory mortality) and moderate-severity (25 to 75 percent overstory mortality) patches, with a limited number and size of high-severity (>75 percent overstory mortality) patches. However, larger high-severity patch sizes sometimes occurred, depending on burning conditions or topography.

Forest structure in presettlement YPMC forests was highly variable and generally characterized by fine-grained (within-stand [<1 ha]) heterogeneity driven by fire's interaction with geologic, topographic, and climatic features that influenced vegetation productivity and structure. In areas with a tendency toward more moderate-severity fires, this fine-grained heterogeneity sometimes gave way to a matrix of more coarse-grained (landscape) heterogeneity. Tree densities were relatively low, and in general, average tree sizes were large. Based on stand reconstructions and historical data from in and around the assessment area, YPMC stands ranged in mean density from about 14 to 314 trees per hectare, or tph, (~120-tph average) depending on edaphic conditions and the proportion of Douglas-fir and white fir in the stand. Portions of the presettlement landscape were characterized by coarse-grained gaps occupied by montane chaparral or resprouting hardwoods, often in areas that had burned one or more times at high severity.

Over the past 150 years, primarily as a result of logging, which removed the largest trees, and fire suppression, which densified the understory and caused openings to fill in with trees (including an expansion into adjacent nonforested areas), fine-grain heterogeneity across the landscape has decreased. Contemporary mean tree density of YPMC forests in the assessment area is 375 tph, with densities ranging from 98 to 916 tph in modern stands for which presettlement reconstructions exist. This increase in density is driven by small-diameter (<60 cm) trees and has caused substantial changes in overall forest structure. For example, average tree size is much smaller than it was in presettlement forests, and the overall distribution of tree size classes vs. density has shifted from a relatively flat curve (small changes in density between tree size classes) to a very steep curve dropping from high numbers of small trees to small numbers of large trees. There are many fewer fine-scale canopy gaps across the landscape, and canopy cover has become more continuous. These changes in stand structure have caused changes in ecosystem processes. For instance, insect and pathogen activity increases with stand density; accumulation of litter and duff has increased nutrient leaching to surface and ground water; and vulnerability to severe fire has increased.

The primary role of fire in YPMC forests has changed from one of forest maintenance (of relatively open-canopy, fuel-limited conditions with dominance primarily by fire-tolerant species) to one of forest transformation, where dense stands of fire-intolerant and fire-tolerant species and heavy fuel accumulations are more likely to burn at high severity, resulting in major ecosystem changes. This shift is the result of fire suppression policies put in place during the establishment of the national forest system in 1905. The mean proportion and patch size of high-severity fire in YPMC forests are generally greater in modern fires than they were prior to the fire suppression period, and as future climates continue to warm, the frequency of large fires with large high-severity patches are likely to increase. Overall fire frequency will also likely increase, but continue to remain much lower than historical frequencies in most areas.

Under presettlement conditions, spatially complex fires created a fine-grained forest matrix with variably sized openings that strongly influenced regeneration patterns. Modern seedling and sapling cohorts are often composed primarily of species that are more shade-tolerant and less fire-tolerant with higher densities than under presettlement conditions on average, despite being highly variable across the landscape. In the wake of large fires that have large stand-replacing patches, conifers struggle to reestablish because of a combination of long distances to viable seed sources and heavy competition from fire-stimulated shrubs. Overall shrub cover at the landscape scale is considered to be within the NRV, however, the pattern of shrub distribution on the landscape has likely been greatly altered. Many fire-suppressed areas have seen reductions in shrub cover as a result of increased shading, while many burned areas have experienced increases in shrub cover within large contiguous patches of high-severity fire. Herbaceous cover in undisturbed mixed-conifer forests is likely lower today than during presettlement times due to shading as well as cover of litter and dead woody material from substantial increases in tree densities.

Contents

Chapter 1: Introduction

Physical Setting

The Northwestern California and Southwestern Oregon Assessment Area (assessment area) includes portions of the Klamath Mountains, the North Coast Range of California, and the southern Cascade Range of Oregon and California (see map in fig. 1). Unlike the Sierra Nevada region, yellow pine and mixed-conifer (YPMC) forests do not make up the dominant forest type in the montane zone of the assessment area. Instead, the area is mostly dominated by Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and mixed-evergreen (multiple broadleaved species plus Douglas-fir) forests because of its proximity to the coast, higher average precipitation, and shorter dry season. YPMC forests are nonetheless an important component of the landscape, covering about 1.5 million ha (3.6 million ac) of the assessment area (fig. 1). These forests are present throughout the lower and mid montane zones in the assessment area, often dominating more xeric sites and areas with poorer soils, and are more prevalent across the eastern portion where maritime climatic influences are reduced (Sawyer 2006, Skinner et al. 2018).

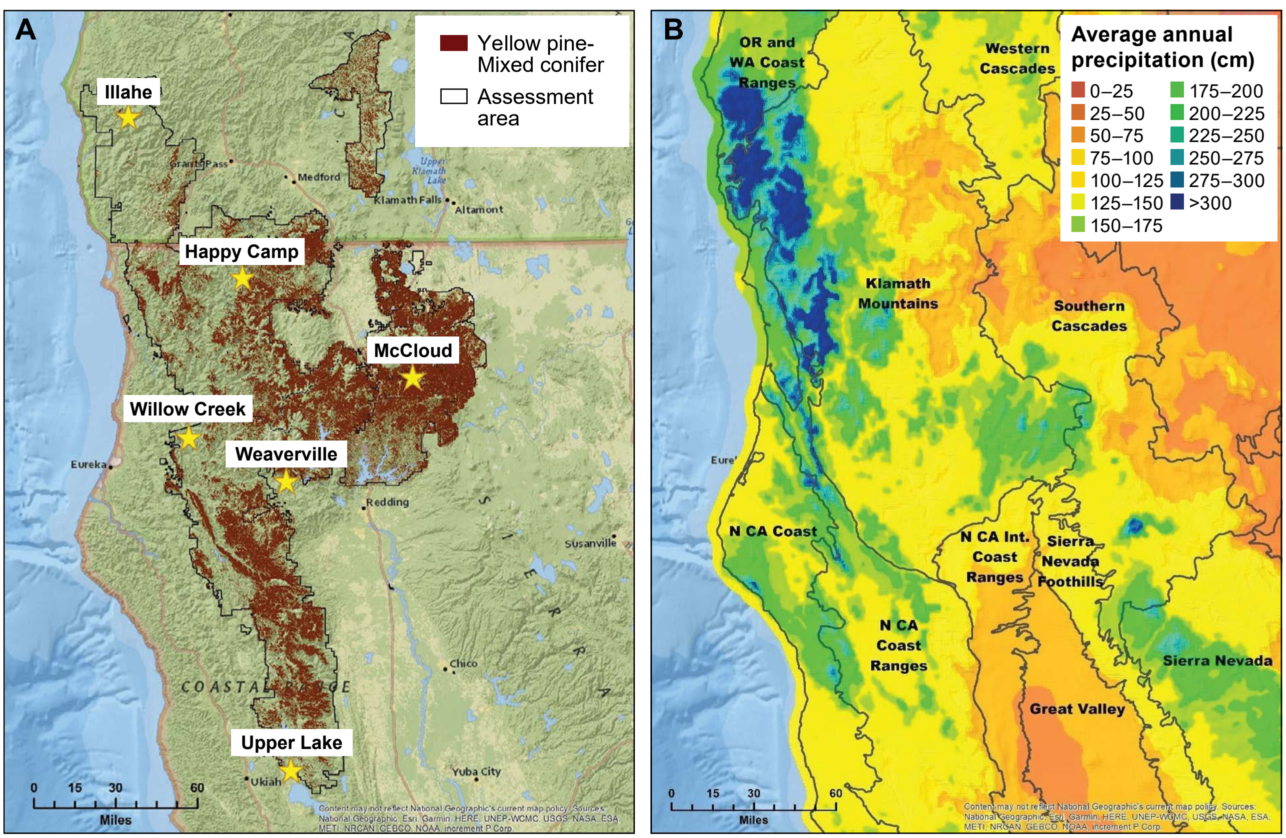


Figure 1—(A) Distribution of yellow pine and mixed-conifer forests in the assessment area, with climate station locations indicated; (B) ecoregions with average annual precipitation (1971–2000) provided (PRISM 2004; https://databasin.org/datasets/3eb601bde5d543bebea01edf856650d2).

In the wettest, coastal portions of the assessment area, redwood (*Sequoia sempervirens* (Lamb. ex D. Don) Endl.) and Douglas-fir-dominated forests (including mixed evergreen) are widespread. Moving inland, climatic water deficit gradually increases until ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) appears and then becomes a dominant species east of about longitude 123° 30′. The transition from low elevation to montane is indicated by a transition in the dominant shade-tolerant species from Douglas-fir to white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), which increases in dominance with elevation until giving way to Shasta red fir (*Abies magnifica* A. Murray bis var. *shastensis* Lemmon) (upper montane zone) and eventually mountain hemlock (*Tsuga mertensiana* (Bong.) Carrière) (subalpine zone) (Skinner et al. 2018). Douglas-fir and mixed-evergreen forests will be addressed fully in the Douglas-fir natural range of variation (NRV) assessment and are only discussed in this assessment when relevant.

Geology in the assessment area is very diverse and plays a major role in governing the distribution of forest types as well as the structure and productivity of vegetation throughout the assessment area (Sawyer 2006, Whittaker 1960). The Klamath Mountains are the most geologically complex portion of the assessment area. They are made up of mostly Paleozoic and Mesozoic metamorphic and sedimentary rocks, including some areas of carbonates, with granitoid and ultramafic rocks distributed throughout. The North Coast Range is largely underlain by Mesozoic sedimentary rocks from the Franciscan Formation, while the portion of the southern Cascades that falls within the assessment area is relatively young and composed primarily of tertiary and quaternary volcanic rocks (Jennings 1977).

YPMC forests are generally found at lower elevations or on south- and west-facing slopes, yet the complex geology present in the assessment area provides a notable exception to this pattern in the form of Jeffrey pine (*Pinus jeffreyi* Balf.) dominated forests that are widespread on ultramafic geological substrates. Ultramafic rocks (e.g., peridotite, serpentinite, pyroxenite, and other related rock types) are surficial outcrops of the earth's upper mantle, and are rich in magnesium and iron and poor in silica and macronutrients necessary for plant growth, such as calcium and potassium. Ultramafic rocks also include elevated levels of toxic elements, such as nickel, chromium, and cobalt, and they promote high pH conditions in both soil and water (Alexander et al. 2007). Ultramafic rocks occur throughout the assessment area in a wide range of elevations and climatic conditions. Outcrops vary greatly in area, with the largest rock bodies found in the Josephine and Trinity ophiolites (both >3000 km^2) in the western and eastern Klamath Mountains, respectively, and numerous smaller outcrops scattered throughout the Klamath Mountains and the North Coast Range (but not in the southern Cascades) (Sawyer 2006). Vegetation composition and structure found on soils developed on ultramafic rocks (colloquially referred to as "serpentine" soils) are often vastly different than on non-ultramafic soils. For example, serpentine areas often support open forests where neighboring, more fertile soils support dense forests; shrublands where there are forests on neighboring soil types; or grasslands where there are shrublands on neighboring soils. The challenging growing conditions on serpentine soils lead to a high number of endemic and rare plant species. California supports one of the richest serpentine floras on earth, and associated endemics comprise almost 15 percent of Californian endemic plant taxa and nearly a third of plant species managed as "sensitive" by the national forests in northwest California (Safford 2011, Safford and Miller 2020). Serpentine plant communities appear to be stable over the long term in regard to climatic change because of the difficult growing conditions and limited number of species capable of adapting to such conditions (Briles et al. 2011).

Carbonate rocks (sedimentary or metasedimentary rocks that are rich in calcium or magnesium) also occur in the assessment area, primarily in the Klamath Mountains. These include limestone, dolomite, and marble and are much less extensive than ultramafic rocks. Carbonate rocks promote alkaline water and soil conditions and also support a number of endemic and rare plant species (e.g., Engelhardt et al. 2012, Van de Ven et al. 2007).

The assessment area falls within the Mediterranean climate zone of North America and is characterized by wet, cool winters and dry, warm summers. The area exhibits two subcategories within the Köppen (1931) Mediterranean climate classification: "Csa"—warm-summer Mediterranean (broadly the western half of the assessment area and high elevations to the east) and "Csb"—hot-summer Mediterranean (much of the eastern half). Compared to the rest of California outside of the assessment area, the annual summer drought (typical for all Mediterranean climates) is shorter and interannual variability in precipitation is lower (Dettinger et al. 2011).

High elevations and northern latitudes near the coast tend to receive copious annual precipitation (more than 200 cm in some areas) and are capable of sustaining a deep snowpack well into summer. Lower elevations, especially at inland locations, tend to receive less precipitation (below 50 cm in some areas), most of which comes as rain during the winter months; however, there is substantial local and regional variation present (Whittaker 1960, WRCC 2016). An important exception to this can be seen in the eastern Klamath Mountains, where the Sacramento, McCloud, and Pit River watersheds are noted for having uniquely high annual precipitation (see fig. 1b). The dry season in most of the assessment area occurs between June and September and can be seen in figure 2 as the area where the precipitation line drops below the temperature line. Summer fog is a major buffer against the summer drought in coastal areas and permits the existence of forests supporting redwood, Sitka spruce (*Picea sitchensis* (Bong.) Carrière), western red cedar (*Thuja plicata* Donn ex D. Don), and other indicators of cool coastal conditions.

Ecological Setting

Throughout the assessment area, biogeographic history, climate, topography, and parent material work in concert to create a heterogeneous distribution of vegetation types.

Throughout the assessment area, biogeographic history, climate, topography, and parent material work in concert to create a heterogeneous distribution of vegetation types. Relict species from earlier climatic periods combine with floras originating from surrounding mountain ranges in California and Oregon to create the most diverse conifer forests in North America, if not the world (Cheng 2004; Stebbins and Major 1965; Whittaker 1960, 1961). There are numerous paleoendemic plant taxa in the region—these are taxa left over from early climatic regimes whose ranges are now restricted to the assessment area. Well-known examples include Brewer spruce (*Picea breweriana* S. Watson), Sadler oak (*Quercus sadleriana* R. Br. ter) and Shasta snow-wreath (*Neviusia cliftonii* Shevock, Ertter, & D.W. Taylor). There are many more neoendemics—species that have evolved recently, presumably due to the very high habitat, climate, and substrate variability in the assessment area. These include many taxa of buckwheats (*Eriogonum* spp.), mustards (family Brassicaceae, including genera such as *Arabis*, *Draba*, and *Streptanthus*), lilies (multiple genera in the Order Liliales, including *Calochortus*, *Lilium,* and *Triteleia*), broomrapes and plantains (including *Antirrhinum*, *Castilleja, Penstemon,* and *Veronica*) and tarweeds (e.g., *Harmonia* and *Hemizonia*) (Sawyer 2007). With respect to the assessment area YPMC tree flora, although it includes a number of rare and relict species, most tree species are shared with other YPMC-type forests across the North American Mediterranean climate zone, and some are widely distributed across the western portion of the United States (e.g., ponderosa pine, white fir, Douglas-fir; Safford and Stevens 2017; Sawyer 2006, 2007).

Indicator Species, Vegetation Classification, and Geographic Distribution

The yellow pine component of YPMC forests in the assessment area is represented by ponderosa pine and Jeffrey pine. Of the two, ponderosa pine is the most widespread geographically, occurring throughout western North America. Jeffrey pine's range is limited mostly to California, with some occurrences in westernmost Nevada, southwestern Oregon and northern Baja California (Barbour and Minnich

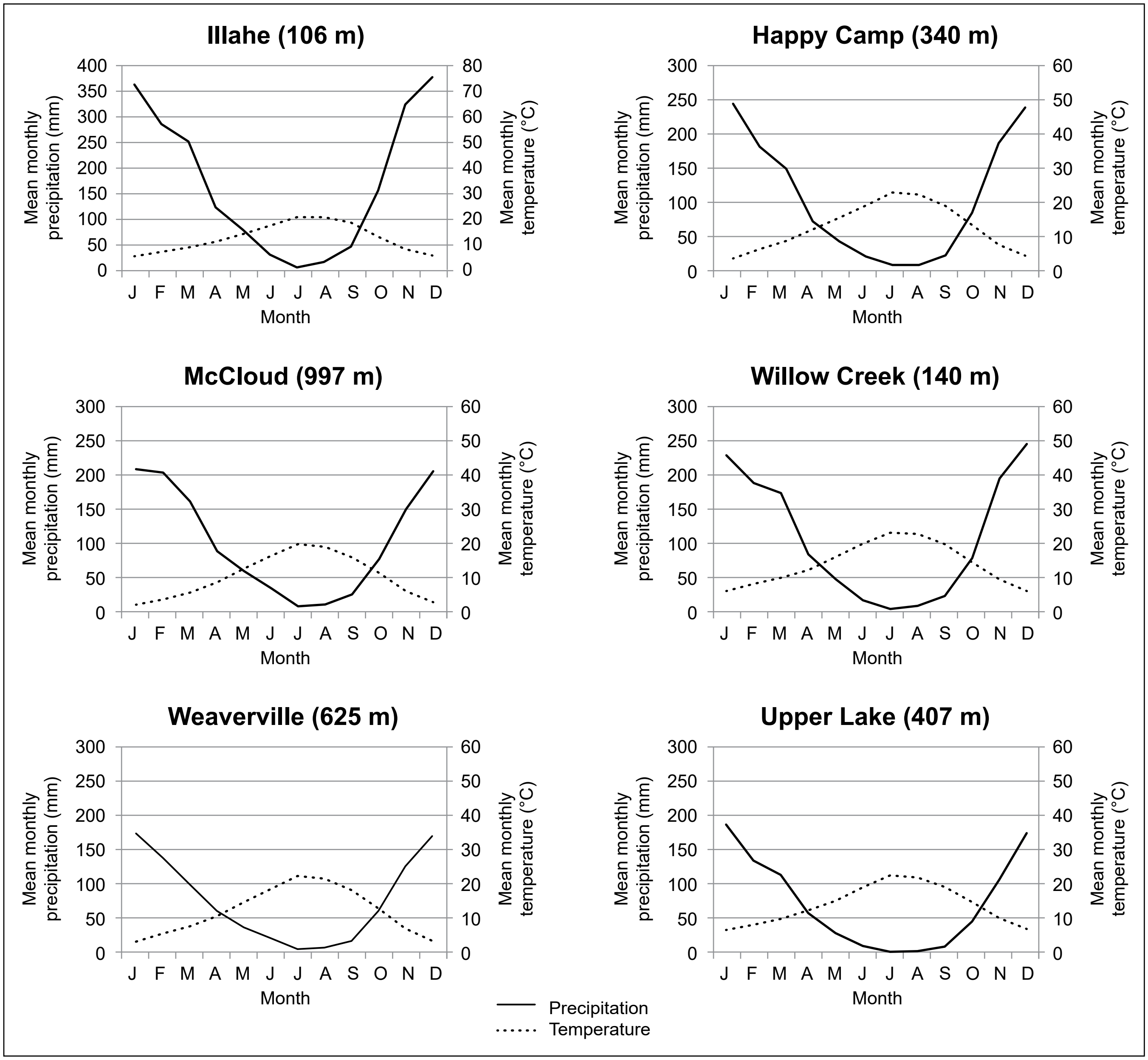


Figure 2—Walter type climate diagrams for six stations in or near yellow pine and mixed-conifer forest in the assessment area; elevations provided in parentheses (see fig. 1 for locations). X-axis represents the months of the year. Dark line represents precipitation (left Y-axis, mm), dotted line is mean monthly temperature (right Y-axis, °C). Dry season length is approximately the period during which the precipitation curve undercuts the temperature curve. Source: https://wrcc.dri.edu/summary/Climsmnca.html.

2000, Haller 1959). The two species are closely related (and are known to hybridize at times), but differences in their ecological tolerances separate where they tend to occur on the landscape. Jeffrey pine is more stress tolerant and will replace ponderosa pine on poorer soils, in drier locations, where cold air collects, and at higher elevations (Haller 1959; Merriam 1899; Sawyer 2006, 2007; Skinner et al. 2018). Its distribution throughout the assessment area is primarily on ultramafic serpentine soils, and when found on nonserpentine soils, it is typically on dry slopes and ridges above 1500 m (Griffin and Critchfield 1972). Ponderosa pine is widespread in low- and mid-elevation forests and is an important component of YPMC forests throughout the Western United States. In the assessment area, ponderosa pine can also be found as a minor component in mixed-evergreen and Douglas-fir forests (Griffin and Critchfield 1972).

The topographic complexity of the assessment area makes it difficult to identify clear ecological zones by elevation. Ponderosa pine-dominated forests are common from about 300 m in the Klamath Mountains up to about 1700 m in the southern Cascades (Merriam 1899; Sawyer 2006, 2007). At lower elevations, ponderosa pine can be found growing on its own, forming open woodlands, or as a main component of mixed-conifer forests. Open ponderosa pine woodlands are common on steep, dry, south- and west-facing slopes and in dry sites throughout the montane zone. Western juniper (*Juniperus occidentalis* Hook.), Douglas-fir, Oregon white oak (*Quercus garryana* Douglas ex Hook.), black oak (*Quercus kelloggii* Newberry), and incense cedar (*Calocedrus decurrens* Torr.) along with other species can be found mixed in with these dry ponderosa pine woodlands.

Mixed-conifer forests in the assessment area support many tree species, but they are dominated by variable combinations of ponderosa pine, incense cedar, sugar pine (*Pinus lambertiana* Douglas), Douglas-fir, white fir, and California black oak. This species mix is similar to mixed-conifer forests found in the Sierra Nevada; however, in the Klamath region, there tends to be a more notable component of Douglas-fir and hardwood species due to the increased rainfall and proximity to the coast (Griffin and Critchfield 1972). The most prominent hardwoods include California black oak, Oregon white oak, canyon live oak (*Quercus chrysolepis* Liebm.), Pacific madrone (*Arbutus menziesii* Pursh), golden chinquapin (*Chrysolepis chrysophylla* (Douglas ex Hook.) Hjelmqvist), tanoak (*Notholithocarpus densiflorus* (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh), and bigleaf maple (*Acer macrophyllum* Pursh) (Skinner et al. 2018).

In the western portion of the assessment area, where maritime climate effects are strong, mixed-conifer stands are mostly dominated by Douglas-fir at both low and middle elevations, but include varying amounts of sugar pine, ponderosa pine, incense cedar, and—with increasing elevation—white fir. As noted above, Douglas-fir-dominated forests are treated in the Douglas-fir NRV assessment. Ponderosa pine and Jeffrey pine woodlands tend to replace denser Douglas-fir-dominated mixed-conifer forests on shallow serpentine soils, upper south-facing slopes and ridgetops, and in drier, low-elevation zones throughout the assessment area (Merriam 1899, Sawyer 2007, Sawyer and Thornburgh 1974, Skinner et al. 2018).

In the eastern portion of the assessment area, white fir becomes the dominant conifer species as elevation increases. The white fir zone (1300–1700 m in the central Klamath Mountains, 1700–1800 m in the eastern Siskiyou Mountains) still contains most of the same mid-elevation conifer species (e.g., Douglas-fir, ponderosa pine, incense cedar, sugar pine) as well as many of the same hardwood species, such as canyon live oak, California black oak, golden chinquapin, and bigleaf maple (Sawyer and Thornburgh 1974, Waring 1969). Above this zone, the forest transitions into one dominated by upper montane and subalpine species, such as Shasta red fir, mountain hemlock, western white pine (*Pinus monticola* Douglas ex D. Don), whitebark pine (*Pinus albicaulis* Engelm.), lodgepole pine (*Pinus contorta* Douglas ex Loudon var. *murrayana* (Balf.) Engelm.), and foxtail pine (*Pinus balfouriana* Balf.), and hardwoods are much less important (Skinner et al. 2018).

The high conifer diversity mentioned earlier in this section can be seen most prominently in what are commonly referred to as "enriched stands" of mixed conifer. Located in the Russian Wilderness in the Salmon Mountains, in an area that has been called the "Miracle Mile" (Kauffmann 2012), one can find 18 conifer species in an area of 1 mi^2 (2.6 km^2) (Sawyer and Thornburgh 1974) (http://blog.conifercountry.com/2014/07/18th-conifer-in-the-miracle-mile). In the Bear Basin Butte Botanical Area in the Siskiyou Mountains, there are 16 different conifer species in a 500-ha area (Kruckeberg and Lang 1997). Similar stands can also be found in sheltered basins of the Scott and Trinity Mountains (Cheng 2004). The high diversity of these stands can be principally ascribed to high landscape diversity, steep elevational and climatic gradients, and the role of biogeographic history, with the Klamath Mountains acting as a biological refugium for many species during climatic changes of the Late Cenozoic Era (Axelrod 1988, Whittaker 1960).

Pacific silver fir (*Abies amabilis* (Douglas ex Loudon) Douglas ex Forbes) and Alaska cedar (*Callitropsis nootkatensis* (D. Don) Oerst. ex D.P. Little), both regionally common in the Pacific Northwest, find their southern limits in the Klamath Mountains. Subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.) and Engelmann spruce (*Picea engelmannii* Parry ex Engelm.) are common in the western portion of the United States outside of California, but are present in only a few isolated stands in the assessment area. Foxtail pine, a species endemic to California with two disjunct populations (the other is in the southern Sierra Nevada), has scattered stands throughout the higher elevation Klamath Mountains (Griffin and Critchfield 1972, Sawyer and Thornburgh 1974). Brewer spruce is considered a relic from tertiary times (Wolfe 1969). It is endemic to the mountains of northwestern

California and southwestern Oregon, occurring on all soil types, but considered one of the rarest conifer species on the North American continent (Sawyer 2007). Port Orford cedar (*Chamaecyparis lawsoniana* (A. Murray bis) Parl.) also has a native range limited to southern Oregon and northern California and is found mostly along streams on ultramafic substrates. Most of these species are not important components of YPMC forests in the assessment area as they are more common at higher elevations or on moister soils, but they contribute to the uniquely high levels of conifer diversity found in the area.

Distribution, structure, and function of the major YPMC tree species are driven by species-specific tolerances and their adaptations to the physical and biotic environments of the assessment area.

Distribution, structure, and function of the major YPMC tree species are driven by species-specific tolerances and their adaptations to the physical and biotic environments of the assessment area (table 1). Yellow pines, for example, have a high tolerance for fire and drought, but struggle in low-light environments. Species such as white fir and incense cedar, on the other hand, are much more tolerant of shade and are considered less fire tolerant until they mature (Burns and Honkala 1990). Douglas-fir also becomes more fire tolerant as it increases in size, and is arguably the most fire-tolerant tree species in the assessment area when mature (Skinner et al. 2018). For a more indepth treatment of the ecology of the key conifer species present in YPMC forests, refer to the yellow pine and mixed-conifer NRV assessment for the Sierra Nevada (Safford and Stevens 2017).

As mentioned earlier, broadleaf/hardwood species are prominent features of the landscape in the assessment area. As with conifers, differences in ecological tolerances among these species drive differences in their distributions and their contribution to YPMC function, structure, and composition (table 1). For example, Oregon white oak and California black oak are extremely drought tolerant as well as resistant to low- and moderate-severity fire when mature, while species like golden chinquapin, Pacific madrone, and canyon live oak are all considered drought tolerant, but are less resistant to fire (USDA FS 2017c). An important characteristic of most of these hardwood species is their ability to resprout, especially after fire. Thus, although many hardwood species may not be as tolerant of fire as associated conifers, these same species are capable of recovering rapidly after fire, provided they have sufficient reserves to draw from.

Temporal Variability in the Ecological Setting: Holocene Forest Development

Shifts in tree and shrub abundance and distribution associated with changes in climate during the Holocene Epoch have been discussed in a number of studies that used pollen, charcoal, and lithological evidence from lake deposits in northern California and southern Oregon (e.g., Briles et al. 2005, 2008, 2011; Crawford et al. 2015; Daniels et al. 2005; Mohr et al. 2000; White et al. 2015). Most of these

Table 1—The ecological tolerances of common conifer and hardwood tree species in yellow pine and mixed-conifer forests in the assessment area

		Ecological tolerance			
Type	**Species**	**Shade**	**Drought**	**Fire[a] (young/mature)**	**Ability to resprout**
Conifer	White fir	High	Low	Low/intermediate	No
Conifer	Incense cedar	High	High	Low/intermediate	No
Conifer	Douglas-fir	Intermediate[b]	Intermediate	Low/very high	No
Conifer	Sugar pine	Intermediate	Intermediate	Low/high	No
Conifer	Ponderosa pine	Low	High	Intermediate/high	No
Conifer	Jeffrey pine	Low	High	Intermediate/high	No
Hardwood	Canyon live oak	High	High	Low/low	Yes
Hardwood	Tanoak	High	Intermediate	Low/intermediate	Yes
Hardwood	Bigleaf maple	Intermediate	Low	Low/low	Yes
Hardwood	Golden chinquapin	Intermediate	High	Low/low	Yes
Hardwood	Pacific madrone	Intermediate	High	Low/low	Yes
Hardwood	Oregon white oak	Intermediate[c]	High	Low/high	Yes
Hardwood	California black oak	Low	High	Low/high	Yes

[a]Fire tolerance does not take into account the ability to resprout after fire.
[b]Douglas-fir is considered shade intolerant in wetter sites and shade tolerant in drier sites.
[c]Developmental stages affect shade tolerance, with tolerance decreasing with age.
Table adapted from Safford and Stevens (2017), with additional information from Minore (1978) and USDA FS (2017c).

studies, especially those within the assessment area, took place in upper montane or subalpine forest or on ultramafic sites. Although these studies show trends primarily in high-elevation forests, they allow for useful generalizations to be made with respect to the response of overall forest composition and fire frequency to shifts in climate (fig. 3). For example,

- The abundance of mesophytic, shade-tolerant species, such as fir and mountain hemlock generally increased during cool, wet periods and decreased during warm, dry periods.
- The abundance of xerophytic, shade-intolerant species, such as pines and oaks generally increased during warm, dry periods and decreased during cool, wet periods.
- Structurally, forests became more open in response to warming and drying.
- Fire activity varied in response to climatic conditions, with fire activity generally increasing during warm, dry periods.
- Vegetation responded differently to climate change on ultramafic substrates compared to nonultramafic substrates, with species composition remaining generally stable on ultramafic substrates and showing variation mostly in relative abundance. On nonultramafic substrates, species composition would change, often moving in elevational shifts in response to warming or cooling.

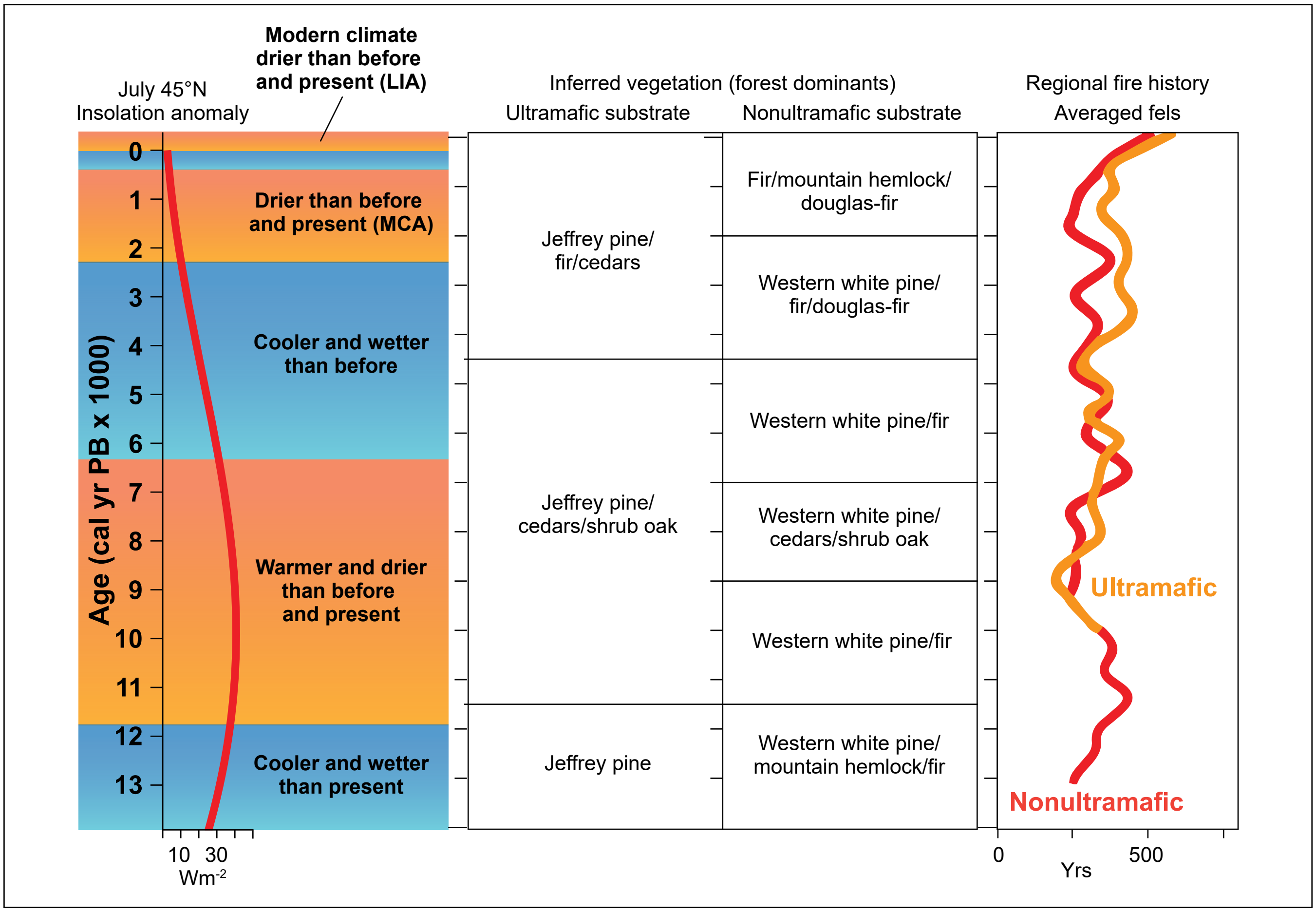


Figure 3—Overview of climate, vegetation, and fire history based on pollen and charcoal data from the assessment area. July 45°N insolation anomaly and general climate history inferred from alkenone-derived sea surface temperatures, speleothem $\delta^{18}O$-derived temperatures, and modeled climate (Barron et al. 2003, Bartlein et al. 1998, Vacco et al. 2005). LIA = Little Ice Age; MCA = medieval climatic anomaly. Figure modified from Skinner et al. 2018 where it was adapted from Briles et al. 2011.

During the Late Glacial Period (15,000 to 11,700 years before present [YBP]), a time of increasing summer insolation and subsequent deglaciation, conifers, such as white pines, fir (*Abies* spp.), and mountain hemlock increased their ranges upslope, and forests on nonultramafic substrates became more closed. Forests on ultramafic substrates largely remained open and dominated by yellow pines. Fire activity was low during this cool, wet period.

The early Holocene (11,700 to c. 8,000 YBP) began after the Late Glacial Period and was characterized by warm, dry conditions. Fire frequency peaked as a result of intensified summer drought conditions. Forests on nonultramafic substrates became more open and dominated by xerophytic species, such as white pines, cedars, and shrub oaks, while fir and mountain hemlock decreased in abundance. Forests on ultramafic substrates were similarly dominated by cedar and shrub oaks, in addition to yellow pine.

In most of the world, the Holocene Thermal Maximum (HTM) occurred between about 7000 and 6000 YBP, with the period 8000 to about 4000 often referred to as the "Xerothermic" or "Altithermal" period (Antevs 1955). There is high variability in the timing of the HTM, however, even within the relatively small geographic area represented by the assessment area. Middle Holocene cooling began as early as about 7,000 YBP in some assessment area (or nearby) paleorecords (e.g., Briles et al. 2011, Worona and Whitlock 1995), but after 6000 or even 5000 YBP in a number of other places (e.g., Mohr et al. 2000). Cooling brought on the expansion of mesophytic species such as firs, which replaced white pines on nonultramafic substrates, and Douglas-fir became more abundant in the Siskiyou Mountains. Forests became more closed on nonultramafic substrates, but remained open on ultramafic substrates, with yellow pine and firs increasing as shrub oaks and cedars decreased.

During the late Holocene (c. 4,000 YBP to present), summers became cooler and wetter and modern vegetation established. Fir and mountain hemlock increased in dominance along with Douglas-fir on nonultramafic substrates, while yellow pines further increased and shrub oaks decreased on ultramafic substrates. Fire activity remained relatively high until about 1,000 YBP, when it decreased to unusually low levels. Centennial scale changes in vegetation and fire frequency during this time are also evident. During the Medieval Warm Periods, between 1,050–850 YBP and 750–600 YBP, the climate became relatively warm and dry. Shade-intolerant taxa increased, as did accumulated charcoal, indicating an increase in fire prevalence and associated shifts in vegetation linked to warming and drying (Briles et al. 2011, Colombaroli and Gavin 2010, Daniels et al. 2005, Mohr et al. 2000). What is referred to as the Little Ice Age (c. 600 to 150 YBP) followed the Medieval Warm Periods. This was a cool, wet period that led to a decrease in charcoal accumulation and an increase in shade-tolerant taxa, such as fir, Douglas-fir, and hemlock (Briles et al. 2011, Colombaroli and Gavin 2010, Mohr et al. 2000). It is important to note that the historical reference period frequently used for comparisons with modern conditions, including the fire regime, is more or less coincident with the Little Ice Age. This is important to keep in mind because current trends are moving toward a climate that is much warmer than the conditions during the Little Ice Age.

It is important to note that the historical reference period frequently used for comparisons with modern conditions, including the fire regime, is more or less coincident with the Little Ice Age.

Modern climate and future projections—

The earth's modern climate has been greatly affected by humans. Carbon dioxide in the atmosphere now exceeds 400 parts per million and continues to rise. This is leading to increases in global temperatures and a subsequent increase in severe weather events (NOAA 2017, Pachauri and Meyer 2014). According to Williams et al. (2020), anthropogenic climate trends superimposed over natural climate variability may be pushing southwestern North America (western portion of the United States and northern Mexico) into an extensive megadrought because of enhanced evaporative demand and early snowpack loss. Mean annual temperatures across the assessment area have warmed by about 1.1 °C since the beginning of the 20th century, with most of the change occurring in mean minimum temperatures (+1.4 °C on average) (Butz et al. 2015, Rapacciuolo et al. 2014). Recent drought years notwithstanding, mean annual precipitation in the assessment area has mostly increased over the past century despite high variability in precipitation trends from station to station (Rapacciuolo et al. 2014).

Although no climate change modeling efforts have specifically targeted the assessment area, studies based on general circulation models (GCMs) that include the assessment area can help identify potential trends of future climate scenarios. Micheli et al. (2018) projected significant increases in temperature (4 to 6 °C increase for average winter and 3 to 5 °C for average summer temperatures) by the end of the century. Precipitation projections on the other hand are highly variable with disagreement as to whether long-term trends will be positive or negative. Despite the disagreement in the amount of precipitation, snow accumulation is projected to decrease considerably, resulting in a reduction of snowpack as the rain vs. snow proportion increases (Micheli et al. 2018). This shift from snow to rain will reduce the amount of available moisture on the landscape during late spring and early summer from snowmelt, while extending the dry season later into the fall, putting added drought stress on existing vegetation and lengthening the already expanding fire season (Westerling 2016).

Wahl et al. (2019) recently looked at the influence of the North Pacific Jet Stream on cool-season moisture and fire extremes in California from 1600 to the present. Until 1903, the position of the Jet Stream was linked to both the amount of winter precipitation and the severity of the subsequent fire season. Typically, high-precipitation years were coupled with low-fire years. This basic relationship between the Jet Stream and precipitation has remained the same since 1903, but the Jet Stream's relationship with fire extremes has weakened. After 1977, the connection between the Jet Stream and the severity of the fire season disappeared altogether. Fire conditions are becoming more thermodynamically controlled, meaning that high-precipitation years are coupling with high-fire years, as seen in 2006 and 2017. This coupling of high precipitation with high-fire years is due to increases in temperature and corresponding reductions in snow-to-rain ratios and is a circumstance that was not observed between 1600 and 1903 (Wahl et al. 2019).

Cultural Setting

The assessment area was inhabited by humans long before European contact. At the time of contact, more than 20 distinct languages were spoken in the region, pointing to the diversity of tribal groups present (Golla 2007). Archaeological patterns since the middle Holocene can be broken into three distinct periods: Borax Lake, Mendocino, and Gunther (Hildebrandt 2007). The Borax Lake Pattern lasted from ~8,000 YBP to ~4,500 YBP and is defined as being a period of subsistence focused on terrestrial habitats. In this period, tribal hunter-gatherer populations were not sedentary and lived and foraged at higher elevations. The Mendocino Pattern, from ~4,500 YBP to ~1,500 YBP, refers to a period in which settlements were established at lower elevation in valleys and along rivers. Hunting camps were still present at higher elevations, but residential sites were predominantly in the lowlands. The Gunther Pattern, which began roughly 1,500 YBP and lasted until European contact, was defined by even more sedentary behavior, during which populations were increasing due to immigration from the north (Golla 2007).

It is clear from various lines of evidence that the presence of indigenous populations prior to European contact had a large effect on vegetation in the region. The pristine wilderness observed by early explorers and settlers was actually the result of millennia of human interactions with their environment (Anderson 2005). Fire, for example, was a vital tool to create and maintain open forests to help facilitate hunting, promote acorn production and the growth of berries and bulbs, and enhance access to other resources such as basket-making materials (Anderson 2005, Hildebrandt 2007, Lewis 1993). Widespread American Indian use of fire is clearly behind the very high fire frequencies in the fire-scar record in redwood forests and oak woodlands in the westernmost portion of the assessment area, where lightning strike densities are extremely low (Taylor et al. 2016, Van de Water and Safford 2011).

The presence of indigenous populations prior to European contact had a large effect on vegetation in the region.

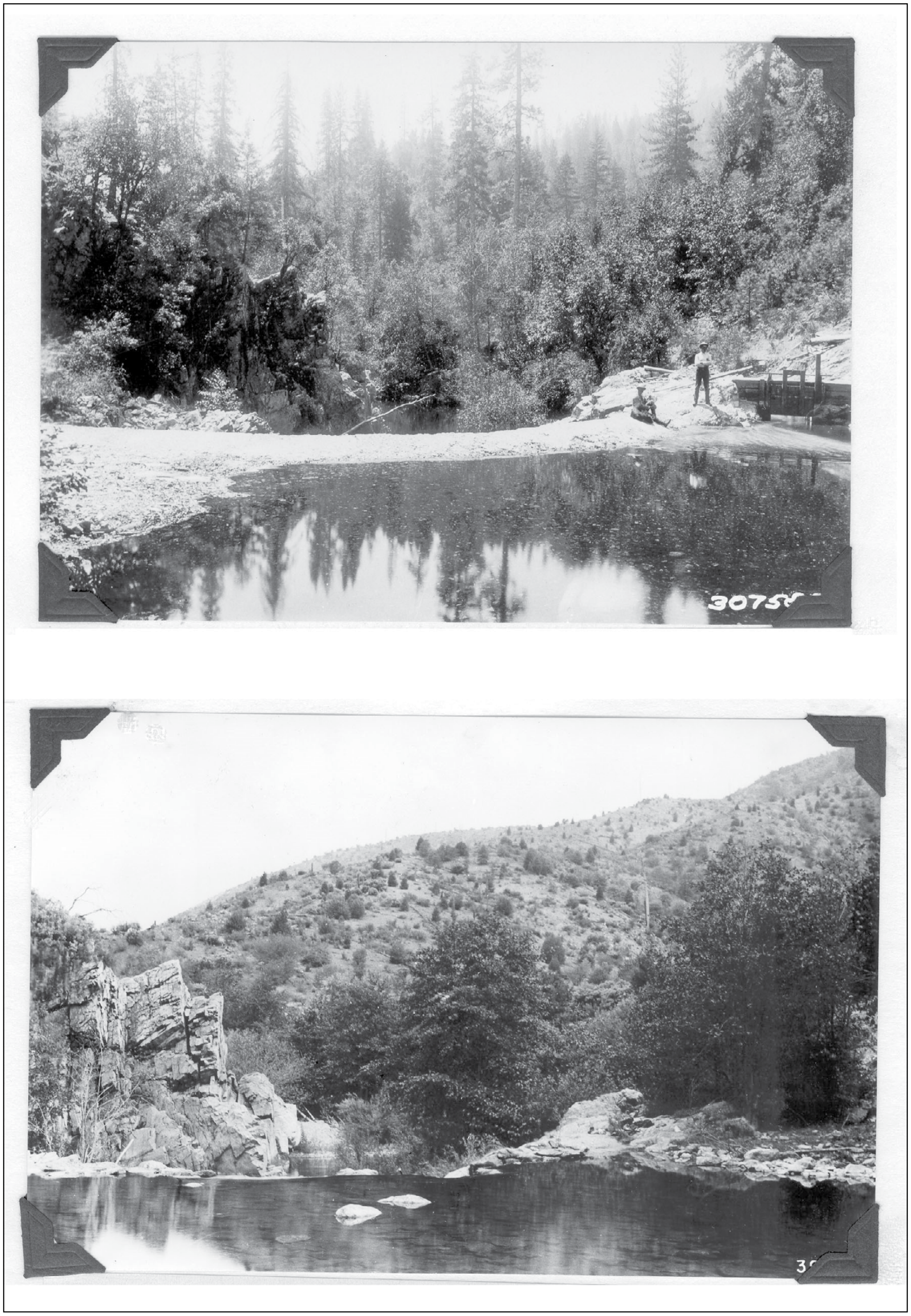
Over the past hundred years, the climate has become warmer and, based on temperature-fire occurrence correlations in the paleo-record, we would expect an increase in fire events and more open forests. In fact, however, the period has been characterized by a decrease in fire and an increase in forest density; both of which stem from the suppression of fires by Euro-Americans (Agee 1991, Skinner et al. 2018, Taylor et al. 2016, Van de Water and Safford 2011).

In lower montane forests of the Sierra Nevada, Taylor et al. (2016) identified four time periods during which fire regimes shifted, and identified the socioecological periods associated with them: (1) 1600–1775 (presettlement [pre-Spanish-Colonial]), (2) 1776–1865 (Spanish-Colonial), (3) 1866–1903 (Gold Rush-settlement), and (4) 1904–2015 (fire suppression). The close coupling of fire regime shifts with socioecological shifts indicates the immense impact humans have had on fire activity in these forests. YPMC forests of the assessment area may not have been affected as strongly (or within the exact same timeframe) as the Sierra Nevada region, but similar fire regime shifts were occurring (Fry and Stephens 2006, Skinner et al. 2009). Today, strong departures from historical fire frequencies are apparent throughout the assessment area (Safford and Van de Water 2014).

Although Russian and Spanish settlers were present in western North America as early as the 18th century, colonial settlements were mostly concentrated in central and southern California along the coast. By the early 19th century, explorers were traveling through the Sacramento Valley into Oregon and along the northern coast, but exploration into the steep, rugged mountains in the assessment area was limited. Fur trappers traveled through the region working the rivers and streams, but never established permanent settlements. The only consistent occupants in these areas at the time were American Indians (Dale et al. 1941, Pullen 1995, Wells 1881). Shortly after the discovery of gold in California, people flocked to California, which initiated the period of Euro-American colonial settlement, drastically increasing the nonnative population, while significantly reducing and displacing the native population. By 1849, a large number of Euro-Americans had migrated west with the intention of occupying the land north of San Francisco Bay and creating an independent state out of the region that is now northern California. According to Wells (1881), the population increased by more than 200,000 (mostly men) from 1849 to 1852. The colonial settlement by Euro-Americans and the introduction of diseases unknown to the indigenous peoples devastated American Indian populations and led to a precipitous reduction in American Indian resource management practices (Anderson 2005, LaLande 1980, Stephens and Sugihara 2006).

After gold was discovered on the Trinity River in 1848 and in the Siskiyou Mountains a few years later, prospectors made their way into the upper, unexplored regions of the Trinity, Salmon, Klamath, and Scott Rivers, beginning the exploitation of resources in the far reaches of the assessment area (Wells 1881). Widespread logging took place near mining operations throughout the mid- to late 19th century to support the mines as well as the increasing populations surrounding them (fig. 4). The following quote taken from Wells (1881) describing Siskiyou County hints at the awareness some of these early settlers had of their already immense impact on the region (Wells 1881: 29):

> We sit in some cavernous depth or perch ourselves upon some commanding peak and think of the long centuries that rolled swiftly by, while the red men called this home and disturbed not with profaning hand the simple



USDA Forest Service, Pacific Southwest Research Station

Figure 4—Before (1905) and after (1939) photos of an area heavily affected by mining operations.

> order Nature had established in her chosen dominion. The deer, the bear, and the antelope roamed its valleys and penetrated the dense forests that covered its mountain sides; the simple natives lived in peace and quietude, so few and so retired that the first white men who passed through scarce saw them at all. This was the condition till thirty years ago, when the magic wand of gold was waved over the mountain tops, and a new race came to supplant the old, to level forests and disembowel the earth, to subdue the soil and deface the brow of Nature with the crown of civilization.

Early timber harvest was concentrated in more accessible areas that supported large stands of yellow and sugar pine and, to a lesser extent, Douglas-fir, white fir, and cedar (Lamm 1944). Access was the most limiting factor, but when railroad companies began operating in the region around the turn of the century, logging operations became much more widespread in the Sacramento River watershed and around Mount Shasta. Though it is often claimed that human-caused fire ignitions also became more prevalent, both accidental and intentional, with the goals of creating easier access to gold-bearing outcrops or increasing forage for livestock (LaLande 1980, Show and Kotok 1924), this claim is not supported by the tree-ring record of fire scars. The tree-ring record generally shows either no change or a decrease in fire frequency (Fry and Stephens 2006; Metlen et al. 2018; Skinner et al. 2009; Taylor and Skinner 1998, 2003; Taylor et al. 2016). As concern grew nationally about wasteful logging and fire becoming a threat to precious timber stands, policies focusing on fire suppression were instituted. Beginning in 1905, forest reserves (later called national forests) were established in order to facilitate better management and protection of forest and watershed resources (Pinchot 1907, Stephens and Sugihara 2006).

The new fire-suppression policy was not universally accepted, and many ranchers and lumbermen managing old-growth timber believed that frequent, low-severity fire was an important contributor to forest health (Cermak 2005, Show and Kotok 1924, Wilkes 1899). A letter to the editor that appeared in *The Oregonian* in September of 1899 was written by L.E. Wilkes, a private citizen at the time, and stated the following (Wilkes 1899):

> This season [autumn] offers an opportunity to employ what I deem the best means of preventing the ravages of forest fires. Much of the debris on the ground in our forests would now burn, if properly fired, and there is no danger of devastating fires getting started this fall. It is very seldom, if at all, that valuable timber is injured by fire, except where there is a large amount of dry, dead material on the ground.…Therefore, if systematic work

> be done by firing extensively over tracts where there is much offal, not only the danger of fires can be averted, but much useless material be put out of the way.

In the following quote from the back of an early Klamath National Forest map, Stuart (1928) informs the reader of the Forest Service's fire prevention policy and addresses those who believe in what he calls "the light-burning fallacy":

> The fire-protection policy of the Forest Service seeks to prevent fires from starting and to suppress quickly those that may start. This established policy is criticized by those that hold that the deliberate and repeated burning of forest lands offers the best method of protecting those lands from the devastation of summer fires. Because prior to the inauguration of systematic protection California timberlands were repeatedly burned over without the complete destruction of the forests, many people have reached the untenable conclusion that the methods of Indian days are the best that can be devised for the present. It is commonly assumed in this argument that controlled burning of the forests, either in the spring or fall, is an easy practice which can be carried out at slight expense, with negligible damage to the forest itself, and with complete or nearly complete removal of the accumulated debris which inevitably forms in any growing forest.

In spite of the findings of researchers and managers that frequent "light fire" on the landscape had little real effect on the timber resource (e.g., Show and Kotok 1924, 1929), the general fear of fire and its effects on tree regeneration led to statewide and national decisions to eliminate it. Fire suppression became the law of the land and—outside of a few national parks and wilderness areas—remains so today, despite increasing awareness of its accumulated impacts on forest ecosystems and the humans living in or near them (Cermak 2005, Stephens and Ruth 2005).

The general fear of fire and its effects on tree regeneration led to statewide and national decisions to eliminate it.

Grazing from domesticated livestock also increased dramatically during the settlement period and was largely centered near valleys and in the foothills during the winter months, moving into the mountains during the summer months (Pinchot 1905). Shepherds used fire on the landscape to remove encroaching woody plants and increase forage for their herds (LaLande 1980). Although there is limited knowledge surrounding the impacts of livestock grazing on the understory plant community of YPMC forests in the assessment area, it is likely that it caused a reduction in herbaceous cover, perhaps leading to a reduction in fine fuels (Fry and Stephens 2006) and thus a reduction in the general spread of fire (Riegel et al. 2018, Skinner et al. 2009). Current grazing impacts are far less extensive than they were in the late 19th and early 20th centuries, owing mostly to a cultural shift away from ranching.

Tribes in the assessment area today continue many of their subsistence and cultural practices where and when possible, within limits posed by land ownership and state and federal policies. There are a handful of examples of current efforts to reintroduce indigenous burning onto the landscape to promote healthier forests and safer communities as well as to provide important cultural resources. The Western Klamath Restoration Partnership, for example, is a collaboration between a number of organizations and agencies and includes members of the Karuk Tribe and the USDA Forest Service. The Partnership includes the entire Salmon River watershed in the western Klamath Mountains and aims to reach conservation targets based on identified values that include fire-adapted communities, restored fire regimes, and cultural and community vitality (Harling and Tripp 2014).

Chapter 2: Methods

Natural range of variation (NRV) is typically considered to be the spatial and temporal variation of ecological conditions, within a period of time and geographical area, that are relatively unaffected by people (Landres et al. 1999). Historical range of variation (HRV) is a related concept that was developed to allow for incorporation of human influences on ecosystems because, in most places on Earth, humans have been major ecological players for millennia (Wiens et al. 2012). In this assessment, we evaluate the period in general of the Holocene and in more detail of the last several centuries when humans have been affecting ecosystem conditions, making it more appropriate to call it an HRV assessment. There is no period with detailed information that is without human influences on the ecosystem, yet Forest Service guidance for the implementation of its 2012 planning rule has adopted the term "natural range of variation," so we have decided to follow suit.

Historical Reference Period

In this NRV assessment, our principal reference period is the 16th century to the late 19th century, which include the three to four centuries prior to significant Euro-American colonial settlement of the assessment area. It is important to highlight that this reference period is coincident with the Little Ice Age, which should be taken into consideration when using NRV reference conditions to help inform management targets (Millar et al. 2007; Safford et al. 2012a, 2012c). We also collected and interpreted information as far back as the beginning of the Holocene Epoch (12,000 YBP) when it was available to better understand patterns and processes from warmer, drier periods in the past because most climate projections for the assessment area project much warmer and somewhat drier conditions (at least during the growing season) by the end of the 21st century.

Spatial Scale

Our historical and contemporary reference data sources are more often than not derived from specific locations or landscapes, but our analysis is intended to apply to the assessment area as a whole. Wherever possible, we sought data that represented the variety of different geographic regions and environmental situations that are found in the assessment area. Usually, though, we simply had to accept the limitations of those data we could find and use inference and our understanding of environmental variation across the bioregion to extend those data points to the larger assessment area. We report the geographic locations of our data sources throughout the report.

Information Sources

This NRV assessment is based on both historical and contemporary reference sites and information sources. Historical data are limited and, in most cases, it was necessary to evaluate information sources that postdated the colonial settlement of Euro-Americans in the assessment area. Whenever possible we used "contemporary reference ecosystems," which are ecosystems that have suffered relatively little degradation and may serve as a more natural reference against which degraded ecosystems may be compared. Because human alteration and degradation of assessment area ecosystems is so pervasive, identification of appropriate reference ecosystems is difficult in all instances, and impossible in some.

In our assessment, we used direct data analysis and interpretation whenever possible, and we resorted to inference where necessary and justifiable. This NRV assessment includes comparisons to current conditions, as well as a summary of the literature regarding possible future trends, whenever that literature existed. Our focus was on peer-reviewed publications, including papers in press or soon to be in press; government publications; Forest Service and other federal and state agency data; and in some cases, academic theses or dissertations. Because information on the historical state of some ecosystems and ecological processes and patterns is scarce, we also refer to published anecdotal information from the mid-19th to early 20th centuries in some cases. We do not refer to anecdotal information from more recent times.

We used data compilations from the Forest Service's Forest Inventory and Analysis (FIA) program to provide current-day data on many forest structure and composition variables. Plots are found across the United States and are located randomly within a grid defined by latitude and longitude. It is important to note that FIA data provide a statistically robust sample of all stand conditions across the assessment area, including areas with reduced tree density and cover owing to natural disturbances or harvest (USDA FS 2017b). These data were provided by the Pacific Southwest Region Remote Sensing Laboratory. We do not include FIA data from the southwestern Oregon portion of the assessment area.

Determination of Deviation from NRV

In this NRV assessment, we relied primarily on a "range of means" approach where we based our assessment on an estimate of the range of means from multiple sources for a given variable. This produces a narrower, more discernible, and probably more meaningful range of variation that can be quantitatively or qualitatively compared to modern data. Determination of deviation from NRV was accomplished by comparing the modern range of variation for some indicator variable with the range of means for the same variable from the NRV period or contemporary reference sites. Direct statistical comparisons were rarely possible, owing to small sample sizes in the reference sources, the lack of measures of statistical variation, orders-of-magnitude differences in sample sizes between current and historical data when multiple historical data points did exist, or the lack of concrete quantitative measures in the historical dataset. Our assessment of current deviations from NRV was necessarily deductive in nature, and we came to conclusions about the status of specific variables based to a great extent on our general knowledge about the ecosystems in question. Table 9 summarizes our conclusions about current deviation (or "departure") from the NRV for key ecosystem elements in yellow pine and mixed-conifer forests in the assessment area.

Chapter 3: Natural Range of Variation Descriptions

Function (Including Disturbance)

Extreme Climatic Events

Drought—

Drought periods can be temporary (a few months) or prolonged (multiple years) and are generally defined as times when precipitation is low and temperatures are high compared to relatively long-term averages, which results in an unmet biological need for water. Part of what defines a Mediterranean climate is the prolonged annual drought period (3 to 4 months in most of the assessment area) during the summer months (fig. 2). While most vegetation in the assessment area is adapted to these annual periods of drought, drought conditions that extend for multiple consecutive years put added stress on plant communities, leading to a greater susceptibility to diseases; pests; and large, high-severity fires (McKenzie et al. 2009). High stand densities can further amplify these effects by increasing competition for resources, as well as increasing live and dead fuel continuity and vulnerability to stand-replacing fire (Logan et al. 2003, van Mantgem et al. 2013).

Natural range of variation (NRV)—Since the beginning of the Late Holocene, drought periods have ranged from 20 to >200 years in length, with periods between droughts ranging from 80 to 230 years (Benson et al. 2002, Stine 1994). During the Medieval Climate Anomaly (A.D. 800–1350), California experienced extensive periods of drought lasting more than a century in some cases (Stine 1994). Although this drought record was derived from evidence in the Sierra Nevada, it can be inferred from evidence of corresponding shifts in vegetation patterns that similar conditions were affecting the assessment area. Shade-intolerant taxa increased throughout the assessment area during the Medieval Climate Anomaly as did accumulated charcoal, indicating an increase in fire prevalence and associated shifts in vegetation during this warm, dry period (Briles et al. 2011, Colombaroli and Gavin 2010, Daniels et al. 2005, Mohr et al. 2000).

Comparison to current—Between 2012 and 2015, California experienced one of the most severe periods of drought during the past 1,200 years. Multiple consecutive years of low yearly precipitation totals combined with record high temperatures contributed to the uncharacteristically severe nature of this event (Griffin and Anchukaitis 2014). While central and southern California were substantially affected by this drought, it was not quite as severe in the assessment area (see fig. 5). Asner et al. (2016) assessed changes (percentage loss) in forest canopy water content during the 2012–2015 period. Lower elevation forests and woodlands were the most affected early on by the drought, while higher elevation forests showed extensive canopy water loss by 2015. Although canopy water loss is not representative of actual tree mortality, it indicates an increase in vulnerability to hotter drought conditions (Allen et al. 2015, Asner et al. 2016).

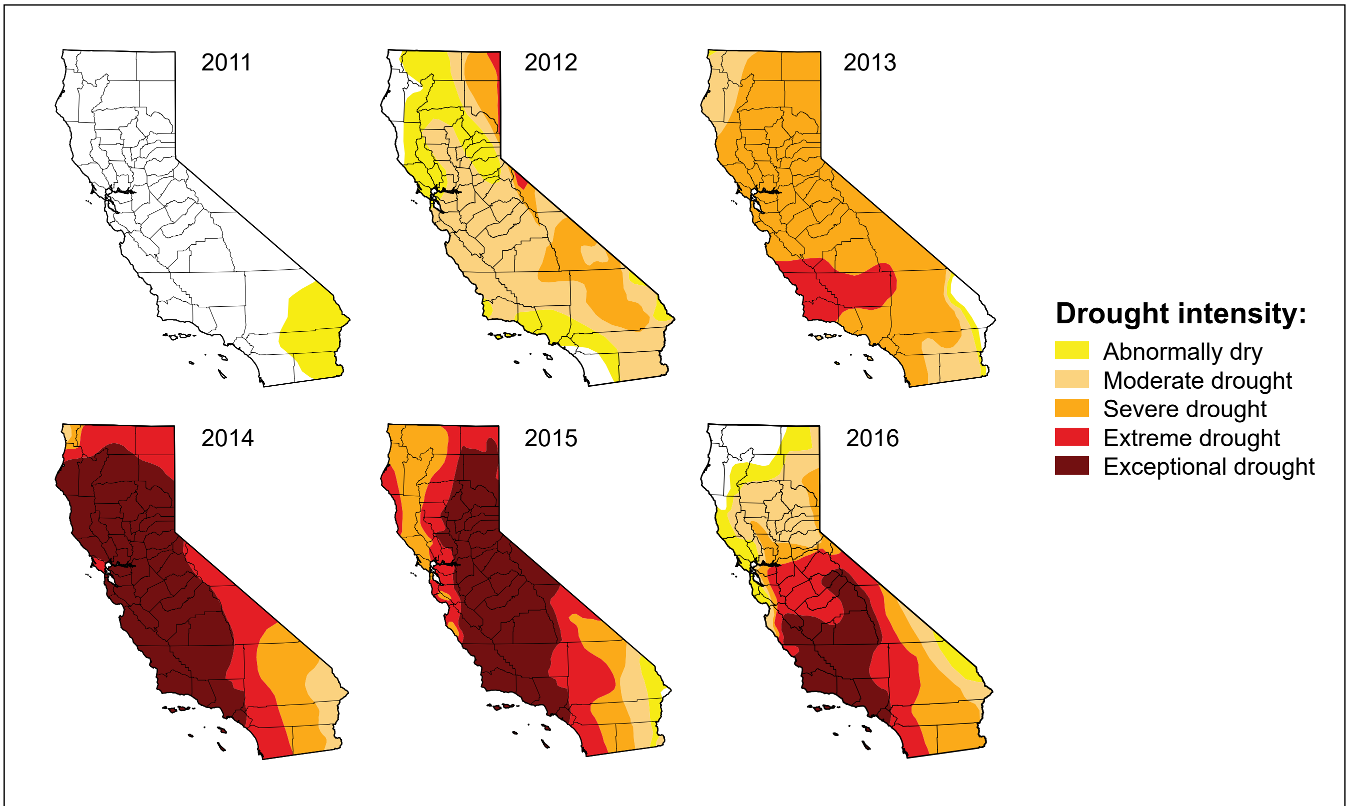


Figure 5—California annual drought conditions from 2011 to 2016, as mapped by U.S. Drought Monitor (http://droughtmonitor.unl.edu).

Despite recent drought conditions in California, the assessment area has shown a general increase in annual mean precipitation over the past century (Butz et al. 2015, Rapacciuolo et al. 2014). Different areas in the region are experiencing contrasting precipitation trends though, with some areas experiencing increased precipitation, while other places are not changing or are becoming drier (Butz et al. 2015, Rapacciuolo et al. 2014). In general, mean and minimum temperatures have both increased, while maximum temperatures have shown no trend or even a general decrease over the past 100 years in northwestern California (Rapacciuolo et al. 2014). Due to increases in minimum temperatures, the percentage of precipitation falling as snow has decreased and the rate of snowpack melting has increased, resulting in associated changes in runoff and recharge rates (Thorne et al. 2015).

Fire suppression policies have amplified the negative impacts of drought on yellow pine and mixed-conifer (YPMC) forests. High tree densities in assessment area YPMC forests are, in large part, the result of the recruitment of shade-tolerant trees in both second-growth and old-growth stands, thus increasing drought-induced stress and the potential for large-tree mortality events (Guarin and Taylor 2005, Leonzo and Keyes 2010, McKenzie et al. 2009; Young et al. 2017). Increased vulnerability to insect infestations and disease is a significant issue during and after large drought events (Guarin and Taylor 2005), and extensive mortality has already occurred throughout California (Moore et al. 2016). Aerial surveys conducted in 2015 in the assessment area provided an estimate of about 1.75 million dead trees associated with drought across the 12.6 million acres surveyed. One hundred and fifty-seven thousand acres contained scattered mortality of ponderosa pine at low intensities as well as scattered mortality of white fir and other pine species. Bost et al. (2019) combined plot and LANDSAT data to show that tree mortality increased in mixed conifer, red fir, and subalpine forest stands in the Russian Wilderness during the 2012–2016 drought, with mortality highest in older forest stands on dry slopes, especially above 2000 m in elevation. In their California study, van Mantgem et al. (2009) found that, in general, tree mortality rates have increased since the mid-1980s, attributing this increase to increases in mean annual temperature and climatic water deficit. It should be noted that the data were limited to 20 plots, all of which were located outside of the assessment area. The apparent buffer to severe drought in the assessment area, as compared to other parts of California, has resulted in increased tree mortality largely being limited to areas that have experienced the most extreme drought conditions (i.e., eastern Klamath National Forest, northern and eastern Mendocino National Forest, southern Shasta-Trinity National Forest) (Moore 2015, Moore et al. 2016).

Future—According to Williams et al. (2020), southwestern North America may be at the beginning of a megadrought that is comparable to the worst megadroughts that have occurred since 800 CE. The major difference between future droughts and droughts during the reference period (Little Ice Age) is that global warming will lead to a prevalence of hot droughts and increased water stress that will affect ecosystems and vegetation more profoundly than droughts occurring under more moderate temperatures (Diffenbaugh et al. 2015, Overpeck 2013). Past changes in the structure and composition of forests in response to cycles of prolonged drought periods during the Holocene provide some insight into how forests in the assessment area might shift in the future. Moreover, the effects of future droughts will likely be further intensified due to land use change and human demand for water.

Past changes in the structure and composition of forests in response to cycles of prolonged drought periods during the Holocene provide some insight into how forests in the assessment area might shift in the future.

According to Micheli et al. (2018), average winter temperatures in the assessment area are projected to increase by 4 to 6 °C and summer temperatures are projected to increase by 3 to 5 °C by the end of the century (2070–2099).

Precipitation projections, on the other hand, are highly variable, with disagreement as to whether trends will be positive or negative. Whatever the case, the ratio of rain to snow is projected to increase significantly, and snow will melt earlier, a product of overall warming temperatures (Micheli et al. 2018). Although there is uncertainty regarding precipitation levels, it is generally believed that there will be more interannual precipitation variability in the assessment area, with wet and dry cycles lasting longer (Butz et al. 2015, Swain et al. 2018).

Longer, hotter, and drier fire seasons have been projected for the Pacific Northwest (including the assessment area), most likely leading to continued increases in the area burned by wildfire (Wimberly and Liu 2014). The susceptibility of dense, mixed-conifer forests to stand-replacing fire—which is especially prevalent during drought conditions—increases the risk of type-conversion to shrubfields as a result of a lack of available seed sources in large severely burned patches combined with highly competitive fire-stimulated shrub species (Tepley et al. 2017). Further, Wahl et al. (2019) have shown that the longer fire seasons in concert with high fuel accumulations and increased landscape connectivity have largely decoupled fire activity from the amount of annual precipitation, making any year a potential major fire year (e.g., the wet years of 2006 and 2017). Reducing tree densities in YPMC forests through management actions, such as thinning and prescribed fire, may reduce the probability of drought-induced mortality (van Mantgem et al. 2016) and the risk of stand-replacing wildfire (Safford et al. 2012b). However, changes in climate might affect management's ability to carry out these treatments because of the potential for reduced burn windows in the future (Wimberly and Liu 2014).

Extreme precipitation—

NRV and comparison to current—Precipitation in California is extremely variable from year to year and is driven primarily by large storms that occur over a handful of wet days per year (ranging from 5 to 15 days). Large multi-day storm events called atmospheric rivers account for 20 to 50 percent of California's precipitation (Dettinger et al. 2011). Located in the wettest part of the state, the assessment area exhibits a smaller coefficient of variation in annual precipitation and a higher average number of wet days than the rest of California. Despite generally more abundant and less variable precipitation in the assessment area, interannual variation in precipitation is still high compared to the rest of the United States (especially as you move away from the coast), and the area remains dependent on relatively few, large precipitation events each year.

It is impossible to assess differences or similarities between presettlement (pre-1850s) and current extreme precipitation events in the assessment area due to a lack of direct evidence. Most documentation of extreme precipitation events prior to

Euro-American colonial settlement in the area is focused on large flooding events that destroyed human infrastructure. The following two quotes were taken from Wells (1881: 40–41) in *History of Siskiyou County*:

> When a heavy rain continues for several days without abating, the streams are unable to carry the water that runs so rapidly down the mountains into the valleys and cañons. The creeks and rivers overflow their banks and mountain torrents rush through gulch and cañon, to collect and form a lake in every valley towards which they run.…These floods now do considerable damage to the crops and farms in the valleys and to mining claims along the rivers; but previous to the advent of white men, there being no improvements of this character to suffer, it was these floods that by their alluvial deposits fertilized and prepared these valleys for the plow.…The earliest information we have of a flood exists in the traditions of the savages, who say that years ago there was a terrible flood in which thousands of natives lost their lives, and hundreds of Rancherias on the banks of rivers were washed away and destroyed. It is an era in their history from which they date events in the Sacramento valley, and occurred in the beginning of the present century, about the year 1805.

There were several more years between 1805 and 1881 with notable precipitation events, according to Wells (1881: 41), all of which were noted most often for the resulting floods rather than the precipitation itself:

> The annals of the Hudson Bay Company also show that the year 1818 was one of excessive storms and tremendous floods.…The winter of 1852–53 was a disastrous one throughout the whole State. The great Sacramento valley was one great sea of water.…The winter of 1861–62 was one that will long be remembered in California, for its devastating floods, that came pouring down from the mountains, sweeping everything before them and leaving ruin, and desolation in their pathway.

The major precipitation events that drive annual precipitation totals in California have a very large effect on streamflow and vulnerability to landslides. The North Coast Range and the Klamath Mountains, with their steep terrain and geologically unstable nature, are highly susceptible to earth movements associated with large precipitation events (Zhao et al. 2012), especially rain-on-snow events. This is especially true when high rainfall events follow other disturbances, such as severe wildfire (DeGraff et al. 2015).

Extreme precipitation events have also been linked with ending persistent droughts along the west coast of the United States. While droughts tend to begin gradually with accumulating water deficits, they have a tendency to end rather abruptly, usually as a result of a few concentrated large storms. Dettinger (2013) found that about 33 to 40 percent of all persistent droughts in California ended because of atmospheric river storms. In northern California and northward, atmospheric rivers during October through March (the wet season) were the reason for ending 60 to 67 percent of the recorded drought periods between 1950 and 2010 (Dettinger 2013).

Future—Precipitation has been modeled under a number of future climate change scenarios, and there is general agreement that future extreme precipitation events are either going to remain largely unchanged or will somewhat increase in number and magnitude (Bell et al. 2004, Cayan et al. 2008, Dettinger et al. 2011). Dettinger et al. (2011), however, projected increases in many extremes, such as the frequency of larger than average atmospheric rivers, the number of years with multiple atmospheric rivers, and the temperature associated with atmospheric river storms. Large flood events have also been projected to increase for multiple areas adjacent to the assessment area (Das et al. 2013, Najafi and Moradkhani 2014), and it can be inferred that the assessment area will experience a similar increase.

Fire

In the simplest of terms, "A fire regime is a generalized description of the role fire plays in an ecosystem" (Agee 1993). The key characteristics used to describe a fire regime include frequency, rotation, magnitude (intensity or severity), extent, and seasonality (Agee 1993, van Wagtendonk et al. 2018). Throughout the assessment area, fire regimes are driven by climate, geology, and topography and their relationships with vegetation composition, structure, and the likelihood of ignition. In YPMC forests, historical fire regimes were typified by frequent fires of low severity because fires occurred frequently enough to impede the buildup of fuels that would support fires of higher severity over large areas. The drought season characteristic of Mediterranean climates typically results in hot, dry weather and fuel conditions conducive to fire each year (Steel et al. 2015). As elevation increases, fuel moisture gradually becomes a more important driver of the fire regime, limiting fuel availability and effectively shortening the fire season (Agee 1993, Miller and Urban 1999).

Fire regime classifications are useful for comparing the role and importance of fire in different ecosystems. Most of these classifications focus on fire frequency and severity because they are relatively easy to measure, and a rich literature exists that describes their ecological importance. Using the Schmidt et al. (2002) classification, presettlement YPMC forests in the assessment area were primarily in Fire Regime I: frequent (0–35-year fire return intervals [FRIs]), mostly low- to moderate-severity fires that served as a regular ecological process to maintain the dominance of fire-tolerant species and generally open-canopied conditions. Today, as a result of human influences since Euro-American settlement (e.g., fire suppression, logging, grazing), YPMC forests in the assessment area have largely transitioned to Fire Regimes III and IV: infrequent (35–200-year fire return intervals), moderate- to high-severity fires that now act as a major disturbances to the system (Safford and Van de Water 2014, Steel et al. 2015).

Below are brief descriptions of fire regime characteristics that will be discussed in greater detail in subsequent sections (Agee 1993, Heinselman 1973, Sugihara et al. 2018):

- *Frequency* is the number of times that fires occur in a particular area and time period. It is usually represented by its inverse, the **FRI**, which is the average number of years between fires.
- *Rotation* is the number of years necessary to burn an area equal to the total area being considered.
- *Severity* is the magnitude of the effect fire has on ecosystem components, such as vegetation and soil.
- *Extent* is the size of an individual fire, the size of different severity patches within a fire, or the annual area burned.
- *Seasonality* is the time of year when fire is most likely to occur.

Prior to Euro-American colonial settlement, lightning ignitions and American Indian burning were the primary causes of fire (Anderson 2005, Pyne 1982, Sawyer 2006, Show and Kotok 1924). Once gold was discovered in the assessment area in the 1850s, Euro-American populations increased rapidly, indigenous populations declined, and most recorded fires were assumed to have been started by settlers (Haefner 1912, Leiberg 1900). Following the decline of indigenous populations, fire activity often switched from frequent, more localized and primarily low-severity fires to less frequent, more extensive and more severe fires (Fry and Stephens 2006, Leiberg 1900, Metlen et al. 2018, Skinner et al. 2009, Taylor et al. 2016). Fire suppression policies were implemented in the early 1900s that are still largely in place today (Stephens and Ruth 2005). Despite a continued focus on fire suppression, policy authorizing the management of wildfires for resource benefit has been in place since the late 1960s and early 1970s (van Wagtendonk 1991), and it is stated in a 1995 federal wildland fire management and policy review that "Wildland fire, as a critical natural process, must be reintroduced into the ecosystem" (USDI and USDA 1995). The elimination of almost all indigenous burning followed by efforts attempting to ubiquitously suppress fire on the

The elimination of almost all indigenous burning followed by efforts attempting to ubiquitously suppress fire on the landscape have changed the amount and type of fire in the assessment area.

landscape have changed the amount and type of fire in the assessment area. Due to the rugged landscape of much of the assessment area, the effectiveness of initial fire suppression efforts was limited to areas that were relatively easy to access (Atzet and Martin 1992, Taylor and Skinner 1998). It was not until the mid-1940s that the success of these efforts increased considerably because of expanding road systems, aerial support, and better resources (Atzet 1996, Pyne 1982). Figure 6 provides a generalized depiction of how frequency, seasonality, severity, and extent have shifted as a result of the elimination of indigenous burning and the onset of fire suppression. The following sections will go into greater depth describing how each fire regime component has been altered.

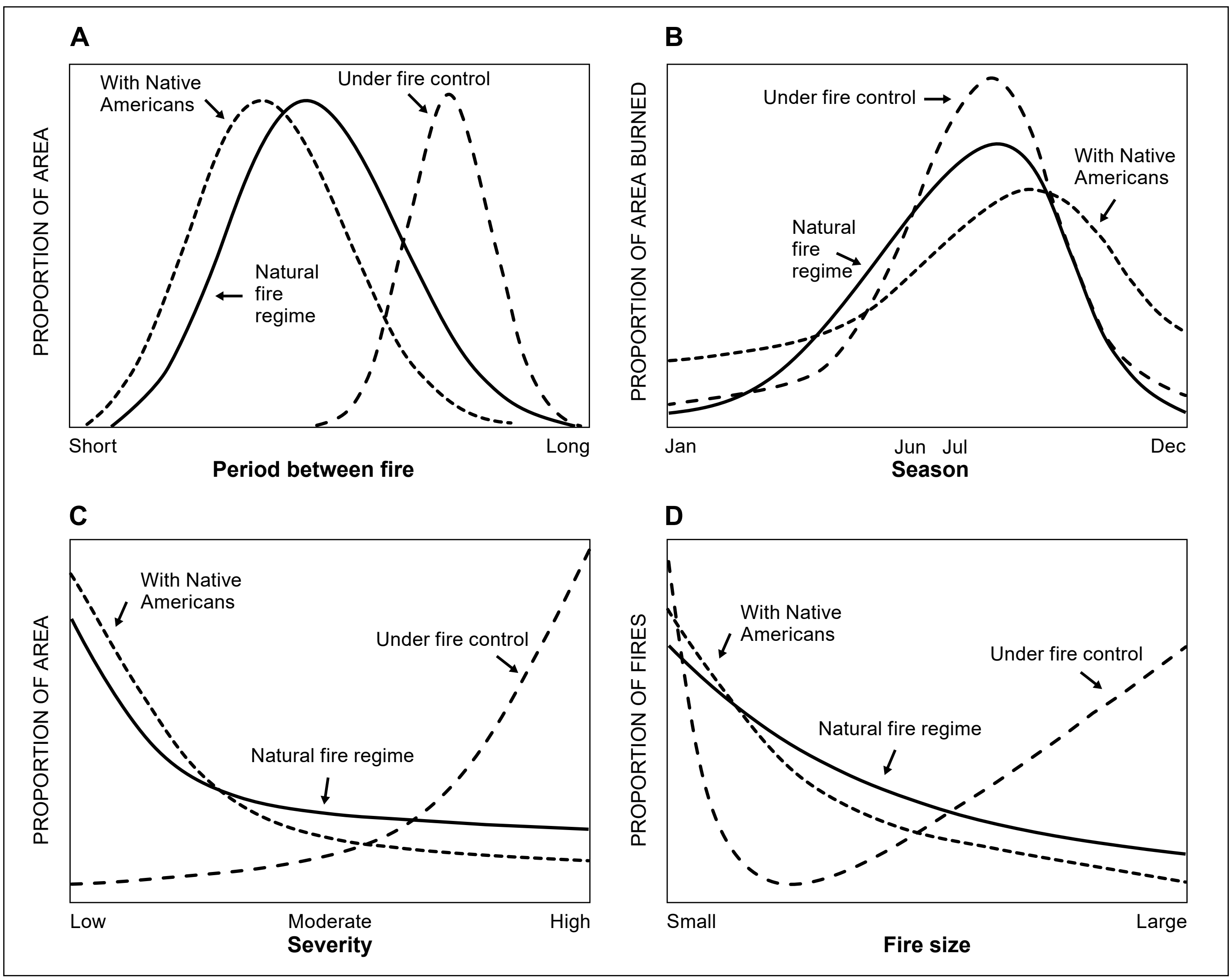


Figure 6—Generalized depiction of four different fire regime components: (A) frequency, (B) seasonality, (C) severity, and (D) extent, and how each differs between a natural fire regime, a fire regime with American Indian burning, and a fire regime under fire suppression. Adapted from Martin and Sapsis (1992); (C) and (D) have been modified to facilitate interpretation. Note that these plots have no scale and are meant to be descriptive, displaying the nature of observed trends as opposed to the exact degree of these trends.

Fire frequency—

NRV—YPMC forests during the presettlement era mostly supported frequent (median FRI <20 years), low- to moderate-severity fires (table 2). Exceptions to this general pattern where FRIs were longer and more variable include white fir-dominated stands at higher elevations, stands near the coast that experience higher atmospheric moisture (Agee 1991, Atzet and Martin 1992, Stuart and Salazar 2000, Thornburgh 1995), and Jeffrey pine stands at higher elevations or on serpentine soils that have low productivity and accumulate fuels very slowly (Atzet and Martin 1992, Skinner and Chang 1996).

Van de Water and Safford (2011) summarized published and unpublished literature to estimate FRIs for pre-Euro-American settlement fire regime types across California. The mean FRI for both yellow pine and dry mixed conifer was approximately 11 years, with a median of 7 years for yellow pine and 9 years for dry mixed conifer. Moist mixed conifer had slightly higher mean and median values of 16 and 12 years, respectively (table 2). There have been substantially fewer YPMC studies conducted in the assessment area compared to other parts of California; consequently, most of the studies contributing to the reference values provided by Van de Water and Safford (2011) come from other regions, particularly the Sierra Nevada. Presettlement fire regimes in northwestern California appear to have supported slightly longer FRIs than drier parts of the state, but FRIs for yellow pine and mixed conifer remain similar, with the greatest variability generally present in moister sites (table 2, Metlen et al. 2018, Van de Water and Safford 2011).

Due to the climatic influence of the Pacific Ocean and patterns of lightning strike density, fires tend to increase in frequency from west to east (Atzet and Wheeler 1982). For example, Stuart and Salazar (2000) reported a median FRI of 27 years with a range of 12 to 161 years in white fir-dominated and codominated sites in the western portion of the assessment area, while Skinner and Chang (1996) reported a median FRI of 11 years with a range of 4 to 56 years for white fir forests in the southern Cascades. Similarly, elevation and aspect have strong influences on fire frequency, with higher elevations experiencing longer return intervals due to cooler, wetter conditions (Agee 1991, Gill and Taylor 2009), and south- and west-facing slopes experiencing shorter median fire return FRIs than north- and east-facing slopes (Beaty and Taylor 2001, Taylor and Skinner 1998). In addition to climate and topography, substrate plays a role in the abundance of fire. On serpentine soils, the reduction in productivity leads to a much lower stand density and slower accumulation of fuels (Alexander et al. 2007, Atzet 1979, DeSiervo et al. 2015). The patchiness and rockiness of ultramafic sites limit the spread of fire,

Table 2—Historical fire return interval estimates for yellow pine and mixed-conifer forests by general location and vegetation type in the assessment area (see fig. 1b for a map of general locations and associated average annual precipitation)

General location	Vegetation type[a]	Time period	Fire return interval: Mean	Median	Min.	Max.	Reference
			Years				
California	Yellow pine	pre-1850s	11	7	5	40	Van de Water and Safford (2011); 24 sources[b]
California	Dry mixed conifer	pre-1850s	11	9	5	50	Van de Water and Safford (2011); 37 sources[b]
California	Moist mixed conifer	pre-1850s	16	12	5	80	Van de Water and Safford (2011); 53 sources[b]
Northern Klamath Mountains	Yellow pine	1680–1883	10	8	1	33	Metlen et al. (2018)
Northern Klamath Mountains	Dry mixed conifer	1703–2000	10	8	1	40	Metlen et al. (2018)
Northern Klamath Mountains	Mixed conifer	pre-1900s	—	—	8	24	Taylor and Skinner (1998)
Northern Klamath Mountains	Douglas-fir-mixed conifer	1700–1900	23	—	3	76	Sensenig et al. (2013)[c]
Northern Klamath Mountains	Douglas-fir-mixed conifer	1626–1849	—	14.5	6	116	Taylor and Skinner (1998)
Northern Klamath Mountains	Moist mixed conifer	1649–1945	10	9	1	37	Metlen et al. (2018)
Northern Klamath Mountains	White fir	—	25	—	—	—	Atzet and Martin (1992)
Northern Klamath Mountains	Jeffrey pine (serpentine)	—	50	—	—	—	Atzet and Martin (1992)
Western Klamath Mountains	Moist mixed conifer	1614–1944	45	16	15.5	104	Stuart and Salazar (2000)
Western Klamath Mountains	White fir	1614–1944	49	40	12	82	Stuart and Salazar (2000)
Central Klamath Mountains	Mixed conifer	1750–1998	7.5	8	1	15	Unpublished data[d]
Central Klamath Mountains	White fir-enriched mixed conifer	pre-1920	29	—	—	—	Thornburgh (1995)
South-central Klamath Mountains	Canyon live oak-mixed conifer	pre-1850s	—	11	7	33	Taylor and Skinner (2003)
South-central Klamath Mountains	Ponderosa pine-mixed conifer	1628–1995	—	11.5	9	22	Taylor and Skinner (2003)
South-central Klamath Mountains	Douglas-fir-mixed conifer	pre-1850s	—	15	3	59	Taylor and Skinner (2003)
South-central Klamath Mountains	Douglas-fir-mixed conifer	1628–1995	—	13	7	61.5	Taylor and Skinner (2003)
Southeastern Klamath Mountains	Ponderosa pine-mixed conifer	pre-1850s	—	11	5	46	Skinner (2003b)
Southeastern Klamath Mountains	Ponderosa pine-mixed conifer	1750–1849	3	2.5	—	—	Fry and Stephens (2006)
Southeastern Klamath Mountains	Dry mixed conifer	1700–2002	10.6	7.8	3	38	Unpublished data[d]

General location	Vegetation type[a]	Time period	Fire return interval Mean	Median	Min.	Max.	Reference
			Years				
Southeastern Klamath Mountains	Mixed conifer	1583–1969	14	10	4	111	Unpublished data[d]
Southeastern Klamath Mountains	Douglas-fir-mixed conifer	pre-1850s	—	14	3	52	Skinner (2003b)
Southeastern Klamath Mountains	Jeffrey pine-white fir	pre-1850s	—	12	4	96	Skinner (2003a)
Southeastern Klamath Mountains	Klamath enriched mixed conifer	1749–1924	—	7	3	44	Skinner (2002)
Southeastern Klamath Mountains	Klamath enriched mixed conifer	1710–1916	—	8	4	64	Skinner (2002)
Northern California Coast Range	Dry mixed conifer	1700–1900	8	6	1	49	Skinner et al. (2009)
Northern California Coast Range	Moist mixed conifer	1700–1900	16	11	3	38	Skinner et al. (2009)
Southern Cascades	Ponderosa pine	—	—	16	8	32	Olson (1994)
Southern Cascades	Dry mixed conifer	1748–1910	8	7	2	29	Metlen et al. (2018)
Southern Cascades	Mixed conifer	pre-1850s	7.7	8.5	2	15	Beaty and Taylor (2001)
Southern Cascades	Mixed conifer	1600–1998	12	11	5	41	Unpublished data[d]
Southern Cascades	Douglas-fir-mixed conifer	1700–1900	17	—	6	50	Sensenig et al. (2013)[c]
Southern Cascades	White fir-mixed conifer	pre-1850s	—	11	3	29	McNeil and Zobel (1980)
Southern Cascades	White fir-mixed conifer	pre-1850s	—	13	5	24	Skinner and Chang (1996)
Southern Cascades	White fir-mixed conifer	pre-1850s	—	10	4	26	Skinner and Chang (1996)
Southern Cascades	White fir-sugar pine	pre-1850s	11.3	—	—	—	Bekker and Taylor (2001)
Southern Cascades	White fir-Jeffrey pine	pre-1850s	5.8	—	—	—	Bekker and Taylor (2001)
Southern Cascades	White fir	pre-1850s	—	16	4	39	McNeil and Zobel (1980)
Southern Cascades	White fir	pre-1850s	—	11	4	56	Skinner and Chang (1996)

Rows in grey include results from the Van de Water and Safford (2011) literature review of fire return interval estimates across all of California and include many of the referenced studies. Individual studies are provided on their own to provide detail on fire return interval values specifically from the assessment area. See figure 1 for a map of general locations and associated average annual precipitation.

[a]Vegetation type generalized from information provided in reference.

[b]Van de Water and Safford's sources include all but one of the pre-2011 references listed below them (unpublished data not included).

[c]Values developed from field-counting xylem rings; cross-dating did not occur and level of precision is ±6 years.

[d]Unpublished data available on file at USDA Forest Service, Pacific Southwest Research Station, Redding, California.

— = Value not provided in reference.

with the most continuous fuels being grasses and occasionally shrubs (Atzet and Wheeler 1984, Jimerson et al. 1995, Whittaker 1960). Median FRIs for 10 Jeffrey pine sites on ultramafic soils in YPMC near Hayfork were found to range from 8 to 15 years, whereas FRIs for 7 sites in upper montane near Mount Eddy ranged from 8 to 30 years (Skinner et al. 2018).

Comparison to current—Numerous studies have noted a large drop in fire frequency in the assessment area during the 20th century (e.g., Atzet and Martin 1992; Fry and Stephens 2006; Metlen et al. 2018; Skinner 2003b; Skinner et al. 2009; Stuart and Salazar 2000; Taylor and Skinner 1998, 2003). About 75 percent of YPMC forests in California have not experienced a single fire since the beginning of the 20th century (Steel et al. 2015). There are currently very large departures from both the mean and median presettlement FRIs for most of the assessment area (Safford and Van de Water 2014). There is some variation, however, with the central Klamath Mountains showing a low to moderate departure and the eastern Klamath Mountains showing a high departure. Notably, the Shasta-Trinity National Forest in the assessment area has more than 500 000 ha (more than 1.2 million ac) of land where current FRIs are more than three times the presettlement FRI. Of the presettlement fire regimes considered, forests and woodlands at lower elevations show the most extreme departures, with the magnitude of departure decreasing as elevation, precipitation, and snow-to-rain ratio increase, and as temperature decreases (Safford and Van de Water 2014). Despite departures being generally lower at higher elevations, fire in upper montane areas in the Klamath Mountains were historically quite frequent and studies have shown departures to be present in these areas as well (Skinner 2003a, Whitlock et al. 2004).

Despite observed contemporary increases, current fire frequencies in the assessment area are still far below presettlement frequencies.

Throughout the Holocene, variations in climate were a major factor in driving fire regime variation. Studies using charcoal accumulation in lake sediments have shown fire frequency to track regional changes in precipitation and temperature (Whitlock et al. 2003). Warm, dry periods through the Holocene were associated with more frequent fire in the region (Colombaroli and Gavin 2010, Daniels et al. 2005, Mohr et al. 2000). Today, as the climate warms, we are already seeing increases in fire frequency associated with extended fire seasons (Westerling 2016) and, in northwestern California, the number of fires > 40 ha has increased since the 1970s (Miller et al. 2012b). Despite observed contemporary increases, current fire frequencies in the assessment area are still far below presettlement frequencies.

Future—With more than 85 percent of Forest Service lands in the northwest California portion of the assessment area burning less frequently now than under presettlement conditions (Safford and Van de Water 2014), forests in the area are on a trajectory toward continued densification, high dead fuel availability, and increased vulnerability to drought, insects, and severe wildfire. As the climate continues to warm, fire season duration is likely to continue to increase, escalating the prevalence of fire on the landscape. Large wildfire activity has already shown marked increases associated with earlier spring onsets (Westerling 2016), and it is likely that this trend will continue. According to Yang et al. (2015), both human- and lightning-caused fire frequencies are predicted to increase (170 percent of the current level under the low-emissions scenario and up to 310 percent in the high-emissions scenario) as the climate becomes warmer and drier, with the proportion of lightning-caused wildfires predicted to become much higher compared to human-caused wildfire occurrences. Using a number of climate scenarios to model future (2070–2099) risk of large fires, Westerling and Bryant (2008) project the probability of large fires (i.e., greater than 200 ha) in northern California to increase by +15 to +90 percent as temperatures increase. Loudermilk at al. (2013) similarly projected increased wildfire activity associated with temperature increases under a high-emissions scenario. The number of large and severe wildfires may continue to increase regardless of precipitation patterns until fuels become limiting (Wahl et al. 2019), yet future frequency of fires burning at low to moderate severity will depend on the ability of managers to reduce fuels on the landscape, for example, incorporating fire into land management across the landscape (i.e., prescribed fire, managed wildfire), or expanding the footprint of mechanical treatments.

Fire rotation—

NRV—Stephens et al. (2007b) provide a pre-Euro-American colonial settlement fire rotation estimate of 27 years for mixed-conifer forests across California. Similarly, Mallek et al. (2013) summarized presettlement fire rotations for major forest types throughout the Sierra Nevada and southern Cascades and found an overall average of about 28 years. When broken out by forest type, the authors found that yellow pine and dry mixed-conifer forests had similar mean rotations with 22 and 23 years (range of means: 11–34 years). Moist mixed-conifer forests had a longer rotation, with a mean rotation of 31 years (between 15 and 70 years) (Mallek et al. 2013). Studies based in the Klamath Mountains found presettlement fire rotations for mixed-conifer forests (mostly Douglas-fir dominated) to be about 19 years (Taylor and Skinner 1998, 2003). Table 3 provides presettlement fire rotation estimates for YPMC forests in and near the assessment area.

Comparison to current—Fire rotations, which are strongly linked to fire return intervals, are well outside the NRV for the assessment area, despite decreases in rotation lengths over the past two decades (Miller et al. 2012b; Taylor and Skinner

Table 3—Historical fire rotation estimates for yellow pine and mixed-conifer forests

Location	Forest type	Fire rotation[a]	Reference
		----- *Years* -----	
Sierra Nevada and southern Cascades, California	Yellow pine	22 (11–34)	Mallek et al. (2013); 9 sources[b]
Sierra Nevada and southern Cascades, California	Dry mixed conifer	23 (11–34)	Mallek et al. (2013); 8 sources[b]
Sierra Nevada and southern Cascades, California	Moist mixed conifer	31 (15–70)	Mallek et al. (2013); 12 sources[b]
Thompson Ridge, Klamath Mountains, California	Douglas-fir/mixed conifer	19	Taylor and Skinner (1998)
Near Hayfork, south-central Klamath Mountains, California	Douglas-fir/mixed conifer	19	Taylor and Skinner (2003)
Cub Creek Research Natural Area, southern Cascades, northern California	Mixed conifer	28.2	Beaty and Taylor (2001)
Thousand Lakes Wilderness, southern Cascades, northern California	White fir-sugar pine	34	Bekker and Taylor (2001)[c]
Thousand Lakes Wilderness, southern Cascades, northern California	White fir-Jeffrey pine	22	Bekker and Taylor (2001)[c]

Rows in grey include results from the Mallek et al. (2013) literature review of fire rotation estimates in the Sierra Nevada and southern Cascades and include the individual referenced studies. Individual studies are provided on their own to provide detail on fire rotation values specifically from the assessment area.

[a]Fire rotation is the length of time necessary to burn an area equal to the area of interest. In the Mallek et al. (2013) estimates, the number given is the mean of the sources used (see next column); the numbers in parentheses represent the range in values.

[b]Mallek et al. (2013) sources include the references listed below them.

[c]The values reported here are likely conservative estimates of fire occurrence because samples were taken from wedges cut from standing trees.

1998, 2003). Miller et al. (2012b) found that regional fire rotation in the assessment area peaked at 974 years in 1984 and decreased to 95 years by 2008 as a result of recent, large fire events in the area. Years with extensive fire have occurred since 2012 and the rotation has probably decreased further since then. In a localized study, Taylor and Skinner (2003) found substantial increases in fire rotations between the presettlement period and suppression period (19 vs. 238 years, respectively) for mixed-conifer forests in the Klamath Mountains. At another locality, Taylor and Skinner (1998) also found a shorter fire rotation during the presettlement period than the suppression period in largely Douglas-fir-dominated, mixed-conifer forests, but the difference was minimal (about 19 compared with 25.5 years, respectively). The relatively shorter fire rotation found during the suppression period in Taylor and Skinner (1998) compared with other studies may be driven by lightning-ignited fires that burned in and around their study area, especially in 1987 (Reider 1988).

Future—As the frequency and risk of large fires associated with high fuel levels and longer fire seasons continue to increase (Miller et al. 2012b, Wahl et al. 2019, Westerling and Bryant 2008, Westerling et al. 2006), fire rotations will likely decrease, a trend already apparent over the past few decades in the assessment area (Miller et al. 2012b). It is possible that, taking into account current wildfire trends, the increasing tendency to manage wildfire for resource benefit, and prescribed fire trends, fire rotations may more closely approximate the NRV by the end of the century. The circumstances under which future fires burn, however, will determine whether they will be ecologically beneficial and help achieve management objectives (i.e., reduce forest densities, promote heterogeneity) or cause negative ecological effects (i.e., increase already high percentages of stand-replacing fire) and more permanently shift vegetation from forest to shrubland (Coppoletta et al. 2016, Lauvaux et al. 2016, Tepley et al. 2017).

Fire severity—

Vegetation burn severity thresholds have often been grouped into three or four categories based on percentage of overstory mortality (e.g., <20 percent, 20–70 percent, >70 percent [Agee 1993] or <25 percent, 25–50 percent, 51–75 percent, >75 percent [Weatherspoon and Skinner 1995]). Remotely sensed fire severity assessments often use a measure based on the normalized burn ratio (NBR). The relativized differenced NBR (RdNBR), based on differences in LANDSAT-TM pre- and postfire images (Miller and Thode 2007), has become a standard fire-severity measure in the western portion of the United States. The Forest Service Pacific Southwest Region fire-severity monitoring program, based on RdNBR calibrated to field-based measures of forest biomass loss (Miller et al. 2009a), defines stand-replacing fire as "high severity," where >90–95 percent of basal area or canopy cover is killed by fire (Miller et al. 2009a, 2012a; Miller and Safford 2012). Various subcategories (e.g., low-moderate severity, high-moderate severity, very high severity) can also be identified (e.g., Welch et al. 2016). In summary, anything with <20–25 percent overstory mortality is considered low severity, leaving moderate severity to represent the large class between 20/25 percent mortality and the upper threshold, whatever it may be.

When discussing fire severity, it is important to clearly distinguish between severity and intensity. Severity specifically refers to the magnitude of effect fire has on the environment and is most commonly used to describe effects on the soil or vegetation. Fire severity is a result of fireline intensity, residence time, and moisture conditions at the time of burning, as well as the fire adaptations of the species affected. Intensity, on the other hand, relates to energy released by the fire without regard to the effect fire has on the ecosystem (Keeley 2009).

NRV—Prior to Euro-American colonial settlement, fire severity in assessment area YPMC forests has typically been described as being predominantly low to moderate severity, or sometimes "mixed" severity (Agee 1991; Fry and Stephens 2006; Leiberg 1900; Odion et al. 2004; Skinner et al. 2009; Taylor and Skinner 1998, 2003). Unfortunately, it is difficult to reconstruct presettlement fire-severity patterns, especially in areas that have experienced extensive management. Estimates are based on a variety of information sources, including modern burning patterns in "reference stands" (relatively undegraded forest stands that are thought to have changed little since Euro-American colonial settlement); historical accounts from early settlers, managers, and scientists (e.g., Leiberg 1900; Show and Kotok 1924, 1929); and documented or inferred changes over time in forest structure (e.g., Leonzo and Keyes 2010). The latter is based on assessing the presence or absence of multiple age classes, with the idea that stands containing multiple age classes (especially those with older trees) have experienced low- to moderate-severity fire, and stands containing single age-class cohorts most likely experienced moderate- to high-severity fire. This assumption may not always hold true, however, and even-aged stands can develop through other means. For example, the result of a good cone crop in the absence of fire in an otherwise open, frequent-fire forest can lead to the establishment of a largely even-aged stand (Brown 2006, Brown et al. 2008, Cooper and Kelleter 1907, Show and Kotok 1924, Skinner et al. 2018, Taylor and Skinner 1998). Obviously, all of these methods incorporate uncertainty. Nonetheless, general agreement exists that YPMC fires were dominated by low-severity fire effects, with the moderate- and high-severity components increasing in importance with moisture, elevation, stand density, and the presence of fire-intolerant species, among other things.

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"Mixed-severity" fire is an unfortunate term, as all fires burn with a mix of severities and, to this point, no one has clearly identified what exact mix of severities qualifies a fire as mixed severity. The term "mixed-severity" encompasses highly variable conditions and is very scale dependent (Brown et al. 2008; Halofsky et al. 2011; Hessburg et al. 2007, 2016; Perry et al. 2011; Skinner et al. 2018). Halofsky et al. (2011) suggest that mixed-severity fires exist only at a particular spatial scale between individual trees and large watersheds—the scale of a forest stand for example—and occur where both weather and fuels exert effects on fire behavior. However, fire behavior in all fires is driven by a mix of weather and fuels. The confusion in understanding what a mixed-severity fire actually is leads to difficulties in comparing historical and current severity patterns, and it becomes necessary to be very specific regarding one's spatial references. Brown et al. (2008: 1,995) provide a good example of how the lack of specificity regarding spatial extents leads to misinterpretation:

> …[W]ithout reference to scale it is possible to conclude that recent variable-severity fires in ponderosa pine forests (i.e., that have included both surface burning as well as large areas of crown mortality) are within a historical range of variability even though areas of crown mortality are orders of magnitude larger than any area that occurred historically.

Basically, the issue seems to be one of spatial scale. Fires either kill trees or don't kill them, and when the spatial pattern of tree mortality is finer than the minimum mapping unit, we get "mixed-severity" fire. As such, the existence of mixed-severity fire is dependent on the resolution of one's mapping.

Note that imprecise use of language is partly to blame for this confusion. For example, Skinner and Chang (1996), referring to the Sierra Nevada and southern Cascades, suggested that fires that "…once varied considerably in severity, are now almost exclusively high-severity…." However, the fires they were referring to exhibited high spatial variability in severity. The change they pointed to was a large increase in the area of the high-severity component, but most modern high-severity fires are still primarily low and moderate severity (see fig. 6 in Safford and Stevens 2017). On the other side, Odion et al. (2014) argued that "a fire regime with a high-severity component **of any amount** should not be classified as low/moderate severity." Stevens et al. (2016) pointed out that this was both inconsistent with the science and illogical because by extension, high-severity fire could not exist where even a single tree survived, and low-severity could not exist where even a single tree was killed. Such a situation would result in all fires being called mixed severity, which underscores how the term is confusing and not useful. For these reasons, scientists and managers rely on fire severity classifications that permit certain levels of mortality in areas of low and moderate severity and permit a certain level of survival in areas of moderate and high severity. Because of the diverse vegetation conditions, variable weather patterns, and complex topography in much of the assessment area, the spatial complexity of fires tends to be high, especially in its western portion (Halofsky et al. 2011).

Topography is a key driver of fire severity patterns both directly and indirectly. Fires tend to burn at lower intensities on lower slopes and north- and east-facing aspects, and higher intensities on mid and upper slope positions and south- and west-facing aspects (Alexander et al. 2006, Taylor and Skinner 1998). However, there have been some cases where severity was higher on east-facing slopes, likely due to vegetation composition or other variables rather than the direct effect of

In the highly dissected Klamath Mountains, temperature inversions are also common and can have an important effect on fire severity patterns.

aspect on fire behavior (Estes et al. 2017, Weatherspoon and Skinner 1995). In the highly dissected Klamath Mountains, temperature inversions are also common and can have an important effect on severity patterns. When smoke is trapped in valleys and canyons, temperatures drop and humidity increases at lower slope positions (Robock 1988, Skinner et al. 2018, Weatherspoon and Skinner 1995), which results in lower severity (Estes et al. 2017, Miller et al. 2012b). Severity patterns are also driven by weather in many cases. Strong, drying winds from the northeast and maritime influences from the west can at times overpower the effect of aspect on fire severity (Thompson and Spies 2009).

Vegetation composition and structure have also been identified as strong drivers of fire severity patterns. Areas of high severity tend to be prevalent in areas with high shrub or knobcone pine (*Pinus attenuata* Lemmon) cover (Estes et al. 2017, Show and Kotok 1924, Skinner and Taylor 2018, Thompson and Spies 2009), whereas hardwood tree litter often dampens fire behavior (and hardwoods resprout after fire, which adds a temporal uncertainty to fire-severity estimation). Higher severity has also been associated with areas having higher densities of small trees (Miller et al. 2012b, Thompson and Spies 2009). Yet, there are areas with active fire regimes that have maintained a matrix where small trees (10–30 cm diameter at breast height [dbh]) make up a large proportion of the total stem density (up to 66 percent) with no evidence of high-severity fire, which is due to the patchy nature of frequent surface fires (Knapp et al. 2013). Regardless of small tree densities, resulting fire effects ultimately depend upon the conditions under which a fire burns and the continuity of the fuels and vegetation that are present.

Taylor and Skinner (2003) used fire scars and tree age classes to determine presettlement fire severity patterns in a 2325 ha watershed near Hayfork on the Shasta-Trinity National Forest. With almost all stands exhibiting multiple age classes and containing stems >250 years old (including many with older white firs), the authors concluded that low- to moderate-severity fires were the dominant fire type on the landscape, but were patchy enough within stands that fire-sensitive white fir could persist in the forest long enough to reach a fire-resistant size. They noted, however, that multi-aged stands were not uniform across the landscape and that patches of even-aged stands—often interpreted as a indicator of high-severity fire——were present historically (Taylor and Skinner 1998, 2003). The interpretation of even-aged stands in this way, however, may inflate estimates of historical proportions of high-severity fire (Skinner et al. 2018). Climate anomalies (i.e., periods of cool, wet conditions) that cause a hiatus in fire activity may allow for patches of even-aged stands to develop (Brown 2006, Brown et al. 2008, Skinner et al. 2018). Fire suppression likely had a similar effect, allowing gaps and large openings to fill in with cohorts of seedlings in the absence of fire (Skinner et al. 2018).

In some cases early observers used similar logic to that of today's researchers to estimate fire frequency and severity of past fire regimes. Referring to the Cascade Range and the Ashland Forest Reserve in and adjacent to the assessment area, Leiberg (1900: 277) stated the following:

> 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 [Douglas-fir], noble fir, white pine, alpine 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 white man. But, on the other hand, the great diversity in the age of such stands…show[s] clearly their origin as reforestations after fires, [and] proves that the fires during the Indian occupancy were not of such frequent occurrence nor of such magnitude as they have been since the advent of the white man.

Leiberg's mention of "the great diversity" of age classes in stands that contain "veteran trees" of less fire-resistant species corroborates the findings of Taylor and Skinner (2003) in the Klamath Mountains and further underlines the importance of patchiness (and in-patch variation) of low- to moderate-severity fires across the landscape.

Areas in drier yellow pine forests historically experienced primarily low-severity fires with relatively limited overstory mortality. According to Cooper and Kelleter (1907), fire's "chief effect upon the [yellow pine] forest is the destruction of brush and litter" in the understory, but "[o]penings thus made in the forest [by fire] are effectually prevented by subsequent fires from coming up to young growth, while chaparral…takes possession of the ground." Similarly, Munger (1917) wrote of the "[l]ight, slowly spreading fires that form a blaze not more than 2 or 3 feet high and that burn chiefly the dry grass, needles, and underbrush start freely in yellow-pine forests…"

According to Leiberg (1900), of the acreage that was observed as "fire marked" (2,975,000 ac) in the southern Cascade Range and Ashland Reserves at the turn of the century (about 50 years after Euro-American colonial settlement began), about 20 percent (587,000 ac) of stands with mill timber (including yellow pine and mixed-conifer species as well as higher elevation species, such as true fir and mountain hemlock) burned with >75 percent overstory mortality. Yellow pine-dominated forests covered almost two-thirds of the forested area included in these observations, while Douglas-fir-dominated types covered about one-third, and higher elevation forest types covered the remainder (~11 percent). Leiberg (1900: 289) noted that, “The amount of damage to the mill timber in a forest stand in this region which may be wrought by a fire varies considerably. It may run as low as 1 per cent in stands of yellow pine, or it may rise so high in stands of mixed growth that it practically amounts to total destruction.” It is important to keep in mind that trees with fire scars on their trunk (also called catfaces) were considered damaged due to the loss in value of the fire-scarred area; such trees were rarely dead or dying. Leiberg (1900) also noted that pockets of dense Douglas-fir found throughout yellow pine forests seemed to burn at higher severities than the surrounding more open areas.

Taylor and Skinner (1998) estimated that between 1850 and 1950, 14 percent of their study area at Thompson Ridge (15 miles north of Happy Camp, near the Oregon-California border) in the Klamath Mountains burned at high severity, while 27 percent burned at moderate severity and 59 percent burned at low severity. The authors also broke down proportions burning at different severities by slope position. Lower slopes experienced the lowest percentage (12 percent) of high-severity fire and the highest percentage (75 percent) of low-severity fire, with 13 percent exhibiting moderate-severity fire. Severity on the upper slopes, on the other hand, was more evenly distributed among classes with 31 percent high severity, 32 percent moderate severity, and 37 percent low severity. Mid-slopes fell in between the two, with 15 percent high severity, 27 percent moderate severity, and 58 percent low severity. Taylor and Skinner (1998) studied an area consisting of a variety of forest types, dominated by Douglas-fir, but with components of mixed conifer, white fir, and hardwood. Additionally, the study period spans the colonial settlement period and the beginning of the fire suppression period, so the data collected certainly include impacts from Euro-Americans and early management policies.

Comparison to current—Since 1910, the 6 years with the largest area burned within the assessment area (1987, 1999, 2006, 2008, 2017, 2020) all experienced widespread lightning events that quickly exceeded the capacity for suppression efforts. These large fire events burned during (mostly) moderate weather conditions for extended periods of time (weeks to months), with some of them burning well into the fall (Estes et al. 2017, Miller et al. 2012b). Because lighting ignitions are typically accompanied by higher humidity and often summer precipitation, they usually begin as less intense, slower spreading fires (Miller et al. 2012b, Skinner et al. 2018). In addition to moderate burning conditions, temperature inversions, which reduce temperatures and increase humidity on lower slopes and in valleys, played a very large role in the severity patterns observed in the years with the highest total area burned between 1987 and 2008 (Estes et al. 2017, Miller et al. 2012b, Robock 1988). Miller et al. (2012b) found that, between 1987 and 2008, the annual mean percentage of fire area burning at high severity was 25 percent (median = 20 percent). When summed across their 22-year study period, 16 percent of the total mapped burned forested area burned at high severity. Modern fires that burn under "moderate" conditions (i.e., absence of extreme fire weather) tend to burn with similar proportions of low, moderate, and high severity to historical fires, especially when temperature inversions are present (Estes et al. 2017, Miller et al. 2012b, Taylor and Skinner 1998).

The proportion of high-severity fire is generally higher when there is less total annual area burned across the region, or when wildfires are human caused.

Furthermore, Miller et al. (2012b) found that the proportion of high-severity fire is generally higher when there is less total annual area burned across the region, or when wildfires are human caused. Recently there have been a number of fires with large proportions of high severity that burned under more extreme burning conditions and were started by humans (i.e., Klamathon, Carr, Delta, Hirz, and Zogg Fires).

Future—Restaino and Safford (2018) summarized future projections for fire severity across California. In all cases, the studies they reviewed expected fire severity and intensity to increase over time, with the partial exception of Lenihan et al. (2003), whose study projected an ultimate decrease in fireline intensity in the Klamath and North Coast regions (after an initial increase) because very large increases in burned area were projected to remove much of the woody biomass. In a study looking at multiple fires that burned in the assessment area in 2006, fire severity was reduced only when an area had burned three times previously (Estes et al. 2017). The highly dissected topography of the Klamath Mountains portion of the assessment area may act to help limit the size of future fires and subsequent size of high-severity patches, whereas the more regular north-south ridgeline systems of the North Coast Range may promote more fire spread (e.g., the Mendocino Complex and August Complex fires in 2018 and 2020, respectively). Topographic, vegetation, and fuel heterogeneity influence fire spread and spatial scale of fire effects. The limited accessibility and rugged terrain of much of the assessment area means that it is often difficult and costly to fight fire. Because we know that low- to moderate-severity effects can be expected under certain conditions—such as large lightning events that intersect with regional inversions, long-burning fires that sample more of the moderate weather conditions that characterize most of the burn season, and where strong topographic controls are present—it may be more cost-effective and ecologically beneficial to manage wildland fire when conditions permit. Future managed wildland fires could play a major role in reducing fuels and enhancing resilience to drought, fire, and other disturbances (Miller et al. 2012b, Skinner et al. 2018).

High-severity patch size—

NRV and comparison to current—All fire regimes (low, moderate, and high severity) include a component of stand-replacing fire, but differ in patch size characteristics (Agee 1998). In a table providing the relative landscape characteristics of western forest fire regimes, Agee (1998) described high-severity patch size within low-severity fire regimes as typically being of small size (~1 ha), with moderate-severity fire regimes containing medium patch sizes that can range anywhere from 1 to 300+ ha. There is a paucity of reference information on high-severity patch sizes in the assessment area, but work summarized in the Sierra Nevada NRV assessment (Safford and Stevens 2017) is germane. Safford and Stevens (2017: 51) wrote that "the NRV of high-severity patch size in assessment area YPMC forests was strongly dominated by a "salt and pepper" pattern of small areas mostly [much] less than a few hectares in size. Patches larger than a few hectares did occur, but they were rarely more than 100 ha. Nonetheless, such larger patches comprised perhaps half of the total high-severity area."

Miller et al. (2012b) carried out an assessment of fire trends in the California portion of the assessment area for the period 1987–2008. Average high-severity patch size was 11 ha (+/− 1.08 SE), but patch size actually dropped during the 22 years of assessment, from about 14 ha to about 8 ha (the year 1988 was a major driver of the trend, with an average high-severity patch size for that year of 23 ha). On the other hand, the maximum high-severity patch size in any year rose by more than four times, from a predicted average (from the regression line) of 124 ha in 1987 to 485 ha in 2008; the mean maximum high-severity patch size remained more or less steady at about 110 ha (Miller et al. 2012b). Some preliminary analyses show that maximum high-severity patch size continued to increase in the decade following the Miller et al. (2012b) study.

Fuels, topography, and weather are all strong drivers of fire spread, which in turn affects patch size distribution. The highly dissected topography of the Klamath Mountains helps limit rapid fire spread and patch size (Skinner et al. 2018). However, the steepness of the terrain allows for larger high-severity patches when ignitions occur at lower slope positions on the landscape, and slope and wind are aligned. On the other hand, the southern Cascade portion of the assessment area and much of the North Coast Range have less dissected topographies across which fire is able to spread more easily (Skinner and Taylor 2018, Stephens et al. 2018).

Collins et al. (2017) discussed the importance of the spatial characteristics of stand-replacing patches and developed a mathematical model to better describe the relationship between patch size and distance to edge (an important variable for postfire vegetation dynamics). The authors used two fires in the Klamath Mountains to validate their method, finding that although the fires were similar in size (~5000 ha) and the proportion of fire area that burned at high severity (~20 percent), they supported very different relationships of patch size and distance to edge. Furthermore, Miller et al. (2012b) found that within individual fires in the assessment area that burned between 1987 and 2008, the mean and maximum high-severity patch size in conifer vegetation types were all greater when (1) fire size was greater and (2) fires started later in the season. The sensitivity of postfire vegetation dynamics to the size and shape of high-severity patches—which are fundamental to patterns of seed dispersal, animal movement, plant regeneration, etc.—makes it important to acknowledge the importance of patch characteristics and not only the overall percentage of high-severity fire (Steel et al. 2018, Tepley et al. 2017).

The sensitivity of postfire vegetation dynamics to the size and shape of high-severity patches—which are fundamental to patterns of seed dispersal, animal movement, plant regeneration, etc.—makes it important to acknowledge the importance of patch characteristics and not only the overall percentage of high-severity fire.

Future—Assuming that maximum high-severity patch size will continue to increase as fire size increases, current trends in fire size (see the “Fire size and annual area burned” section below) make it safe to infer that maximum high-severity patch size and total area burned at high severity will continue to increase over time (Miller et al. 2012b).

Fire size and annual area burned—

NRV and comparison to current—Fire size and the amount of total area burned each year are driven to a great extent by weather interacting with topography and landscape conditions; today, both variables also respond strongly to fire suppression actions. Historically, winter precipitation and drought strongly influenced the interannual variability of fire size (Wahl et al. 2019), while the area burned each year was largely driven by the number of ignitions. Furthermore, landscape conditions played a larger role in year-to-year differences in fire size and area burned because frequent fire limited fuels across the landscape (Safford and Stevens 2017). Contemporarily, the amount of winter precipitation no longer appears to have the same influence on fire size (as seen in 2006 and 2017) and years with fewer ignitions no longer appear to limit burned area (as seen in 2018). This is largely due to increased temperatures, longer fire seasons, and higher fuel loading across the landscape (Wahl et al. 2019). In an analysis of fires between 1910 and 2008, Miller et al. (2012b) showed that climatic drivers of fire size in the assessment area shifted from drought-related variables before 1987 to summer precipitation-related variables thereafter, and the overall variance explained in fire size by climatic variables was much higher after 1987. This agrees with multiple other studies in the western portion of the United States that have found increasing importance of climatic variables in explaining fire size (Miller et al. 2009b, Westerling et al. 2006; see other references in Miller et al. 2012b). Miller et al. (2012b) noted that this was at least partly a result of the accumulation of fuels over the many decades of fire suppression, which has resulted in the transition of formerly fuel-limited forest types, such as pine-dominated YPMC (LANDFIRE Fire Regime I) (Schmidt et al. 2002), to climate-limited forest types with much higher densities of shade-tolerant species (Fire Regimes III and IV). High diversity in substrate productivity and associated vegetation composition and structure in the assessment area still play an important role in dictating fire's ability to spread across the landscape; however, the high level of fuel continuity present due to fire suppression has reduced structural diversity across the landscape, making it a less important driver of interannual variability. In addition to factors that drive year-to-year differences, topography should not be overlooked when it comes to overall factors that influence fire size and annual area burned. In the Klamath Mountains, for example, fire spread historically was largely controlled by the region's extreme topography, with sharply contrasting slopes, abundant ridgelines, and large rivers limiting fire runs (Skinner et al. 2018, Taylor and Skinner 2003).

Stephens et al. (2007b) estimated that prior to Euro-American settlement, approximately 13 percent of the roughly 5.5 million ha of land covered by mixed-conifer forests in California burned in an average year when assuming a median FRI of 8 years. Using a more conservative median FRI of 20 years (which is closer to the actual fire rotation, Mallek et al. 2013), the estimate drops to about 5 percent annual area burned. However, the annual area that burned across California wildlands between 1950 to 1999 was only a fraction of the area that burned in an average year before Euro-American colonial settlement (Stephens et al. 2007b). According to Safford and Stevens (2017), about 5 percent of the landscape covered by YPMC forests in the Sierra Nevada and southern Cascades likely burned in an average year, with a high amount of variability between years. Although no studies estimate presettlement annual area burned specifically for the assessment area, it is probably safe to assume that, based on the frequency of fires recorded for YPMC forests in the assessment area, the proportion of the area burned annually was likely similar to that of the Sierra Nevada and southern Cascades.

Despite a lack of information on the proportion of area burned annually at the assessment area level, there have been several efforts to determine this proportion for locations within the assessment area. Taylor and Skinner (2003) conducted a study looking at historical fire regimes in a 2325-ha area in the south-central

Klamath Mountains. The authors found the median proportion of area burned annually for the presettlement period (pre-1850) was approximately 5 percent. This is about five times the median proportion of area burned annually during the fire suppression period (1905–1995). In another study, Taylor and Skinner (1998) reported fire rotations for a 1570-ha study area composed largely of Douglas-fir-dominated mixed conifer in the northwest Klamath Mountains. Based on these reported fire rotations, we are able to calculate that, on average, about 5 percent of the study area burned annually during the presettlement period. The smallest proportion of area burned annually was observed during the fire suppression period (1905–1992) when an average of about 4 percent of the study area burned annually. When fire data from 1987 are removed, a year in which most of the study area burned, the average proportion of area burned annually drops to 3 percent (Taylor and Skinner 1998).

For contemporary fires, Stephens et al. (2018) reported a decrease in the number of hectares burned between the 1950s and 1970s in the North Coast Range portion of the assessment area, but found an increase in more recent years. For the California portion of the assessment area, Miller et al. (2012b) found that mean and maximum fire size, along with total area burned for fires >400 ha, were higher during the 1987–2008 period than any other recorded period since the beginning of the 20th century. This trend in fire size was largely driven by four years (1987, 1999, 2006, and 2008) when clusters of large fires were caused primarily by widespread lightning events that exceeded the capacity of fire suppression efforts (Miller et al. 2012b). Since the publication of Miller et al. (2012b) there have been many additional large-fire years in the assessment area, including 2012, 2014, 2017, 2018, and 2020.

Since effective fire suppression has been in place, there has been a high level of variability in fire size, with mostly very small fires burning most years (because of fire suppression) combined with a handful of years with very large fire events. This variability is in part due to some years having more extreme fire weather than others (Trouet et al. 2009), but it is also due to the successful suppression of fires burning under mild conditions, which lowers the average size of fires and burned areas in years without extreme fire weather. When widespread thunderstorms are accompanied by significant precipitation, the resulting fires are generally easy to suppress as the moisture, including high humidity, keeps them from spreading far before initial crews arrive. Conversely, widespread lightning not accompanied by significant precipitation can start many fires that spread quickly and may be already beyond control when initial attack forces arrive (Miller et al. 2012b, Skinner et al. 2018).

Some of the largest fires in the recorded history of both California and Oregon have burned in the assessment area during the past two decades.

Some of the largest fires in the recorded history of both California and Oregon have burned in the assessment area during the past two decades. Some noteworthy examples include the August Complex (418 077 ha burned in 2020, California's largest forest fire), the Biscuit Fire (202 350 ha burned in 2002, Oregon's largest forest fire), the Mendocino Complex (185 805 ha in 2018), the Carr Fire (92 936 ha in 2018), the Klamath Theater Complex (77 715 ha in 2008), the Slater Fire (63 617 ha in 2020), the Red Salmon Complex (58 428 in 2020), and the Big Bar Complex (57 060 ha in 1999), all of which (except the Mendocino Complex and the Carr Fire) were started by multiple lightning ignitions (CDF 2021). As of press time, the 2021 fire season has been no exception to this trend. While it is likely that large fires caused by lightning occurred in the assessment area historically, fires exceeding 40 000 ha were certainly much less frequent than they are today, and they were nonexistent in the assessment area during the first half of the 20th century (Collins et al. 2019, Weatherspoon and Skinner 1995).

In a study assessing large wildfire (>405 ha) trends throughout the Western United States, Dennison et al. (2014) found that in most ecoregions, including the Sierra/Klamath/Cascades ecoregions, there has been an increase in the number of large fires and total fire area between 1984 and 2011. Furthermore, the authors found a trend of large fires occurring earlier in the fire season at a rate of about 1 day earlier per year (Dennison et al. 2014). In a study looking at a longer time period (1916–2003), Littell et al. (2009) linked increased wildfire area burned with low precipitation, high temperatures, and a negative Palmer Drought Severity Index (PDSI) immediately leading up to and during the fire season, causing fuels to dry more quickly.

Future—Fire activity and annual area burned are strongly linked to climatic conditions in Western United States forests (Agee 1993; Lenihan et al. 2003; Trouet et al. 2006, 2009, 2010; Westerling and Bryant 2008; Westerling et al. 2006, 2011). Based on evidence of increasing fire activity and annual area burned over the past several decades, projections of drought probabilities (e.g., Abatzoglou and Williams 2016; Westerling et al. 2006, 2011), and longer fire seasons allowing for large fires in most years regardless of annual precipitation (Wahl et al. 2019), area burned and the risk of large wildfires are expected to increase in the future (Abatzoglou and Williams 2016, Loudermilk et al. 2013, Scheller et al. 2018, Westerling and Bryant 2008, Yang et al. 2015). Abatzoglou and Williams (2016) attribute the increase in burned area to increases in fuel aridity and suggest that while increased fire activity may eventually limit burned area due to reductions in fuel availability, this effect will be delayed due to the high levels of fuel currently in the forest. Furthermore, because YPMC forests in the assessment area have mostly transitioned from

LANDFIRE Fire Regime I to Fire Regimes III and IV, they will be more sensitive to interactions between increases in fuel aridity due to higher fuel loads from a century of fire suppression, drier fuels from increasingly drier summers (and drought years), and lengthening of the fire season (Westerling and Bryant 2008, Westerling et al. 2006).

Restaino and Safford (2018) summarized the results of seven studies whose future burned area projections included the assessment area. In all cases, large increases in burned area were projected, with most studies projecting increases of >100 percent in burned area by the end of the century. In a study using a range of future climate and development scenarios, Westerling et al. (2011) projected substantial increases in large-fire occurrence and total burn area by the year 2085, with large portions of northwestern California potentially seeing a 300 percent increase in total area burned. According to Barr et al. (2010), annual area burned across the Klamath Basin (an area that is largely within the assessment area) is expected to increase from 2.7–3.3 percent per year to 11–22 percent per year by 2075–2085. This increase (400 to 667 percent) would result in about 133 550 ha burning in an average year.

The impact of climate change on the risk of large wildfires may differ depending on the vegetation type (Westerling and Bryant 2008). Highly productive forests that have a climate-limited fire regime are expected to be at an increased risk of large wildfires as fuels become more available as a result of increases in temperature and longer fire seasons. Less productive forests limited by water availability or edaphic conditions, on the other hand, tend to be fuel limited when fire is frequent enough, ultimately reducing the risk of large fires after a certain point. However, areas where fire regimes were historically fuel limited currently have such a buildup of fuels that they are now more limited by climatic conditions (i.e., there is always sufficient fuel to burn) and in some cases, limited by lack of ignitions where American Indian burning was a major component of the historical fire regime.

Trouet et al. (2009) found that years with high fire weather index values in northern California and southern Oregon (i.e., a high fire growth potential based on low fuel moisture and a dry and unstable lower atmosphere) were associated with high annual area burned. Lower atmospheric stability and low moisture levels also tend to be associated with dry lightning days that can further increase the chances of large areas burning (Rorig and Ferguson 1999). Over the past several decades, years with the largest areas burned were also years with regionwide dry lightning events (Miller et al. 2012b), with 2020 being the most extreme example of this phenomenon.

Fire seasonality—

NRV and comparison to current—In the Mediterranean type climate of the assessment area, fire season (the portion of each year in which we expect wildfire activity) is linked to the 3- to 6-month annual drought period that occurs during the summer months (van Wagtendonk et al. 2018). According to Westerling et al. (2011), 94 percent of ignitions and 98 percent of area burned in the continental Western United States occur between May and October, typically peaking between July and September (van Wagtendonk et al. 2018, Taylor and Skinner 2003). Fire season length varies, however, depending on precipitation patterns and resulting fuel moisture levels (fuel availability). Forests at lower elevations and on south- and west-facing slopes tend to burn earlier in the season because fuels become drier sooner (Beaty and Taylor 2001, Show and Kotok 1929, van Wagtendonk et al. 2018).

Most studies conducted in the assessment area that have looked at the relative position of fire scars (i.e., earlywood, latewood, ring-boundary) to determine the historical seasonality of fires have determined that presettlement fires burned predominantly either late in the growing season or during the dormant season (Fry and Stephens 2006; Skinner 2006a; Skinner et al. 2009; Taylor and Skinner 1998, 2003). One exception to this pattern is presented by Skinner et al. (2009), who assessed the seasonality of fire between 1700 and 1900 in the Mendocino National Forest. While they found similar results for their moist mixed-conifer sites, their dry mixed-conifer sites had more than 80 percent of scars formed during the growing season, with less than 20 percent forming at the ring boundary. The authors suggest that this observed difference may be partly due to differences in recording tree species that were available in the different studies (i.e., Douglas-fir vs. ponderosa pine and incense cedar), but that it is more likely due to the proximity of their dry study sites to the Sacramento Valley. This likely allowed the seasonality of fire to be influenced by frequent indigenous burning in adjacent woodlands and grasslands (Anderson 2005, Skinner et al. 2009). Similar to the results of Skinner et al. (2009), Metlen et al. (2018) found 70 percent of the scars in their study formed during the growing season, while 30 percent formed at the ring boundary in stands adjacent to the oak woodlands of the Rogue Valley.

Although the overall seasonal pattern of fire risk being driven by fuel loads and fuel moistures has remained similar over time—with July and August being the high fire months (Show and Kotok 1929, van Wagtendonk et al. 2018)—fire season length is expanding rapidly (Westerling 2016, Westerling et al. 2006). According to Westerling et al. (2006), between 1970 and 2003, fire season length in the Western United States increased by more than 2 months. This was attributed primarily to the fire season beginning earlier over time, driven by higher spring and summer temperatures initiating an earlier snowmelt (Westerling 2016).

Furthermore, the observed shift toward a lower snow-to-rain ratio in recent years associated with increases in minimum temperatures has further exacerbated this by reducing snowpack, which results in fuels drying earlier in the season (Mote 2006, Rapacciuolo et al. 2014, Thorne et al. 2015).

Future—Fire seasons across the Western United States have been increasing in length because of warmer temperatures and earlier spring snowmelt associated with climate change (Abatzoglou and Williams 2016, Westerling 2016, Westerling et al. 2006). Climate projections indicate that warming will continue and that, while future precipitation trends are uncertain, snow-to-rain ratios will continue to decrease (Diffenbaugh et al. 2015, Micheli et al. 2018, Wimberly and Liu 2014). This may have a particularly large impact on fires in the assessment area because of the continuous nature of fuels in many areas that usually receive a snowpack. Furthermore, as the climate becomes warmer and drier, lightning-caused fires, which are typically concentrated at higher elevations, are expected to increase (Yang et al. 2015). Restaino and Safford (2018) summarized findings in the literature and found that projections for increased fire season length by 2100 ranged from an additional 3 to 8 weeks for California as a whole.

Fire seasons across the Western United States have been increasing in length because of warmer temperatures and earlier spring snowmelt associated with climate change.

Grazing

NRV and comparison to current—

In the assessment area, deer, antelope, elk, and other wildlife were abundant on the landscape and grazed the land for millennia prior to Euro-American colonial settlement. Large herds were described occasionally roaming the valleys (Merriam 1899), but the relatively constant movement of these herds likely reduced their impacts on the land. Heavy grazing began in earnest in the 1860s and 1870s with the arrival of Euro-Americans and their livestock. In the beginning, a large proportion of grazing animals was sheep (for example, the Ashland Reserve, located in the northern portion of the assessment area, was grazed by large flocks of sheep from 1870 to at least 1924). However, cattle became the predominant livestock after World War I because of the demand for beef (LaLande 1980).

Grazing alternated between mountains and valleys according to season and resource availability. During the summer months when snow cover was not an issue, cattle grazed in mountain pastures (LaLande 1980, Pinchot 1905). The grazing season in the mountains of northern California typically extended from early June to mid-October, according to Pinchot (1905). Livestock densities were not regulated in any fashion, and by the early 1900s, it was already clear that overgrazing—throughout the West—was having detrimental effects on the landscape, and regulations were needed to control the extent of livestock grazing (Pinchot 1905).

The designation of the Forest Reserves in the 1890s and early 1900s was, in part, due to concerns about deleterious levels of livestock grazing on public lands.

Since heavy livestock grazing led to a depletion of forage, intentional burning and seeding were sometimes carried out in order to increase forage (Hayes 1959, LaLande 1980, Minore 1978). Fires were set to keep brush and timber from encroaching on pastures and fire was also set in forested areas in an attempt to convert the land for grazing (LaLande 1980, Skinner et al. 2018). In denser mixed-conifer forests, grazing typically only occurred in openings created and maintained by fire, as these more densely forested areas lacked much natural forage (Leiberg 1900). In large, privately owned areas in the North Coast Range, unsuccessful attempts were made to convert well-stocked conifer lands into grazing lands, which resulted in the establishment of either hardwood-dominated forests or sparsely stocked conifers (Oswald 1970).

The reduction in herbaceous cover associated with intensive grazing practices may have also reduced fire frequency, especially in dry, frequent-fire forests (Belsky and Blumenthal 1997, Borman 2005, Fry and Stephens 2006, Norman and Taylor 2005, Skinner and Chang 1996, Skinner et al. 2009, Riegel et al. 2018). According to early reports and journals (e.g., Leiberg 1900), however, herbaceous cover was limited in densely forested areas, consequently grazing may not have had a major impact on the fire regime in moist mixed-conifer forests. Little is known about historical herbaceous cover in YPMC forests in the assessment area (see the "Composition" section below), thus determining any long lasting effects of grazing on the understory community is difficult to impossible.

The observed reduction in forage across grazing landscapes in the West coincided with periods of both drought and elevated precipitation, which likely magnified the effects of livestock grazing in different ways (Borman 2005, Cook et al. 2011, Pinchot 1905, USDI 1951). Drought periods probably further reduced herbaceous cover and influenced the location of grazers, with a severe drought in the 1850s forcing the movement of cattle out of the Sacramento Valley and into northeastern California (Riegel et al. 2018). Additionally, an extremely wet period during the early 1900s coupled with heavy grazing may have helped promote tree seedling establishment at the onset of the fire suppression era (Cook et al. 2011). Although grazing still occurs on national forest lands, it is much more regulated as a result of federal actions, such as the Endangered Species Act of 1973 and the Rescissions Act of 1995, which required National Environmental Policy Act (NEPA) compliance for grazing allotments on national forest lands (Public Law 104-19, Section 504). Thus, the impacts of modern grazing are generally less extensive and intensive than during the 1800 and early 1900s.

Future—

Because active grazing allotments are still present throughout most of the assessment area, cattle grazing will likely continue into the foreseeable future. Future impacts of grazing may be exacerbated with future fires and drought conditions, yet grazing can also be used as a tool to assist with fuel reduction in targeted areas.

Insects and Diseases

There are a number of native and nonnative insects and pathogens that affect tree growth and survivorship in YPMC forests. Table 4 provides a list of the major insects and pathogens present in the assessment area and the main YPMC tree species that they affect. For a more complete listing and detailed description of insects and diseases that affect California forests, refer to the "California Forest Insect and Disease Manual" compiled by the Forest Service's Forest Health and Protection program and the California Department of Forestry and Fire Protection's Forest Pest Management forest health specialists (USDA FS and CAL FIRE, n.d.). For more details concerning what insects and diseases affect hardwoods of the Pacific Northwest, please refer to Niemiec et al. (1995).

Bark beetles are the most destructive group of insects in YPMC forests. Although a baseline level of tree mortality is always present, under favorable stand and climate conditions, bark beetle outbreaks can occur with major impacts to populations of affected tree species. Western pine beetle (*Dendroctonus brevicomis*), Jeffrey pine beetle (*D. jeffreyi*), mountain pine beetle (*D. ponderosae*), Douglas-fir beetle (*D. pseudotsugae*), and fir engraver beetle (*Scolytus ventralis*) are five of the main bark beetles affecting conifers in the assessment area.

The most damaging group of diseases affecting forest trees in the area is root diseases. Main ones include Heterobasidion root disease (*Heterobasidion* spp.), black stain root disease (*Leptographium wageneri*), Armillaria root disease (*Armillaria* spp.), and laminated root rot (*Phellinus sulphurascens*). Collectively, they affect all of the main YPMC species.

NRV and comparison to current—

There is limited information regarding insects and pathogens prior to Euro-American colonial settlement in the assessment area. Using what we know about insect and disease ecology, however, we are able to infer changes and trends that may have occurred over time. Changes in environment, insect or pathogen abundance, and host prevalence and susceptibility are important components driving the incidence of insect outbreaks and disease (Cobb and Metz 2017, McDowell et al. 2008, Raffa et al. 2008). It is likely that the open and spatially

Table 4—Major insects and diseases affecting yellow pine and mixed-conifer forests

	Host tree[a]										
Agent	**Pp**	**Pj**	**Pl**	**Ac**	**Cd**	**Pm**	**Qk**	**Qq**	**Ama**	**Ame**	**Ld**
Heterobasidion root disease, *Heterobasidion* spp.	✓	✓	✓	✓	✓	✓				✓	
Black stain root disease, *Leptographium wageneri*	✓	✓	✓			✓					
Armillaria root disease, *Armillaria* spp.	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	
Sudden oak death, *Phytophthora ramorum*[b]				✓		✓	✓		✓	✓	✓
Madrone canker, *Fusicoccum aesculi* or *Botryosphaeria dothidea*										✓	
White pine blister rust, *Cronartium ribicola*			✓								
Dwarf mistletoe, *Arceuthobium* spp.	✓	✓	✓	✓		✓					
Western pine beetle, *Dendroctonus brevicomis*	✓										
Jeffrey pine beetle, *Dendroctonus jeffreyi*		✓									
Mountain pine beetle, *Dendroctonus ponderosae*	✓		✓								
California fivespined ips, *Ips paraconfusus*	✓	✓	✓								
Fir engraver beetle, *Scolytus ventralis*				✓							
Douglas-fir beetle, *Dendroctonus pseudotsugae*						✓					
Douglas-fir tussock moth, *Orgyia pseudotsugata*				✓							
Pandora moth, *Coloradia pandora*	✓	✓	✓								
Flatheaded fir borer, *Melanophila drummondi*				✓		✓					
Western oak looper, *Lambdina fiscellaria somniaria*								✓			

[a]Host species and codes: Pp = ponderosa pine, *Pinus ponderosa*; Pj = Jeffrey pine, *P. jeffreyi*; Pl = sugar pine, *P. lambertiana*; Ac = white fir, *Abies concolor*; Cd = incense cedar, *Calocedrus decurrens*; Pm = Douglas-fir, *Pseudotsuga menziesii*; Qk = California black oak, *Quercus kelloggii*; Qg = Oregon white oak, *Quercus garryana*; Ama = bigleaf maple, *Acer macrophyllum*; Ame = Pacific madrone, *Arbutus menziesii*; Ld = tanoak, *Notholithocarpus densiflorus*

[b]Not all species identified as hosts are severely affected by sudden oak death; the disease only causes mortality in tanoak and a number of true oaks (e.g., coast live oak, black oak, Shreve oak, canyon live oak).

Table adapted from Safford and Stevens (2017) with additional information compiled from Niemiec et al. (1995) and the California Forest Insect and Disease Manual (https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev3_046410.pdf).

heterogeneous nature of YPMC forests maintained by frequent fire prior to Euro-American colonial settlement made these forests more resistant to insects and diseases than they are today. Fire suppression has led to an increase in overall stand densities making forests more susceptible to water stress and subsequent insect attacks, especially by beetles (Fettig et al. 2007), and there is strong evidence that thinning treatments are effective in reducing beetle-caused mortality in YPMC forests (Fettig et al. 2012, Restaino et al. 2019).

In 1899, A.D. Hopkins described beetle galleries found on dying sugar pine and western yellow pine in McCloud, California. At the time, these galleries were attributed to western pine beetle, however the patterns he described were most likely caused by mountain pine beetle (Snyder 2020). It was noted that infestations appeared to be associated with dying trees that were girdled by settlers and farmers as they cleared land (Webb 1906). According to Miller and Keen (1960), A.H. Hodgson conducted the first intensive survey focused on measuring the amount of beetle-killed timber in an area just west of Yreka on the Klamath National Forest in 1911. The amount of pine mortality within a given stand ranged from 10–20 percent in less heavily infested areas to 40–50 percent in more heavily infested areas (Miller and Keen 1960).

One study in central Oregon documented six major outbreaks between 1800 and 2000 of a yellow pine defoliator called the Pandora moth (*Coloradia pandora*). Pohl et al. (2006) showed that moth outbreaks are temporally linked to drought, which affects both the host susceptibility to defoliation as well as survivorship and reproduction of the pandora moth itself. Outbreaks generally occurred at 40- to 50-year intervals. Although the pandora moth does not typically cause mortality in yellow pines, it can severely defoliate individual trees and cause related reductions in radial growth (Pohl et al. 2006).

Working in the Russian Wilderness in 2015 in upper montane and subalpine forests, DeSiervo et al. (2018) found high levels of red fir and subalpine fir mortality, driven by interactions between tree stress (as a result of mistletoe (*Arceuthobium abietinum*) and *Cytospora* fungus) and fir engraver beetle (*Scolytus ventralis*). Lodgepole pine was also experiencing elevated mortality, apparently as a result of mountain pine beetle infestation, while ponderosa pine and Douglas-fir were among the tree species that had the lowest levels of recent mortality (DeSiervo et al. 2018).

The combination of prolonged drought and high tree densities have made forests in California highly susceptible to insects and diseases. According to a report published by the California Forest Pest Council, as of 2016, there were an estimated 102 million trees that died since the beginning of California's most recent drought, which began in 2012 (CFPC 2016); by 2017 the number had risen to about 150 million. Mortality was mostly in low-elevation pine and mixed-conifer forests and has been concentrated primarily in the southern Sierra Nevada. In 2016, the height of the mortality event, more than 4 million acres of surveyed forests in California exhibited elevated levels of tree mortality as a result of bark beetles (e.g., western pine beetle and mountain pine beetle) and wood borers (e.g., fir engraver beetle and flatheaded fir borer). Mortality in the assessment area, although not as severe as in other parts of California, was found primarily in yellow pine and mixed-conifer forests, with the largest mortality events occurring in dense forests at lower elevations and on south- and west-facing slopes (CFPC 2016).

While most of the recent mortality has been attributed to beetles, there are a number of diseases of concern that are also causing considerable mortality in YPMC forests. White pine blister rust (*Cronartium ribicola*), a disease that targets five-needle pines (white pines), first appeared in the assessment area in 1929. Blister rust has caused extensive mortality in sugar pine, western white pine, and whitebark pine throughout northern California (Kliejunas and Adams 2003). Annosus root disease (*Heterobasidion occidentale*), a widespread disease present throughout California, is known to affect all western conifers (Schmitt et al. 2000). In pines, this fungus causes mortality by girdling them at the root crown, while in true firs, it more commonly causes root and butt rot that weakens the trees, ultimately leading to structural failure or increased susceptibility to bark beetles. Annosus root disease is known to readily infect freshly cut stumps, causing trees adjacent to the infected stumps to become infected themselves. Stands that have a history of logging-related disturbance tend to be the most severely affected. Furthermore, increased conifer densities associated with fire suppression have increased the ability of this root disease to spread (Schmitt et al. 2000).

Port Orford cedar, an ecologically and economically important tree native to southwestern Oregon and northwestern California, has experienced widespread damage from a different root disease caused by *Phytophthora lateralis*, a fungus closely related to sudden oak death (*Phytophthora ramorum*). This pathogen is an invasive nonnative that has been a concern for managers since its introduction in the early 1950s (Zobel et al. 1985). Although Port Orford cedar is only a minor component of the mixed-conifer forest type, it contributes to the assessment area's uniquely diverse forests. Because *P. lateralis* is transported via aquatic spores, it is limited by dry conditions and its dispersal is largely controlled by habitat connectivity. This connectivity can take the form of road vectors, for example, in which humans have facilitated spread over relatively large distances, or through dense host populations in which the pathogen can spread easily from one individual to the next (Jules et al. 2002, Zobel et al. 1985).

It is suggested that as the climate continues to warm and drought conditions become more prevalent, the impact of bark beetles on conifer forests is likely to become more severe.

Future—

It is suggested that as the climate continues to warm and drought conditions become more prevalent, the impact of bark beetles on conifer forests is likely to become more severe (Bentz et al. 2010, Raffa et al. 2008, Scheller et al. 2018). According to Scheller et al. (2018), persistent drought conditions will likely lead to increased beetle outbreaks, subsequently decreasing ecosystem carbon storage capacity. These outbreaks can also lead to shifts in forest composition because species such as incense cedar are not affected in the same way as pines. Similar to increases in beetle outbreaks, a warmer, drier climate is also expected to increase the impact of certain pathogens such as Armillaria root disease via increasing host susceptibility. Impacts of other pathogens, such as sudden oak death, on the other hand, may decrease due to temperature and moisture directly affecting pathogen reproduction, spread, infection, and survival. If the climate becomes wetter (rather than drier) in addition to warming, pathogen impacts associated with host susceptibility may stay the same, while pathogens directly affected by moisture availability may increase (Kliejunas 2011, Sturrock et al. 2011).

Logging

Although trees were cut and used by American Indians and early explorers prior to Euro-American colonial settlement, large logging operations were not established in the assessment area until the late 19th century. Here we provide a short summary of Euro-American logging in the assessment area in order to provide some context for current forest conditions.

The first sawmills operating in the assessment area were built in the early 1850s, and by the turn of the century extensive logging operations had been established in areas with relatively gentle topography that were abundant in large, desirable pines, such as the southern Cascades and parts of the eastern Klamath Mountains. In the early 1900s, stumpage fees for trees were highest for sugar pine, followed by yellow pine, and then Douglas-fir. Other less desirable species, such as white fir and incense cedar, brought about one-fifth to one-quarter the rate of sugar pine (Lamm 1944). Most lumber mills present in less accessible areas supported early settlers and mines and were relatively small-scale operations (Bower 1978, Stuart 1928), with the exception of the Lamoine Lumber and Trading Company, which was a 35-mile-long railroad logging system in Shasta County that stretched from Lamoine to Trinity Center (Smith 2009).

A forester named R.Y. Stuart described the status of sawmills on the Klamath National Forest in 1928 (Stuart 1928: 5):

> The virgin pine and fir stands of the Klamath National Forest constitute a reserve supply of 12½ billion feet, which will someday be cut to supply California's growing timber needs.…About a dozen small sawmills supply the local demand for timber in the more remote areas. There is only one large mill at present in the Klamath Forest, which cuts Government and privately owned stumpage. The rough topography of the country makes logging difficult and railroad construction expensive, and the high percentage of Douglas fir in the forest precludes the development of logging operations on a large scale in the near future.

There were also two very large operations on the Shasta National Forest at the time—the McCloud River Lumber Company, which, at its peak, owned or controlled more than 600,000 acres of timberland (Moore, n.d.), and the Weed Lumber Company, which relied on an extensive railroad network (Weed 2000).

Wartime and postwar demand for timber led to heavy timber harvesting starting in the 1940s (LaLande 1980). By the mid-1950s, Port Orford cedar, Douglas-fir, white fir, and sugar pine were being harvested from the central Siskiyou Mountains. With the goal of shifting the forest composition toward more desirable conifer species (e.g., sugar pine, ponderosa pine), the Forest Service initially tried to eliminate incense cedar from mixed-conifer stands through logging operations. This policy ended relatively quickly as it was realized that larger cedars provided excellent lumber if diseased trees were removed, and that once stands were cut over, incense cedar was capable of regenerating prolifically as long as there was adequate fire protection (Mitchell 1918).

According to Hayes (1959), logging in the Douglas-fir forest type in southwestern Oregon was mostly by clearcutting, whereas partial cutting was the most common method in ponderosa pine forests. The author also noted that mixed-conifer forests required different cutting practices because of the number of age classes and species present, the low merchantable volumes of timber in many places, and the apparently mediocre quality of the old growth. Additionally, clearcut areas in both the interior Douglas-fir zone and the mixed-conifer zone were noted as having a lack of sufficient reproduction, often being replaced by brush (Hayes 1959).

The selective logging of certain species followed by fire suppression has led to shifts in the species composition and structure of forested stands (Hasel 1932, Laudenslayer and Darr 1990). As larger diameter trees were preferentially logged, harvested stands became less structurally diverse and dominated by high densities of smaller diameter trees. In A.A. Hasel's progress report for the Forest Service that evaluated the effects of logging on the Klamath National Forest (Hasel 1932), it was noted that in a stand of yellow pine-dominated mixed-conifer forest that experienced a partial cut (41.3 percent of all trees >11.6 inches [29.5 cm] dbh), the percentages of ponderosa pine and incense cedar decreased, while Douglas-fir and white fir increased (for trees ≥4 inches [10 cm] dbh). Hasel stated that seedling establishment on the Klamath National Forest was limited between 1916 and 1920, but showed a dramatic increase between 1920 and 1930 almost entirely as a result of incense cedar. By volume, the species that made up the highest percentage of the stand after logging was white fir, followed by Douglas-fir, incense cedar, and ponderosa pine. Sugar pine was only a minor component of the stands studied, so there were not large changes in volume or density for this valuable timber species (Hasel 1932).

Early logging practices, especially clearcutting, produced shrubfields and heavy fuel loading from logging slash (which was left onsite), creating a greater likelihood of the stand burning at high severity in the event of a fire (Cooper and Kelleter 1907, Laudenslayer and Darr 1990, Show and Kotok 1929). According to Cooper and Kelleter (1907), "The [fire] danger is further emphasized by the presence of inflammable chaparral areas and the vast quantities of slash on the logged lands. Fire on these areas is of the hottest character, and once started is extremely difficult and often almost impossible to check." Show and Kotok (1924, 1929) also stressed the importance of managing the heavy fuels that were left over after logging operations, noting the destruction by slash fires and the difficulty in controlling them. Because of the fire danger posed by abundant slash from logging operations, there was a movement toward fuels management in these areas, usually taking the form of piling and burning the slash (Cooper and Kelleter 1907, Hasel 1932, Show and Kotok 1924).

The combination of logging and fire suppression throughout the assessment area has led to substantial changes in the density and composition of YPMC forests. Overall, compared to the average presettlement forest stand, stands harvested for timber on the national forests (and then subject to long-term exclusion of fire) tend to be lacking large trees, are more homogeneous in density and size-distribution, have a higher density of shade-tolerant trees, and support elevated surface fuel. These stand conditions tend to be more susceptible to stand-replacing fire and increased tree mortality due to drought stress and insect outbreaks. In contrast, forests where harvest was part of a thinning and fuel reduction treatment are substantially less prone to mortality from fire and drought stress, and dominance of fire-tolerant species has increased. See the "General Forest Structure" and "Composition" sections below for an in depth discussion of these changes.

The combination of logging and fire suppression throughout the assessment area has led to substantial changes in the density and composition of yellow pine and mixed-conifer forests.

Nutrient Cycling

NRV and comparison to current—

Although information on nutrient cycling prior to Euro-American colonial settlement is not available, it is possible to infer changes that have likely occurred based on our knowledge of how climate, site conditions (such as topography and parent material), disturbance, and biotic communities affect nutrient cycling (Foster and Bhatti 2005). While water availability is typically the most limiting factor for tree growth in Mediterranean-type climates (Stephenson 1990), the abundance of serpentine soils in the assessment area makes nutrient availability a key limitation in many areas (Alexander et al. 2007). Serpentine soils are low in essential macronutrients, such as nitrogen (N) and phosphorous (P), have a low calcium-to-magnesium ratio, and contain various heavy metals, influencing the composition, biomass, and structure of vegetation. On both serpentine and nonserpentine substrates, nutrient dynamics are chiefly driven by decomposition rates. In general, because precipitation during the growing season is minimal, YPMC forests exhibit slow decomposition rates (Hart et al. 1992), but the rate generally increases from east to west, with increasing precipitation and decreasing profundity of the dry season. Because of the assessment area's diverse topography and variation in annual precipitation, there is likely a high level of spatial variation in decomposition rates.

Because decomposition rates in YPMC forests are slow, other ecosystem processes play an important role in accelerating the rates of nutrient cycling. There are several studies from other parts of California that have assessed the impact of fire, or the lack thereof, on nutrient dynamics in dry mixed-conifer forests that can provide valuable insight into how nutrient cycling today may be different from the presuppression period. When fire occurs, pH usually increases and calcium, magnesium, and potassium cations are released and made available for plant uptake. Immediately after fire, however, soil nutrients are extremely vulnerable to being removed by erosion and leaching (Murphy et al. 2006, St. John and Rundel 1976). Furthermore, nitrogen volatilization can initially cause a decrease of available nitrogen in the system, but this decrease is usually counteracted by the establishment of nitrogen-fixing plants (e.g., *Ceanothus* spp.) following fire (Johnson et al. 2005, St. John and Rundel 1976, Yelenik et al. 2013). Frequent fires would not only have consumed needles, small twigs, and leaves of trees, but would have created conditions that supported more herbaceous vegetation that would have increased the input of nutrients from fine material and the recycling of fine roots.

A study conducted in the Klamath National Forest assessed variability in both symbiotic (i.e., N-fixing shrubs) and asymbiotic (i.e., free-living soil N fixers) biological nitrogen fixation after fire (Yelenik et al. 2013). The authors found that asymbiotic N fixers were phosphorous limited, while symbiotic N fixers (specifically, in this case, *Ceanothus integerrimus*) were more limited by competition with non-N fixers than by nutrients. This study was conducted in relatively productive Douglas-fir-dominated stands with a large hardwood component and high competition between recovering N-fixing shrubs and later successional tree species, possibly leading to lower fixation rates. The authors also proposed that the steep slopes and rocky soils of the Klamath Mountains reduce the ability for soils to form with a sufficient organic horizon, also negatively affecting N fixation. Despite N fixation assisting in the recovery of nitrogen in forests after fire, the authors suggest that N inputs from certain N-fixing shrubs may not be sufficient to actually restore the amount of N lost to fire. Fire suppression has likely caused pre- and postfire differences in N levels to be much larger than they would have been historically because of the elevated level of biomass in fire-suppressed stands. Yelenik et al. (2013) noted that under a more frequent low-severity fire regime, fires likely promoted shrub growth within intact older stands, reducing the loss of N to fire and increasing overall N-fixation rates.

Work conducted by Miesel et al. (2009) in the assessment area suggests that forest density and species composition in YPMC forests can influence pH as well as soil C:N ratios five to six years after thinning treatments. In their study, they compared stands that were thinned to favor pine species with stands that were thinned to favor large trees regardless of species. The pine-preference treatment did not differ in pH from the control, while the size-preference treatment had a small but significant decrease in pH. Furthermore, the size-preference treatment had a large and significant increase in total inorganic N compared to the pine preference and control. Both treatments showed significant decreases in soil C:N ratios compared to the control (Miesel et al. 2009). These observed differences in treatments indicate that variability in tree densities and species compositions that were historically present as a result of frequent fire likely created additional heterogeneity in soil and forest floor nutrient availability and cycling across the landscape. Lack of frequent fire, increased tree densities, and shifts in species composition have likely changed nutrient dynamics.

Future—
Although it is difficult to project future trends in nutrient cycling, we are likely to see a continuation of fire suppression effects on nutrient dynamics in which nutrients will continue to accumulate in systems lacking fire and will periodically be released during severe fire events, leading to an increased likelihood that released nutrients will be lost to runoff following fire. Restoration efforts to thin and reintroduce fire may help to prevent large losses to high-intensity fire events.

Tree Mortality

NRV and comparison to current—
Background tree mortality is constantly occurring in YPMC forests despite being overshadowed by large, abrupt tree mortality events caused by drought, disease, insect outbreaks, or fire (e.g., Berner et al. 2017, Guarin and Taylor 2005, McKenzie et al. 2009, Young et al. 2017). According to Stephenson and van Mantgem (2005), background tree mortality rates present in otherwise healthy forests are relatively low (<2 percent per year). Using repeat observations of trees distributed throughout Oregon and Washington, Reilly and Spies (2016) documented mortality rate distributions in different vegetation and disturbance types between 1992–1997 and 1997–2007 (with a minimum of 4 years and a maximum of 10 years between each remeasurement). The study found biotic disturbances (i.e., insects and pathogens) to be widespread, but causing relatively low levels of mortality (<2.5 percent/yr), while abiotic disturbances (i.e., fire, wind, landslides) were less pervasive, but caused intermediate levels of mortality (5–25 percent/yr). Mortality rates in mature/

old-growth stages of structural development were generally higher than rates in early- and mid-stage forests. Results from this study further suggest that fire and insects will continue to be key drivers in forest change, especially in drier forest types (Reilly and Spies 2016).

To assess the mechanisms behind background tree mortality in coniferous forests, Das et al. (2016) evaluated mortality factors in unlogged, old-growth forests in the Sierra Nevada over a 13-year period. The authors found anywhere from one to six mortality factors contributing to individual tree mortality, however, almost 90 percent of dead trees had only one or two mortality factors. Almost 60 percent of all dead trees assessed were killed by biotic agents, with insects (primarily bark beetles) causing the most mortality, followed by diseases. Growth suppression was the second most common factor, occurring most frequently in small, slow-growing trees, followed by mechanical damage. Broken stems were common in intermediate and large trees, while intermediate and small trees were sometimes crushed by larger falling trees. Although there were few significant positive relationships between factors, ordination results suggested that suppressed trees were highly vulnerable to mortality via other factors, and that bark beetles were drawn to trees that were already stressed. Elevation also appeared to play a role in the importance of mortality factors, with biotic factors peaking at mid-elevation, suppression generally declining with elevation, and mechanical factors increasing with elevation (Das et al. 2016).

Mortality rates in older forests throughout the Western United States have shown substantial increases over the past several decades, with California exhibiting the highest mortality rates compared to the Pacific Northwest and interior West regions (van Mantgem et al. 2009). van Mantgem et al. (2009) suggested that regional warming is likely the main contributor to mortality rate increases. Increases in temperature are thought to contribute to tree mortality because of the combined effects of increasing drought stress on trees and enhancing the growth and reproduction of insects and pathogens (Das et al. 2016, Raffa et al. 2008, van Mantgem et al. 2009). Interestingly, a study conducted in Yosemite National Park showed that, although mortality patterns on both south- and north-facing slopes were similar due to drought, the density of dead trees was higher on north slopes than on the south slopes, likely due to denser stand conditions caused by fire suppression and subsequent bark beetle outbreaks during drought overpowering the potential buffering effect of topography (Guarin and Taylor 2005).

Future—

A warming climate and increased occurrence of drought are projected to increase background mortality rates throughout Western United States forests and increase susceptibility to beetle outbreaks. Based on observations of recent drought-related mortality in the assessment area and California, future tree mortality may increase more in areas that already exhibit a higher climatic water deficit (i.e., areas that are hotter and drier, such as lower elevations and areas further from the coast) (Moore 2015, Young et al. 2017). Raffa et al. (2008) suggest that efforts to reduce the extent of susceptible hosts (i.e., reduce densities and increase forest heterogeneity and diversity) could reduce the probability of future landscape-level mortality events caused by beetle outbreaks. Removing smaller trees by thinning or fire may also help prevent large-tree mortality during drought periods (van Mantgem et al. 2016, Restaino et al. 2019). Scheller et al. (2018), on the other hand, found that fuels treatments (removal of younger conifers and forest thinning) did not show much effect on bark beetle outbreaks because of the ability of beetles to disperse over large areas. The authors suggest more aggressive crown thinning to increase the distance between host trees and reduce outbreak severity. It might be thought that increases in tree mortality associated with a warming climate could help to reduce higher forest densities and associated fire risk owing to fire suppression, but mortality agents other than fire do not consume appreciable amounts of tree biomass, leading to increases in dead fuel loadings and fire risk (Hicke et al. 2012). In addition, unlike fire, which preferentially kills smaller trees, beetle-driven mortality is focused on larger trees, which are already very depleted in California forests (Restaino et al. 2019).

Future tree mortality may increase more in areas that already exhibit a higher climatic water deficit.

Wind Events

NRV and comparison to current—

Historical information on the effect of windthrow on YPMC forests in the assessment area is essentially nonexistent, but it is likely that large windthrow events were rare. According to Atzet and Martin (1992), windthrow was the third most prevalent disturbance agent (6 percent of 838 plots) over a 300-year period across a variety of forest types in the assessment area. Notable windthrow events have mostly been concentrated in Oregon and Washington, largely affecting forests in the Coast Range and western Cascades (i.e., north of the assessment area) in which large areas of forest have been blown down in single wind events (e.g., Knapp and Hadley 2011, Lynott and Cramer 1966). The famous 1962 Columbus Day Storm caused more devastation in the Pacific Northwest than any other recorded windstorm, with more than 11 billion board feet of timber damaged in Washington and Oregon, 98 percent of which was on the west side of the Cascade Range. Strong southerly winds associated with this event blew through the northern portion of the assessment area, however, the full extent of damage that occurred within the assessment area is unknown (Lynott and Cramer 1966).

More recent windthrow events can help us to infer the probable impacts from events prior to Euro-American colonial settlement. Hilimire et al. (2013) reported on a 2011 extreme wind event in high-elevation YPMC and red fir forests near Mammoth Lakes in the Sierra Nevada. Average size of windthrown trees was about 55 cm dbh, which was 2.2 times that of the average stand tree diameter prior to the wind event. Red fir and lodgepole pine were the most affected by the wind event, with white fir, Jeffrey pine, and western white pine being much less affected. The type of windthrow damage (i.e., uprooted vs. snapped) did not differ among species or tree size. Because the observed windthrow damage primarily affected larger trees, it is likely that events such as this can have a long-term impact on stand structure, composition, and regeneration.

In the assessment area, wind events at a smaller scale (relative to those discussed above) tend to play an important role in creating openings in stands dominated by white fir (Agee 1991). In 1996, high winds following accumulated snowfall caused stems to snap and trees to be uprooted over several thousand acres in the central Klamath Mountains (Skinner et al. 2018). This event generated considerable fuel accumulation, which, according to Jimerson and Jones (2003), contributed to widespread high-intensity fire in the 1999 Megram Fire that burned about 50 000 ha in the Trinity Alps Wilderness (Agee and Skinner 2005, Jimerson and Jones 2003).

Future—It has been suggested that extreme wind events may increase because of climate-driven changes in weather patterns (Peterson 2000). If these projections are correct, we may see increased impacts of severe wind events in the assessment area.

Structure

Forest Landscape Structure

NRV—

In the assessment area, structural diversity of YPMC forests on the landscape scale is driven by many of the same elements that have created such high plant diversity in the area. Soil fertility differences that are influenced by the area's complex geology, combined with steep elevational gradients and complex topography, create a diverse landscape with spatially complex disturbance regimes (Coleman and Kruckeberg 1999, Skinner et al. 2018). Aspect, slope, and elevation are key drivers of species composition and vegetation structure. Dense mixed-conifer stands tend to be found on deep soils, north and east aspects, and at higher elevations, while open, yellow pine-dominated stands tend to be found on shallower soils, more southerly and westerly aspects, and at lower elevations, with Jeffrey pine or

California foothill pine (*Pinus sabiniana* Douglas ex Douglas) typically found on ultramafic soils. The addition of key ecosystem processes, such as fire, to these forests greatly increases heterogeneity across the landscape. Although information about the historical spatial patterns of YPMC forests at both the local and landscape level in the region is limited, there is some information available in early Forest Service documents, historical accounts from early explorers, historical photographs, and inferences made from age structures in current stands.

Haefner (1912) described how forest patterns on the Siskiyou National Forest were largely driven by soil characteristics when fire is absent. He noted that forests found on the sandstone and limestone formations located in the northwestern portion of the Siskiyou National Forest were largely Douglas-fir dominated, while higher precipitation on the western portion (more coastal) supported denser forests than on the eastern portion (more inland) where sugar pine and fir became more common. On granitic soils, which cover large areas in the middle of the Siskiyou National Forest and small patches in the northwestern portion, Douglas-fir stands were still present, but Douglas-fir-dominated, mixed-conifer stands became more dominant. Rockier sites on these granitic soils tend to be less productive and support a less dense forest. YPMC forests in this region of the assessment area were described as being mostly restricted to serpentine soils with mixes of yellow pine, sugar pine, incense cedar, and Douglas-fir present in more open stands (Haefner 1912).

Fire is a key factor in the structural dynamics of YPMC forests in the assessment area. Frequent, patchy, low- to moderate-severity fire helped to create a mosaic of uneven-aged forest across YPMC sites (Taylor and Skinner 2003). These historically more spatially complex fires created a fine-grained forest matrix with variably sized openings that led to greater heterogeneity across the landscape (Taylor and Skinner 1998). Wetter, more dense, mixed-conifer areas probably exhibited a coarser grained landscape structure as a result of more severe fire effects caused by higher fuel loads and fuel continuity (van Wagtendonk et al. 2018).

These historically more spatially complex fires created a fine-grained forest matrix with variably sized openings that led to greater heterogeneity across the landscape.

Frequent, low-severity fire was an important ecological process that maintained YPMC structure, but areas of severe fire effects could reset the successional clock in portions of the landscape. Early-seral areas in YPMC forests are often dominated by montane chaparral species, which respond positively to the enhanced light and nutrients provided by the open canopy. Undisturbed successional processes usually lead to the eventual reestablishment of conifers, however, if these areas reburn, chaparral can dominate the site for many decades (Odion et al. 2010, Show and Kotok 1924, Tepley et al. 2017). Figure 7 provides an example of an area in 1933 located north of French Gulch in the Clear Creek watershed in Shasta County that

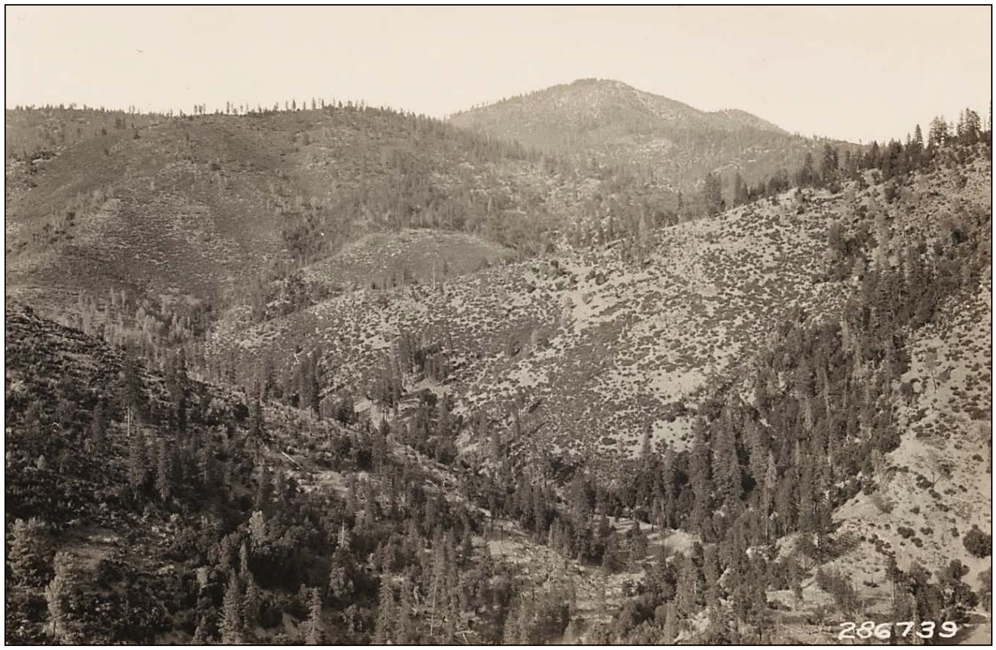

Figure 7—June 26, 1933 eastward view from Shasta County showing area severely deteriorated by repeated fires. Note semi barren chaparral of Oregon white oak. Wieslander Vegetation Type Mapping Collection courtesy of the Marian Koshland Bioscience, Natural Resources & Public Health Library, University of California, Berkeley.

had repeated fires prior to the photo being taken, leading to large deforested areas covered in chaparral. These areas are very steep, hot, and dry and have burned a number of times in recent decades as well. Frequent burning historically would have kept these areas quite open or dominated by chaparral (once it captured the site).

In a survey of Mount Shasta by C. Hart Merriam (1899: 30), he described the forested areas as being broken up on the landscape by extensive patches of chaparral:

> On the south and west the open pine forests of the basal slopes are interrupted by extensive parks, which from a distance appear to be meadows of waving grass. A nearer view shows this to be an illusion, the broad fields of green being in reality impenetrable thickets of chaparral—a chaparral of unyielding manzanita and buck brush (*Arctostaphylos patula* and *Ceanothus velutinus*).

The chaparral described by Merriam (1899) is mostly found on south and west facing aspects and the foothills of the Shasta Valley. These areas are very different than the chaparral areas shown in figure 7. The latter areas in the Mount Shasta area are found on sites that are much wetter, more productive, and tend to be dominated by shrub tanoak (*Notholithocarpus densiflorus* (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh), Sierra chinquapin (*Chrysolepis sempervirens* (Kellogg) Hjelmqvist), or knobcone pine. Once an area is captured by chaparral, it is difficult for forests to recover due to strong competition for resources and the likelihood—which increases as the climate warms—that the chaparral stand will burn frequently enough to kill tree regeneration. Areas that become dominated by knobcone pine tend to reburn severely and then quickly regenerate to knobcone pine because of seed banks stored in serotinous cones that are released by fire (Keeley et al. 1999, Reilly et al. 2019).

Some of these extensive areas of montane chaparral probably existed prior to Euro-American colonial settlement. This is especially true for upper slope positions—particularly on warm aspects and where fires tended to run uphill—and along ridgelines, where soils were shallower and tree recruitment is more difficult. It is also the case that numerous chaparral areas established in the late 19th century as the result of fires set (intentionally or not) by early settlers, especially in areas of logging slash. In the early 1900s, Show and Kotok (1929) calculated that about 14.8 percent (422 900 ha [1,045,000 ac]) of what now comprises the Klamath, Shasta-Trinity, and Mendocino National Forests was covered in brush as a result of fire and logging. In 1948, southwestern Oregon was reported to have about 127 500 ha (315,000 ac) of nonstocked burns and old cutovers (areas clearcut before 1940) (Hayes 1959). In response to the public's objection to establishing forest reserves (today's national forests) in northern California, Gifford Pinchot stated the following: "It is true that natural reproduction is prolific in northern California, but vast areas of forest land have been converted into chaparral by wasteful cutting followed by repeated fires" (Potter 1905: 77). Haefner (1912) also remarked on how "the chaparral areas should not form the large acreage that they now do. . . .The charred stump, tree trunk, and fallen log tell plainly that fire was the cause, and has done its work." Haefner (1912) attributed these early fires primarily to Hudson Bay trappers in the area, followed by "Indians, miners, hunters, and stockmen." It is important to note, however, that many of the early foresters discounted lightning as a major driver of fire, assuming a priori that people were to blame. This may have led to overestimates of the extent of anthropogenic influences on the prevalence of chaparral fields.

Fire suppression over the past century has decreased forest openings and increased the distance between these openings, creating a more homogenous landscape.

Comparison to current—

Current landscape structure of YPMC forests in the assessment area has been largely driven by the alteration of ecosystem processes. Fire suppression over the past century has decreased forest openings and increased the distance between these openings, creating a more homogenous landscape (Skinner 1995). Logging on much of the landscape has further eliminated a great deal of the structural heterogeneity within stands through the selective removal of large (mostly pine) trees. The lack of fire has led to forest densification caused by the infilling of mostly shade-tolerant, less fire-tolerant tree species such as white fir and incense cedar. The Jeffrey pine- and California foothill pine-dominated forests found on less productive serpentine soils are exceptions to this trend. Increasing densities of smaller diameter trees on these serpentine sites are composed largely of species capable of tolerating the challenging edaphic conditions, and densities remain relatively low (Sahara et al. 2015).

The biggest concern regarding the densification and homogenization of YPMC forests today is the increased likelihood of large-scale forest biomass loss to fire, insects, and pathogens. At lower and middle elevations, YPMC forests are especially susceptible to biomass loss to fire because fuel levels continue to increase and warming climates are drying fuels, reducing snowpack, and lengthening the fire season (Miller and Safford 2012, Westerling et al. 2006). Miller et al. (2012b) found that mixed-conifer forests generally burned with a higher percentage of high severity when they remained long unburned (no fire since at least the early 20th century), and that conifer forests with small-diameter trees had a higher percentage of high-severity fire compared to forests with medium- to large-diameter trees. The implication is that areas in the assessment area that have not burned in the past 100 years and have experienced a large infilling of small-diameter trees, especially following logging, are much more likely to experience stand-replacing fire.

Mean and maximum fire size and total area burned in the assessment area have shown recent increases, driven primarily by widespread lightning events (Miller et al. 2012b), although, more recently, by human-caused fires. Miller et al. (2012b) found that periods with large areas of burning in concurrent fires occurring over months tended to result in relatively less severe fire effects because most days in the fire season are not extreme weather days. This suggests that wildfire, when permitted to burn under moderate weather conditions, may be an effective way to reintroduce some of the structural heterogeneity that has been lost because of logging and fire suppression. Under dry, hot, windy conditions, however, large wildfires in the assessment area may exhibit trends similar to those seen in YPMC forests in other parts of California (Miller and Safford 2012). The 2018 fire season resulted in several large, human-caused fires that burned large areas at high severity (i.e., Carr, Delta, and Klamathon Fires). Although the average severity of wildfires in the assessment area did not increase between 1984 and 2008, the total area of high-severity fire and the maximum size of high-severity patches have increased substantially since the mid-1980s (Miller et al. 2012b, Tepley et al. 2017). This trend is resetting succession and creating a very coarse-grained landscape, further reducing fine-scale heterogeneity across the landscape.

Future—

The important role that fire has historically played in driving landscape patterns in the assessment area suggests that future patterns will also be strongly dictated by fire trends, especially under climate warming. Using a range of different climate scenarios, Westerling and Bryant (2008) projected that the number of large fires could increase anywhere from 15 to 90 percent, depending on increases in temperature. It is also likely that increases in high-severity fire combined with rising climatic water deficit will impact the ability of mixed-conifer forests to recover after fire (Tepley et al. 2017). The combination of increased large, high-severity fires and continued fire suppression is likely to greatly "coarsen" the landscape forest structure over time, leading to large, hard-edged patches of different seral stages that were comparatively rare under historical/reference conditions.

At the state scale, Lenihan et al. (2008) used three different climate scenarios to model potential changes in vegetation throughout California. Under all simulations it was projected that extensive areas of evergreen conifer forest would be converted to forest types with a major hardwood component, shrubland, or grassland by the end of the 21st century, with the warmest and driest scenarios causing the largest potential loss of conifer forest. Lenihan et al. (2008) model broadly defined physiognomic vegetation types, and it is impossible to differentiate YPMC forests from other conifer forests in their outputs.

General Forest Structure

NRV—

Accounts from early explorers and surveyors as well as descriptions published in the early days of the Forest Service give valuable insight into the different components of stand structure in YPMC forests prior to Euro-American colonial settlement. Consistent throughout these early documents is the frequent mention of large trees and the visible effects of fire. It is also clear that stand structures were highly variable.

In discussing yellow pine stands, Sudworth (1908) described ponderosa pine as occurring in open stands with little to no understory as a result of frequent fire. Photographs taken in 1925 of YPMC stands in Shasta County, as part of Albert E. Wieslander's Vegetation Type Mapping (VTM) survey of California, demonstrate the open nature of these stands as well as the structural diversity present (fig. 8). Mostly pure stands of yellow pine were considered "semidense" to "open," according to Wieslander and Jensen (1946), with the trunks of mature trees rarely closer than ~10 m, and their crowns seldom touching (Sudworth 1908).

In a Forest Service document describing the control of forest fires, Cooper and Kelleter (1907) describe an area owned by the McCloud River Lumber Company in

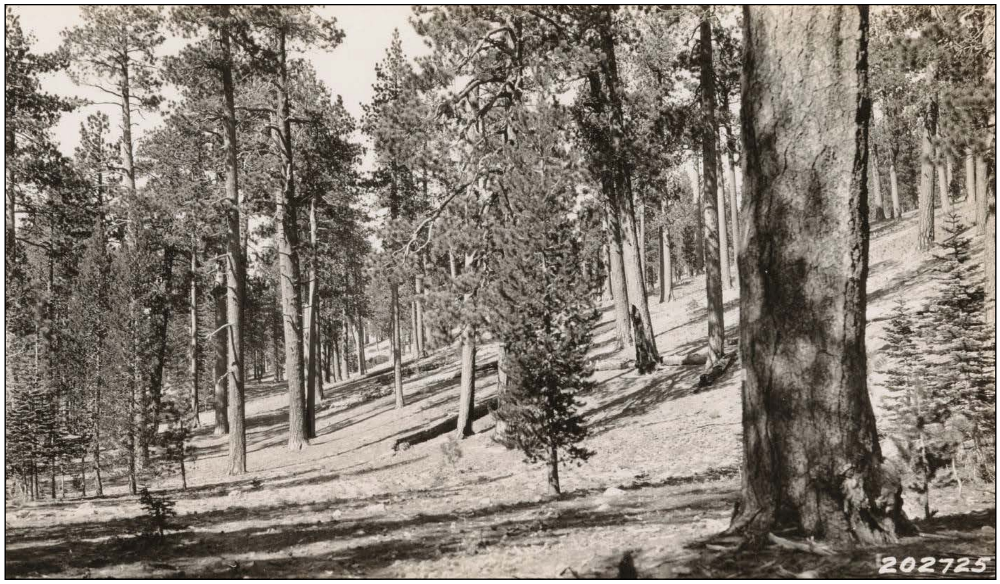

Figure 8—October 1925 open stand of yellow pine/mixed-conifer forest in Burney, California. Wieslander Vegetation Type Mapping Collection courtesy of the Marian Koshland Bioscience, Natural Resources & Public Health Library, University of California, Berkeley.

Siskiyou County as having two distinct types of forest: the "pine type" and the "fir type," separated largely by moisture availability and soil conditions. The pine type is described as occurring in rocky, dry areas, and composed of nearly pure stands of yellow pine with small amounts of white fir, Douglas-fir, and sugar pine mixed in. The fir type, on the other hand, is described as occurring in wetter areas with deep soils, and composed of both Douglas-fir and white fir, incense cedar, and some sugar and yellow pine mixed in.

Cooper and Kelleter (1907: 5–6) described the "oldest pine" type as "…of excellent quality, but [it] grows in rather open stand, a result of ancient Indian fires. Within the last sixty years, however, fires have done little damage in the virgin timber, although prevalent on the cut-over lands since lumbering began."

Cooper and Kelleter (1907: 7) also described how fire along with insects and diseases influenced stand structure in yellow pine forests and helped to maintain forest openings:

> The effect of fire on virgin timber is not always at once apparent. The mature trees, particularly yellow pine, are well adapted to resisting the effect of an ordinary ground fire, and apparently its chief effect upon the forest is the destruction of brush and litter. In reality, however, the trees are often seriously injured, particularly where fires follow one another at short intervals. Growth is checked and the trees weakened; and insect attacks and

> fungus diseases follow. In addition the trees are gradually eaten through at the base, and eventually die or are blown over. Openings thus made in the forest are effectually prevented by subsequent fires from coming up to young growth, while the chaparral, which sprouts from the roots and is not permanently eliminated, even though completely burned back, takes possession of the ground.

Cooper and Kelleter (1907: 6) described the fir type thusly: "The trees are large in size, particularly the [Douglas-fir] and yellow pine, and the forest is very dense. Fires have not been prevalent for many years in this type, and there is a dense undergrowth of white fir."

Mixed-conifer forests were highly variable in their density and composition. A VTM survey photograph taken in the late 1940s shows a dense stand of mixed-conifer forest containing ponderosa pine, sugar pine, Douglas-fir, and white fir in Seiad Valley on the Klamath National Forest (fig. 9). Although it is apparent that fire has not entered the stand for some time, an important characteristic to note is

Figure 9—November 3, 1949 view near Horse Creek Road about 7 miles north of Horse Creek post office. Pine-Douglas-fir type of old- and young-growth age class. From left, prominent trees are sugar pine, Douglas fir, and white fir. Trees in background are white fir, Douglas-fir, incense cedar, ponderosa pine, and sugar pine. Note the 52-inch sugar pine to the right in the background. This site had obviously not burned in some time. *Note person circled in red for scale. Wieslander Vegetation Type Mapping Collection courtesy of the Marian Koshland Bioscience, Natural Resources & Public Health Library, University of California, Berkeley.

the large variation in size class distributions present (note the very large sugar pine in the background of fig. 9). Sudworth (1908) described these mixed forests as having considerable young growth and brushy ground cover. White fir seldom formed pure stands on the landscape, but could make up three quarters of a mixed-conifer stand in the absence of fire. Similarly, incense cedar rarely formed pure stands on its own, but was common in mixed-conifer stands and could make up 50 percent of a given stand at times (Sudworth 1908). The heterogeneous nature of fire on the landscape allowed for shade-tolerant, fire-intolerant species, such as white fir and incense cedar, to become abundant in the understory of many mixed-conifer stands, occasionally avoiding fire long enough to become part of the established overstory.

During a geological survey of California led by one of the first California state geologists, Josiah Dwight Whitney, William Brewer journaled the following passage describing their travel along the west side of Mount Shasta on September 11, 1862 (Brewer 1930: 312):

> We soon take to the woods and follow a trail directly toward the mountain—the first four miles up a very gentle slope, among trees that must be seen to be at all appreciated—pines, firs, and cedars, all of species peculiar to the region west of the Rocky Mountains.…Fire had been through the woods and hundreds of trees had fallen, some this year, but more in past years.…These gigantic trees, straight as arrows, formed a magnificent forest. The last four miles was up a very steep slope, nearly a thousand feet per mile—part of the way through the pines, part through thick chaparral of manzanita.

Harrison G. Rogers traveled with Jedediah S. Smith of the Rocky Mountain Fur Company as a clerk during their northward journey through northern California and southern Oregon. On May 21, 1828, Rogers wrote what appears to be his only journal entry describing the composition of the forest and the size of the trees, while traversing through the mountains of northwestern California (Rogers 1918: 245):

> The timber in this part of this country is principally [Douglas-fir], pine, and white ceadar [sic], the most of the ceadar [sic] trees from 5 to 15 feet in diameter and tall in proportion to the thickness, the underbrush, hazle [sic], oak, briars, currents, gooseberry, and Scotch cap bushes, together with aldar [sic], and sundry other shrubs too tedious to mention; the soil of the country rich and black, but very mountainous, which renders the traveling almost impassable with so many horses as we have got.

In addition to the tremendous size of trees, often noted in early accounts was the important shrub component in YPMC forests. Shrubs such as *Arctostaphylos* spp. and *Ceanothus* spp. were commonly mentioned as present either in the understory of YPMC forests or in open patches where fire had been. It is clear, however, that the size and distribution of shrub patches were highly variable across the landscape.

Comparison to current—

Structural and compositional diversity in stands of YPMC forests has been reduced by harvest of most of the large tree component and nearly a century of fire exclusion in forests adapted to frequent low- to moderate-severity fires. Densities in mixed-conifer forests throughout California have increased along with the prominence of fire-sensitive species in these forests. The result has been a shift toward a more coarse-grained forest mosaic and overall structural simplification, evidence of which has been thoroughly described for mixed-conifer forests in the Sierra Nevada and southern Cascades (e.g., Parsons and Debenedetti 1979, Safford and Stevens 2017, SNEP 1996, Vankat and Major 1978), as well as, to a lesser extent, in the assessment area (e.g., Skinner 1995; Taylor and Skinner 1998, 2003). Because early suppression efforts may not have been as effective in the rugged, far reaches of the assessment area as similar efforts in the Sierra Nevada and southern Cascades, vegetation change associated with fire suppression may not be as advanced in certain areas (Stuart and Salazar 2000; Taylor and Skinner 1998, 2003).

Densities in mixed-conifer forests throughout California have increased along with the prominence of fire-sensitive species in these forests.

According to Beardsley and Warbington (1996), only about 13 percent of the total forest land (land covered at least 10 percent by crowns of live trees) in national forests in northwest California today can be considered old growth by meeting large (>30 inches [76 cm] in diameter), live tree minima (5 to 35 trees per hectare, depending on forest type and site class). About 10 percent meets the old-growth criteria when considering their minima for both standing dead trees and logs, and 8 percent meets the criteria when considering only "pristine" stands (i.e., undisturbed by humans), although this latter condition included areas where fire was absent during the 20 years preceding the study and does not adequately consider the role of historical fire frequencies in altering standing and downed dead material. Of all of the forest types considered by Beardsley and Warbington (1996), Jeffrey pine supported the most pristine old-growth remaining, relative to its total area. In the assessment area, Jeffrey pine occurs mostly on ultramafic/serpentine soils or in high-elevation sites on unproductive soils. Low levels of structural change on these sites are due to slow growth rates and the minimal impact of logging on such poor timber ground. The structural dynamics of these sites are driven primarily by limited nutrient availability, and thin tree cover leads to notable cover and diversity in the shrub and herb communities (Sawyer and Thornburgh 1977, Whittaker 1960).

As noted above, not all of these sites have resisted structural change over time, and wet maritime sites—especially on peridotite, an ultramafic rock that is more easily weathered than serpentinite—can support surprising amounts of biomass where fire and other disturbances do not periodically occur (Grace et al. 2007, Sahara et al. 2015).

Tree density—

NRV and comparison to current—Compared to the abundant studies on historical tree densities conducted in Sierra Nevada YPMC forests (Safford and Stevens 2017), studies in the assessment area are sparse (table 5). Even so, it is clear that the same trends of increasing tree densities prevalent in the Sierra Nevada have occurred in the assessment area as well. In comparisons between historical (1930s) VTM plots and contemporary (2000s) Forest Inventory and Analysis (FIA) plots, McIntyre et al. (2015) documented substantial shifts in tree densities across California. In the California portion of the assessment area, the authors found a decrease in large trees (>61 cm dbh) from an average of 30.6 trees per hectare (tph) to 16.7 tph, while small trees (10–30 cm dbh) increased from 229 tph to 412 tph on average. Overall densities of trees >10 cm dbh increased from 314 to 519.5 tph. It is important to note that the VTM inventory sampled very little of the assessment area, only the area around the Smith River in northwesternmost California (mostly Douglas-fir and mixed-evergreen forest), the low mountains encircling the upper Central Valley near Redding and extending to Mount Shasta, and a small area in the western part of the Mendocino National Forest (see supplemental fig. 3 in McIntyre et al. 2015). Some of these areas are not necessarily representative of the rest of the assessment area, however, due to the comparatively heavy logging and mining operations that were present along the Sacramento River corridor.

Historically, tree densities were highly variable in the assessment area. Based on stand reconstructions and historical data from in and around the assessment area, YPMC stands ranged in mean density from about 14 to 314 tph, with an average of 120 tph, depending on edaphic conditions and the proportions of Douglas-fir and white fir present in the stand (Hagmann et al. 2017, Hasel 1932, Leonzo and Keyes 2010, McIntyre et al. 2015, Metlen et al. 2013, Ritchie 2016, Sahara et al. 2015, Sensenig et al. 2013, Taylor and Skinner 2003). Leonzo and Keyes (2010) found the mean stand density for relict trees in an unlogged ponderosa pine/mixed-conifer forest to be about 42 tph (range 2 to 291 tph). Due to fire suppression, modern stand densities had increased to a mean of 425 tph (range 0 to 1,571 tph). The authors also determined that major densification began in the late 1930s and early 1940s based on the ages of encroaching trees (Leonzo and Keyes 2010). In a 1930s progress report for the Forest Service that evaluated the effects of logging on YPMC forests

Table 5—Historical and current tree densities in yellow pine and mixed-conifer forests in and near the assessment area

		Historical			Current			
Location	Forest type	Average density (range)	Diameter size class[a]	Time period	Average density (range)	Diameter size class	Time period	Reference
		Trees/ha	*cm*		*Trees/ha*	*cm*		
Klamath Mtns and North Coast Range	All forest types (includes Douglas-fir and mixed evergreen)	314	>10	1930s	520	>10	2000s	McIntyre et al. (2015)
Northern Klamath Mountains	Douglas-fir-mixed conifer	55	All	1800[b]	101 (90–160)	All	1990	Sensenig (2013)
Central Siskiyou Mountains	Douglas-fir-mixed conifer	103[c] (38–150)	≥37	1949–1951	—	—	—	Whittaker (1960)
Central Siskiyou Mountains	Mixed conifer (serpentine)	130[c] (76–272)	≥25	1949–1951	—	—	—	Whittaker (1960)
Eastern Siskiyou Mountains	Mixed conifer	124 (40–299)	≥10	1911[b]	435 (69–991)	≥10	2011	Metlen et al. (2013)
Western Klamath Mountains	White fir	371[d]	All	1961– 1963	425[d]	All	1993–1995	Talbert (1996)[e]
Southwestern Klamath Mountains	Mixed conifer	186 (19–655)	trees >100 years old[f]	c. 1900[b]	487 (85–1,900)	≥5	2000s	Taylor and Skinner (2003)
Southeastern Klamath Mountains	Ponderosa pine-mixed conifer	42 (2–291)	trees >100 years old[f]	—	467 (0–1,571)[g]	≥5	2000s	Leonzo and Keyes (2010)
Eastern Klamath Mountains	Ponderosa pine-mixed conifer	211	≥10	1910	—	—	—	Hasel (1932)
North Coast Range	Jeffrey pine savanna (serpentine)	14	All	1850–1939[b]	236	All	1940–2009	Sahara et al. (2015)
Southern Cascades	Douglas-fir-mixed conifer	57	All	1800[b]	98 (40–150)	All	1990	Sensenig (2013)
Southern Cascades	Dry mixed conifer	76 (25–296)	≥15	1914–1924	306 (62–796)	≥15	2014	Hagmann et al. (2017)
Southern Cascades	Moist mixed conifer	83 (25–245)	≥15	1914–1924	279 (75–796)	≥15	2014	Hagmann et al. (2017)
Southern Cascades	Ponderosa pine	248	≥8.9	1930s	916	≥8.9	2012	Ritchie (2016)
Southern Cascades	Ponderosa pine	48 (25–100)	≥15	1914–1924	279 (75–1,124)	≥15	2014	Hagmann et al. (2017)

[a] Estimates based on reconstructions include all trees old enough to be present during the historical time period and are likely an underestimate of historical stand density.

[b] Reconstructed from current stand.

[c] Conifers only.

[d] Stand density index rather than density.

[e] Reported in Stuart and Salazar (2000).

[f] Underestimate of historical stand density.

[g] Range is for encroachment trees only.

in the Klamath National Forest, stand densities prior to logging for all trees ≥10 cm dbh were 177 and 245 tph for the two plots surveyed. Ponderosa pine made up the majority of both stands, followed by incense cedar, Douglas-fir, white fir, and sugar pine (Hasel 1932).

Whittaker (1960) assessed (1) stem densities along a moisture gradient on both quartz diorite and ultramafic substrates and (2) distribution of trees in relation to elevation on quartz diorite in the central Siskiyou Mountains. These surveys were conducted between 1949 and 1951 in areas that had not been logged, but had experienced about half a century of fire suppression at the time of sampling. Stem densities for all sites were reported for stems >1 cm dbh. Densities for conifer stems ≥37 cm and sclerophyll (e.g., hard-leaved broadleaf species such as tanoak and live oak) stems ≥20 cm on quartz diorite, and for conifer stems ≥25 cm on ultramafics were also reported. On quartz diorite, overall stem densities averaged 1,790 stems/ha, yet the average density for conifers ≥37 cm was only 103 stems/ha and the average density for sclerophylls ≥20 cm was 88 stems/ha. Similarly for ultramafics, overall stem densities averaged 546 stems/ha but conifer stems ≥25 cm averaged 130 stems/ha. The high overall stem densities can be mostly attributed to smaller size classes, the abundance of which is likely a consequence of both the large proportion of sclerophyllous hardwoods present at this lower elevation and more than 50 years of fire suppression. When considering the elevation gradient, the sclerophyll component becomes largely nonexistent above 1370 m (4,500 ft) and the large (≥37 cm) conifer component increases (ranging from 87 stems/ha between 460 and 670 m [1,500 and 2,500 ft] to 212 stems/ha between 1920 and 2140 m [6,300 and 7,000 ft]) (Whittaker 1960).

Taylor and Skinner (2003) used fire scars and tree age classes to reconstruct historical fire regimes and forest structure in mixed-conifer forests with a higher prevalence of Douglas-fir and white fir. The authors separated plots at the landscape scale into groups based on the age class distribution of each species in 20-year age classes for stems >100 years old to identify groups of plots with similar prefire suppression age structures. The authors determined mean stem densities for all stems >100 years old as well as for stems in each age class group. Increases in stand densities over the previous 100 years ranged from 25 percent to more than 400 percent, depending on the group. It is important to note, however, that when using the density of stems >100 years old for comparison, younger trees that may have been present prior to fire suppression and did not survive to the present day are not accounted for. Nevertheless, the observed increases in stand density were driven largely by shade-tolerant species (Taylor and Skinner 2003).

Similar results were found in a study based in south central Oregon just to the northeast of the assessment area in which historical forest structure and composition data (1914–1924) were compared to current data (2014) (Hagmann et al. 2017). Tree density for all trees ≥15 cm dbh in historical ponderosa pine and mixed-conifer forests was 68 tph (range 25 to 296 tph). The current mean density is 293 tph, ranging from 62 to 1,124 tph. Hagmann et al. (2017) further separated the stands they analyzed into ponderosa pine, dry mixed-conifer, and moist mixed-conifer types. The ponderosa pine type had the lowest mean historical density (48 tph), followed by dry (76 tph) and moist mixed conifer (83 tph). Current mean densities for these three types are 279, 306, and 279 tph, respectively. One difference between the results of this study and that of other studies looking at shifts in stand densities is that the increase in small-tree densities was largely driven by ponderosa pine rather than white fir or Douglas-fir.

In a Jeffrey pine savanna located on serpentine soils at the western edge of the assessment area, Sahara et al. (2015) found a large increase in tree densities for Douglas-fir, Jeffrey pine, and Port Orford cedar associated with fire suppression. The mean density of trees for the 1850–1939 period (last recorded fire was in 1940) was 14.2 tph compared to 236.4 tph for the 1940–2009 period (Sahara et al. 2015). The results of this study contrast with the results of a study conducted by Damschen et al. (2010) that showed a decline in conifer density on serpentine soils between 1950 and 2007 at an interior site near the eastern edge of the assessment area. The differing results of these studies are likely due to the two study sites having very different climates, with the Damschen et al. (2010) study site receiving an approximate mean annual precipitation of 787 mm, compared to 2350 mm at the Sahara et al. (2015) study site. Furthermore, the proximity to the coast of the Sahara et al. (2015) study allows for important dry-season inputs from fog. Damschen et al. (2010) ascribed their decrease in conifer density to a warming and drying climate, which was also the cause of tree declines seen on serpentine sites during early Holocene warming in a paleoecological study by Briles et al. (2008).

There are some exceptions, but taken as a whole, modern forests contain much higher stem densities than they did during the presettlement period.

Based on the results of all the studies described above, it is clear that tree density has generally increased in the assessment area, with most studies ascribing the change to the long-term lack of fire. There are some exceptions, but taken as a whole, modern forests contain much higher stem densities than they did during the presettlement period.

Modern tree densities (trees >10 cm dbh) summarized from FIA plots located in the assessment area in yellow pine and mixed conifer forests average 169 tph (± 177 std dev) and 274 tph (± 228 std dev), respectively (USDA FS 2017a). This compares to an overall mean of about 400 tph in the modern estimates listed in table 5. The FIA modern values are lower than most of the modern densities in table 5 because the FIA data are an unbiased spatial sample across the entire California portion of the assessment area and include areas that have recently been disturbed by logging or fire (including clearcuts and stand-replacing fire). The modern tree density data in table 5 were all collected in stands that had not experienced fire or other major disturbances. In the FIA data, minimum trees per hectare for both yellow pine and mixed conifer across the landscape is 0, but maximum trees per hectare is 841 for yellow pine and 1,287 for mixed conifer, highlighting the high variability in density on the landscape. From table 5, the mean historical density is about 120 tph. If we compare this to the modern FIA numbers, density increases have ranged from about 40 to about 130 percent, whereas the direct local comparisons made in table 5 suggest an average density increase of about 230 percent, ranging from 60 to 900 percent. The more profound increase apparent in undisturbed forest stands is a better measure of the actual ecological trends involved, while the FIA-based comparison serves to indicate net density changes across the assessment area that takes into account the effects of more recent human and natural disturbance.

Future—Forest densities will probably continue to increase in most of the assessment area as conifer establishment continues and fires occurring under moderate fire-weather conditions continue to be suppressed. Increased tree mortality due to drought and disturbances, such as insect outbreaks and high-severity fire may counteract this to some extent, especially if increased drought conditions hinder the establishment of conifers after such events (Lauvaux et al. 2016, Skinner et al. 2018, Tepley et al. 2017).

Tree size and size class distribution—

NRV and comparison to current—Based on historical accounts, remnant old-growth, and stand reconstructions, average and maximum tree sizes in YPMC forests throughout the assessment area appear to have been larger in presettlement times than they are today. Tree sizes (height and diameter at breast height) for tree species prevalent in YPMC forests are described in detail by Sudworth (1908) and shown in table 6. While these sizes are not based specifically on individuals found in the assessment area, they provide an overview of what Sudworth considered to be common sizes of mature trees at the beginning of the 20th century. It could be argued, however, that Sudworth was only focusing on the largest individuals to give an idea of their growth potential when left undisturbed by fire. Nonetheless, in other accounts from the assessment area, reported tree sizes were mostly in agreement with Sudworth's reported height and diameters (Leiberg 1900, Merriam 1899), with the exception of estimates made in Brewer's 1862 account, which were often greater (Brewer 1930).

According to Brewer in 1862, cedars near Mount Shasta were over 30 m tall with diameters of 120 to 180 cm. Firs and sugar pines were often about 60 m tall

Plates

The following 21 photographs provide an overview of the variety of stand conditions found in yellow pine and mixed-conifer forests in the assessment area. The approximate locations, forest composition, and brief notes on site history are provided for each photograph.

Successional forest on Giant Crater lava flow

USDA Forest Service photo by Hugh Safford

Plate 1—Giant Crater lava flow erupted about 10,600 years ago. Photo shows slow rate of succession on high silica lavas in a dry environment. Near Jot Dean Cave, south flanks of Medicine Lake volcano, Shasta-Trinity National Forest, 1670 m (5,500 ft). Dominant trees include Jeffrey pine, lodgepole pine, incense cedar, and white fir.

Lightly thinned stand of mature mixed conifer

USDA Forest Service photo by Hugh Safford

Plate 2—Mature mixed-conifer stand about 7 years after light thinning. Biggest trees are around 100 cm (40 inches) in diameter. South flanks of Medicine Lake volcano, on much older lava flows (>100,000 years old), Shasta-Trinity National Forest, 1630 m (5,350 ft). Dominant species include Jeffrey pine, white fir, incense cedar, and some sugar pine.

Thinned and burned stand of mixed conifer

USDA Forest Service photo by Hugh Safford

Plate 3—Mixed-conifer stand that has been thinned and then burned within the last 5 to 10 years, McCloud Ranger District, Shasta-Trinity National Forest, 1230 m (4,050 ft). The stand is composed mostly of ponderosa pine with black oak in the understory (adults were probably girdled in the past) and some incense cedar mixed in. Understory consists mostly of serviceberry (*Amelanchier* sp.).

Fire-suppressed old-growth mixed-conifer forest

USDA Forest Service

Plate 4—Enriched mixed-conifer forest in the Sugar Creek Research Natural Area (RNA), Salmon Mountains, Klamath National Forest, 1500 m (5,000 ft). Sugar Creek RNA protects one of the most diverse conifer forests in the world, with 18 conifer species found in an area of 1 square mile (2.8 sq km). More than a century without fire has led to heavy accumulation of fuels and high densities of small-diameter trees from shade-tolerant species, such as white fir and Douglas-fir, putting old-growth stands at risk of burning at high-severity in the future.

Mixed-conifer forest

USDA Forest Service photo by Gabrielle Bohlman

Plate 5—Mixed-conifer forest on the Rogue River-Siskiyou National Forest near the California-Oregon border, 1430 m (4,700 ft). The area contains ponderosa pine, Douglas-fir, incense cedar, and sugar pine with hardwoods interspersed throughout. Most of this landscape has not seen fire in more than 100 years.

Fire-suppressed young mixed-conifer stand

USDA Forest Service photo by Andrew Mueller

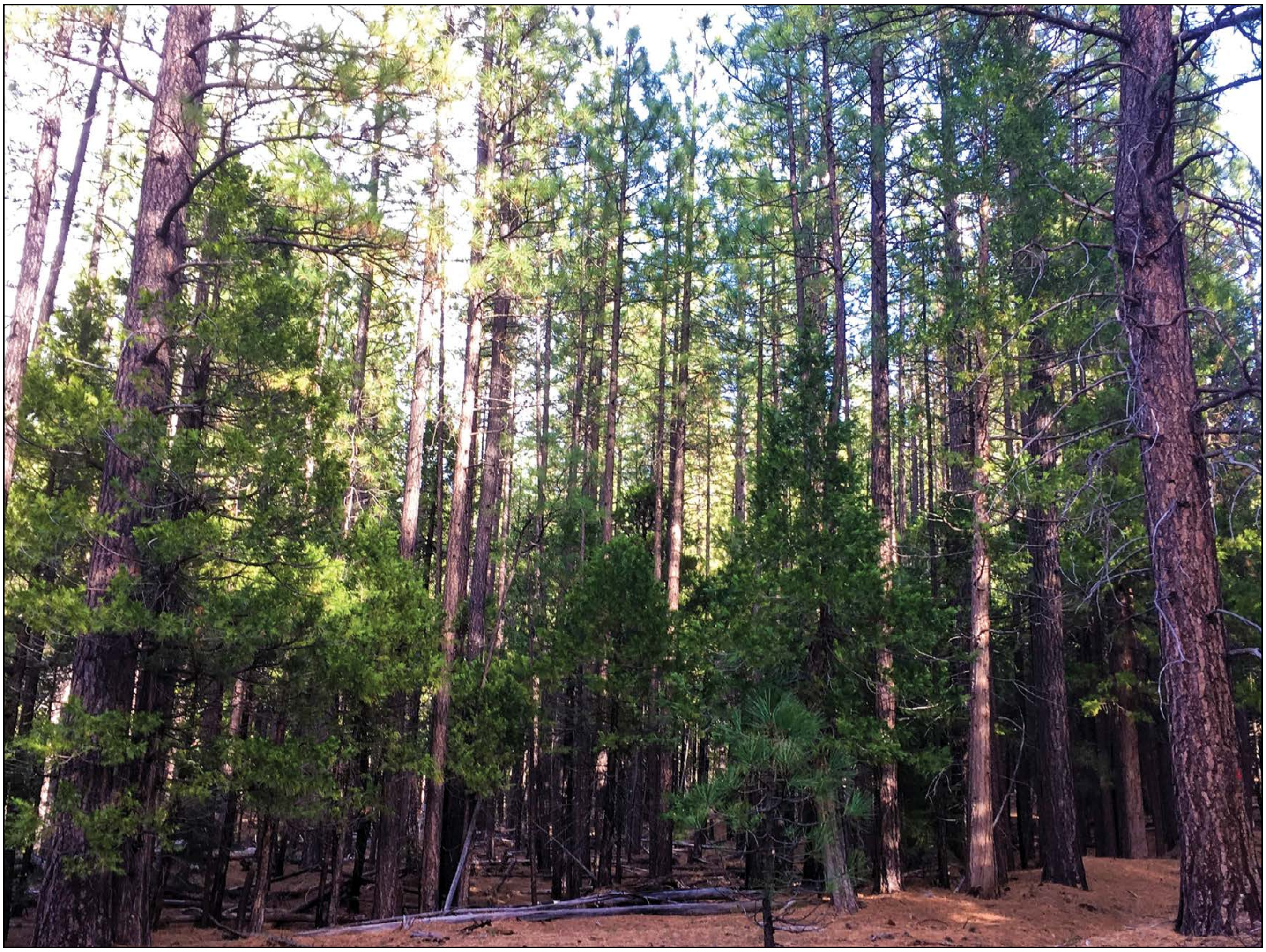

Plate 6—Dense mixed-conifer stand that experienced logging associated with extensive mining in the Kangaroo Creek area followed by fire suppression, Salmon/Scott River Ranger District, Klamath National Forest, 950 m (3,100 ft). Dominant conifers include ponderosa pine and incense cedar, with a dense ingrowth of incense cedar in the understory as a result of fire suppression. Site has not seen fire for more than 100 years.

Burned stand in mixed-conifer forest

USDA Forest Service photo by Julie Nelson

Plate 7—Mixed-conifer forest that burned at low to moderate severity 15+ years prior, Slate Creek watershed, north of Shasta Lake, Shasta-Trinity National Forest. This area reburned recently (after this photo was taken) in the 2018 Delta Fire. Dominant species include Douglas-fir, ponderosa pine, and black oak.

High-severity burn patch in mixed-conifer forest late-successional reserve

USDA Forest Service photo by Gabrielle Bohlman

Plate 8—The 2018 Ranch Fire burned through the southernmost late-successional reserves (LSRs) located on the Mendocino National Forest. One of the 100-acre LSRs (northern spotted owl activity centers) that burned almost completely at high severity. Dominant conifers in this stand included ponderosa pine, Douglas-fir, and black oak. Prior to the fire, this LSR had high fuel loading and high densities of small-diameter trees in the understory.

Late-successional reserve before and after the 2018 Ranch Fire

USDA Forest Service photo by Gary Urdahl

USDA Forest Service photo by Gary Urdahl

Plate 9—The top photo shows a mixed-conifer stand located in one of the southernmost LSRs on the Mendocino National Forest. Many of these areas burned in the 1996 Fork Fire and then reburned in the 2018 Ranch Fire. The bottom photo shows what many of these areas looked like immediately after burning in the Ranch Fire. The Ranch Fire burned so hot in some of these LSRs that it consumed nearly all of the down woody debris that was present after the Fork Fire.

Jeffrey pine and incense cedar woodland on ultramafic serpentine soils

USDA Forest Service photo by Julie Nelson

Plate 10—Jeffrey pine and incense cedar woodland on peridotite in the Rattlesnake Creek Terrane, southern Trinity County, east slope of Upper Hayfork Creek watershed, 1240 m (4,000 ft). Ultramafic soils often support very low biomass, which reduces burn frequency and severity.

Upper montane conifer forest on ultramafic serpentine soils

USDA Forest Service photo by Carl Skinner

Plate 11—Upper montane forest dominated by Jeffrey pine and incense cedar with huckleberry oak (*Quercus vaccinifolia*) and pinemat manzanita (*Arctostaphylos nevadensis*) in the understory. This forest is located along the Pacific Crest Trail near High Camp Creek above Cement Bluff Lake, about 2000 m (6,600 ft). Some of the large, old trees are more than 600 years old in this area.

Gap regeneration in mixed-conifer forest

USDA Forest Service photo by Gabrielle Bohlman

Plate 12—Gap regeneration in mixed-conifer forest about 6 miles east of Happy Camp, California, on the Happy Camp/Oak Knoll District, Klamath National Forest, 945 m (3,100 ft). Overstory includes Douglas-fir, incense cedar, ponderosa pine with madrone, Douglas-fir and ponderosa pine regeneration. This area burned at low severity in the 1987 China Fire and was the site of a prescribed burn in 2018.

Burn mosaic across landscape of mixed-conifer forest

USDA Forest Service photo by Gabrielle Bohlman

Plate 13—Landscape that burned in the 2016 Gap Fire just to the north of the Klamath River on the Klamath National Forest, 900–1260 m (3,000–3,800 ft). The patchy burn mosaic is typical of fires that burn under more moderate weather conditions in the Klamath Mountains. Many patches of high-severity fire are apparent and are larger than would have been expected under prefire suppression fuel loads.

Mixed-conifer forest with hardwood component

USDA Forest Service photo by Gabrielle Bohlman

Plate 14—Mixed-conifer forest about 4 km west of Seiad Valley, California, on the Klamath National Forest, 825 m (2,700 ft). Dominant conifers include ponderosa pine and Douglas-fir with a hardwood component that includes madrone and dogwood in the overstory. This site had not seen fire in the past 100 years.

Large patch of high-severity fire in mixed-conifer forest

USDA Forest Service photo by Gabrielle Bohlman

Plate 15—The 2014 Happy Camp Complex burned across this east-facing slope just to the south of the Klamath River, Klamath National Forest, 1070 m (3,500 ft). This photo was taken 5 years after fire and is emblematic of how these mixed-conifer systems tend to recover after high-severity fire. Note the abundant shrub cover and the resprouting hardwoods. Given the large distance to seed source in a patch like this, succession to conifers may take a half-century or more.

Low-severity fire effects in ponderosa pine forest

USDA Forest Service photo by Gabrielle Bohlman

Plate 16—Dry ponderosa pine forest 5 years after burning at low severity in the 2014 Little Deer Fire. This site is at about 1600 m (5,300 ft) in elevation, near Little Deer Mountain in the southern Cascades on the Klamath National Forest.

Large legacy ponderosa pine and treated stands nearby

USDA Forest Service photo by Carl Skinner

Plate 17—Stands originally dominated by large ponderosa pines, Douglas-fir, and sugar pine at approximately 1370 m (4,500 ft) elevation in the Ashland Watershed, Siskiyou Mountains, southwestern Oregon. Fabian Uzoh (USDA Forest Service) and Marty Main (consulting forester for City of Ashland) are shown for scale next to a large legacy ponderosa pine. Photo was taken when gathering fire scar samples used in Metlen et al. (2018). Fire scar data show these stands generally burned with median intervals of approximately 7 years or less before the fire suppression era. The photos below show similar nearby stands that have been thinned with fuels piled (left) and the condition of nearby stands several years after thinning and broadcast burning treatments (right).

USDA Forest Service photo by Carl Skinner

USDA Forest Service photo by Carl Skinner

Fire-suppressed moist mixed-conifer stand with legacy trees before and after treatment

The Nature Conservancy photo by Evan Barrientos

The Nature Conservancy photo by Evan Barrientos

Plate 18—Moist mixed-conifer stand with legacy trees located in the Ashland watershed on the Rogue River-Siskiyou National Forest, southwestern Oregon. This site is near the Windburn Ridge site included in Metlen et al. (2018). Elevation is approximately 1370 m (4,500 ft). Pretreatment species composition was dominated by Douglas-fir and white fir with scattered ponderosa and sugar pine legacy trees. These photos show the fire-excluded stand before (top) and after (bottom) commercial thinning and pile burning conducted by the Ashland Fire Resiliency project (ashlandwatershed.org). Mean and median fire-return intervals for this area from 1504–1887 were 15 and 12 years, respectively, with the last recorded fire in 1883.

Shrub response after high-severity fire

USDA Forest Service photo by Gabrielle Bohlman

USDA Forest Service photo by Gabrielle Bohlman

Plate 19—The 2014 Little Deer Fire burned at high severity across this northeast-facing slope of Little Deer Mountain, Klamath National Forest, 1700 m (5,600 ft). These photos were taken 5 years after the area burned and show the now shrub-dominated landscape. Very little conifer regeneration occurred in these areas because of long distances to living seed sources; hot, dry conditions; and competition for resources.

Conditions after repeated high-severity fire

USDA Forest Service photo by Eric Knapp

USDA Forest Service photo by Eric Knapp

Plate 20—This area initially burned at high severity in the 1999 Megram Fire and then reburned in the 2006 Backbone Fire. Large woody fuels generated in the first fire fueled the second fire. The lower photo shows seedlings of young conifers killed in the severely burned areas of the Backbone Fire that had been growing in the shrub stands.

Prescribed burn in thinned stand of mixed-conifer forest

The Nature Conservancy photo by Evan Barrientos

Plate 21—Prescribed burn in a thinned stand of mixed-conifer forest located in the Ashland watershed on the Rogue River-Siskiyou National Forest, southwestern Oregon. This site is near the Horn Gap site included in Metlen et al. (2018). Elevation is approximately 1420 m (4,700 ft). Dominant tree species prior to treatment included Douglas-fir and white fir with ponderosa and sugar pine legacy trees. The historical mean and median fire-return intervals for this area were 7 and 8 years, respectively, with the last recorded fire in 1910.

Table 6—Height and diameter at breast height for major tree species in yellow pine and mixed-conifer forests

Tree species	Height		Diameter	
	Meters	*Feet*	*Centimeters*	*Inches*
Yellow pine	38–43	125–140	90–120	36–48
Sugar pine	49–55	160–180	120–210	48–84
Incense cedar	23–27	75–90	76–122	30–48
White fir	43–55	140–180	110–152	42–60
Douglas-fir	55–58	180–190	110–180	42–60

Measurements given are those for ordinarily found individuals according to Sudworth (1908).

with some sugar pines reaching over 70 m or more in height. Brewer measured a tree that had fallen and found its base diameter to be about 210 cm and its length from the base to the point at which the tip had burned off to be about 70 m (Brewer 1930).

Whittaker (1960) postulated that three historical phases in the northern assessment area (presettlement; postsettlement, but prefire suppression; postfire suppression) had different effects on the relationship between tree size classes and stem density (i.e., the size class distribution): (1) the presettlement period, represented by a convex curvature for the largest size classes, where frequent low-severity fire would have been common on the landscape; (2) the historical period, represented by a "flattened" intermediate section caused by an increase in the mortality of young trees due to severe fires provoked by Euro-Americans; and (3) the more recent fire suppression period, represented by a concave curvature in the distribution (the so-called inverse J-shaped curve), with the smallest size classes reaching high densities because of greater rates of young tree survival in the absence of fire. Whittaker (1960) considered the stands included in his study to be a "fire-adapted vegetation of a summer-dry climate in which fires of varying frequencies and intensities and varying sources—white man [Euro-American], Amerind [American Indian], and lightning—have for a very long time been part of its environment."

Size class distributions were reported for two stands before and after selective logging in 1910 in a report produced by Hasel (1932). The initial distribution exhibited a fairly gradual decline between small and large trees and shifted to a steeper decline through the removal of the largest trees in the stand. Both plots were considered to be low quality because of relatively low precipitation and poor soil conditions in the area, and the last severe fire in the area had occurred 15 years prior to logging. In the initial stand, the smallest size class that was included was

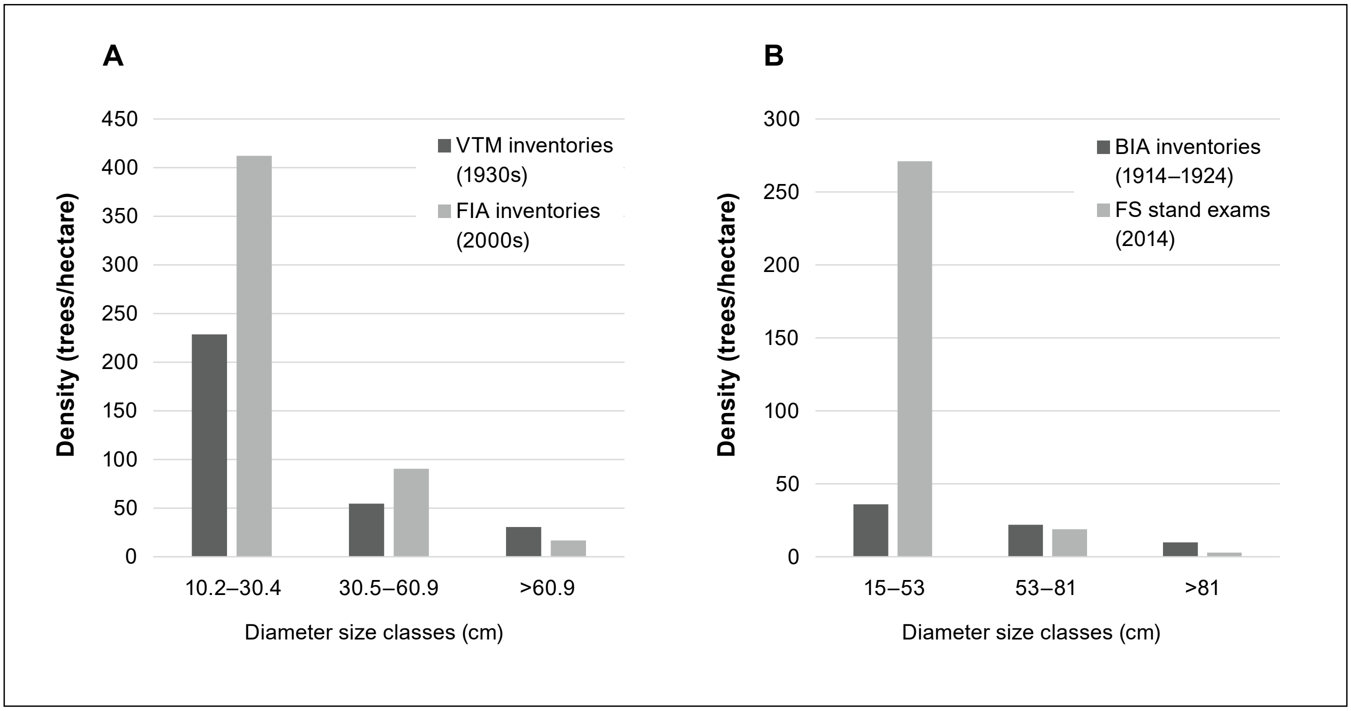

Figure 10—Size class distributions of trees from two comparisons of early 20th century inventories with early 21st century inventories. (A) Distributions sampled by the Forest Service Vegetation Type Mapping inventory in the 1930s vs. the Forest Inventory and Analysis inventory in the early 2000s in assessment area plots; (B) distributions sampled by Bureau of Indian Affairs inventories between 1914 and 1924 vs. Forest Service sampling from 2014 in a 39 000-ha area in the southern Oregon Cascade Mountains. Source: McIntyre et al. (2015).

4 to 11 inches (10 to 28 cm) and comprised the majority of stems on both plots (42.5 and 56.9 percent); trees >24 inches (61 cm) in diameter were 12.4 and 23.4 percent of stems in the two plots. After selective cutting, larger trees made up 3.5 and 10.6 percent, respectively, while the smallest size class increased to 56 and 67 percent of stems in the two plots.

McIntyre et al. (2015) compared Forest Service inventories in the 1930s and the most recent FIA inventories from the early 2000s, without differentiating by forest type. They found decreases in large trees (>61 cm dbh) in the assessment area along with substantial increases in small trees (10 to 30 cm dbh), leading to a notably more inverse J-shaped distribution of size classes (fig. 10a). Hagmann et al. (2017) carried out a more local comparison of historical (1914–1924) inventory data with current conditions in an area of ponderosa pine and mixed conifer just northeast of the assessment area. They found mean densities of very large trees (≥ 81 cm dbh) had decreased from 10 to 3 tph, while smaller trees (15 to 53 cm dbh) had increased from 36 to 271 tph. As with McIntyre et al. (2015), the overall distribution of size classes shifted from a flat decline between small and large trees in the early 1900s to a very steep decline (an inverse J-shaped curve) today (fig. 10b).

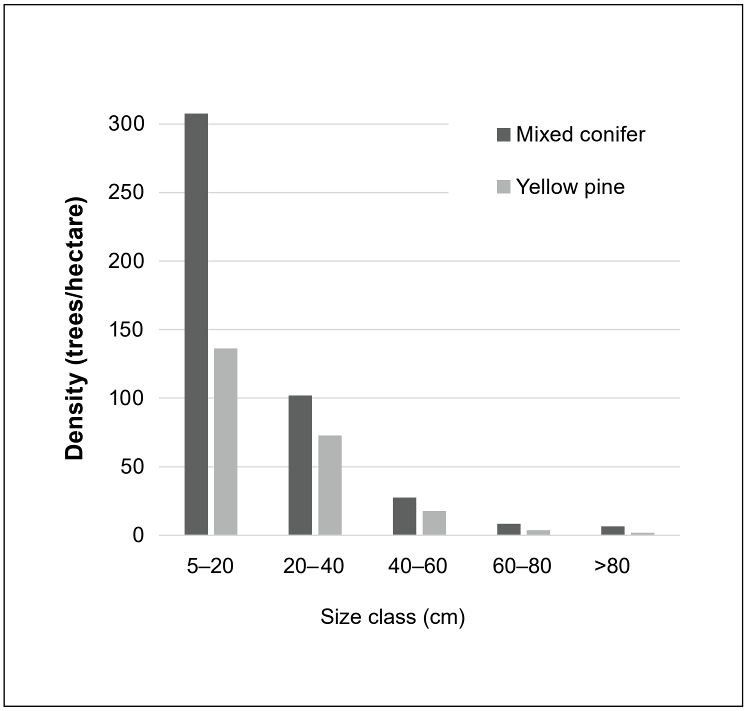

Figure 11—Current average size class distribution of trees in yellow pine and mixed-conifer forest in the California portion of the assessment area based on FIA plot data (USDA FS 2017a).

Figure 11 shows the current average size class distribution of trees in YPMC forests within the assessment area based on FIA plot data (USDA FS 2017a). Although the number of historical and reference sources is limited in the assessment area, the evidence is relatively clear that in YPMC and related forest types, (1) average tree size is much smaller in the modern forest than in the presettlement forest; (2) the number of small trees has increased tremendously over the past century or more, with many areas showing two to five times higher small-tree densities, while at the same time the number of large trees has decreased greatly; (3) the overall distribution of tree size classes vs. density has shifted from a relatively flat curve (relatively small differences in density between tree size classes) to a very steep curve, dropping from high numbers of small trees to low numbers of large trees.

If current trends continue, size class distributions will likely be further skewed toward smaller diameter trees as a result of high recruitment of shade-tolerant, less fire-tolerant species and mortality of larger trees because of increased competition, drought stress, and bark beetle-driven mortality.

Future—If current trends continue, size class distributions will likely be further skewed toward smaller diameter trees as a result of high recruitment of shade-tolerant, less fire-tolerant species and mortality of larger trees because of increased competition, drought stress, and bark beetle-driven mortality. A possible exception to this may be in situations where high density-dependent mortality of smaller trees occurs (e.g., after low- and moderate-severity fire).

Tree basal area—

NRV and comparison to current—Direct comparisons between current forest stands and reconstructed presettlement stands at the same locations show that modern basal areas in assessment area YPMC forests are notably higher than presettlement basal areas (table 7). The apparent increase in basal area can be attributed to the strong increase in small-diameter trees, which more than compensates for declines in basal area due to losses in large-diameter trees. Hagmann et al. (2017) found that the percentage of basal area comprised by large trees (≥53 cm dbh) dropped from a mean of 83 percent to 26 percent between the beginning of the 20th century and the beginning of the 21st century. Drier ponderosa pine sites had a larger drop in basal area present in large trees (86 to 14 percent) than in the moist mixed-conifer forest sites (81 to 33 percent), but both indicate a clear shift in basal area distribution from large trees to small trees. Leonzo and Keyes (2010) studied a fire-suppressed relict forest and found that 54 percent of current basal area was in the large relict trees and 46 percent was in trees that have entered the stand since fire suppression. Additionally, the percentage of basal area composed of young "encroaching" trees was found to be higher at lower elevations, suggesting a greater level of change in elevations below 1000 m.

In their California-wide study, McIntyre et al. (2015) found that basal areas in their North Coast region (most coincident with the assessment area) had slightly increased between the 1930s and 2000s from a mean of 32 to 35 m^2/ha (note that McIntyre et al.'s analysis grouped all forest types). Based on FIA data, yellow pine and mixed-conifer forests in the assessment area currently support mean basal areas for all trees >5 cm dbh of about 12 and 23 m^2/ha, respectively, with a range of 0 to 46 m^2/ha for yellow pine and 0 to 154 m^2/ha for mixed conifer. As noted previously, FIA samples all stand conditions, including postdisturbance. Unfortunately, we do not have an assessment area-wide dataset for YPMC forests from the presettlement or historical periods.

***Future*—**If current trends continue, basal area may continue to increase in the assessment area, especially in more productive areas. Alternatively, as the footprint and severity of disturbances increase, there may be a decrease in basal area across the landscape, or at the very least, a more coarse-grained distribution of biomass as large high-severity fires cause large reductions in tree biomass and biomass in fire-suppressed areas continues to increase.

Table 7—Historical and current basal area in yellow pine and mixed-conifer forests in and around the assessment area

		Historical			Current			
Location	Forest type	Basal area	Diameter size class[a]	Time period	Basal area	Diameter size class	Time period	Reference
		m²/ha	*cm*		*m²/ha*	*cm*		
Klamath Mountains and North Coast Range	All forest types (includes Douglas-fir and mixed evergreen)	32	>10	1930s	35	>10	2000s	McIntyre et al. (2015)
Northern Klamath Mountains, Southern Cascades, and mid-Coastal	Douglas-fir-mixed conifer	20–35	All	1900[b]	43–59	All	1990	Sensenig (2013)
Eastern Siskiyou Mountains	Mixed conifer	18.6 (3–52)	>10	1911[b]	45 (14–78)	>10	2011	Metlen et al. (2013)
North Coast Range	Jeffrey pine savanna	0.01	>7	1870[b]	20	>7	2009	Sahara et al. (2015)
Southern Cascades	Dry mixed conifer	19 (4–59)	≥15	1914–1924	28 (7–71)	≥15	2014	Hagmann et al. (2017)
Southern Cascades	Moist mixed conifer	20 (7–43)	≥15	1914–1924	31 (7–71)	≥15	2014	Hagmann et al. (2017)
Southern Cascades	Ponderosa pine	21	≥8.9	1930s	—	—	—	Ritchie (2016)
Southern Cascades	Ponderosa pine	10 (3–16)	≥15	1914–1924	22 (8–46)	≥15	2014	Hagmann et al. (2017)

[a]Estimates based on reconstructions include all trees old enough to be present during the historical time period and are likely an underestimate of historical densities because smaller trees cannot be accounted for.

[b]Reconstructed.

Canopy cover—

NRV and comparison to current—Although early descriptions of YPMC forests focused more on the diameter and height of trees rather than their canopy cover, 19th and early 20th century observers often remarked on the overall condition of these forests. Based on these descriptions, it is clear that canopy cover across the landscape was highly variable. Yellow pine forests were frequently noted for their open nature, which was often attributed to frequent fire (Cooper and Kelleter 1907, Leiberg 1900). Leiberg (1900) writes, "the open character of the yellow-pine type of forest anywhere in the region examined is due to frequently repeated forest fires more than to any other cause…" But even in these predominantly open stands, pockets of denser forest, often Douglas-fir and white fir dominated, could be found near drainages and other areas of deeper soil or higher soil moisture. Moist mixed-conifer forests, on the other hand, were described as denser than yellow pine-dominated forests and supporting higher percentages of true fir and Douglas-fir (Cooper and Kelleter 1907). Figure 8 shows the openness of the canopy in yellow pine-dominated stands, while figure 9 shows the higher canopy cover and greater structural diversity in more mixed stands. It is important to note that at the time these photographs were taken (1930s and 1940s) fire suppression had been occurring for at least a couple of decades, potentially influencing the density of smaller trees and the groundcover.

Modern tree canopy cover[1] in the assessment area based on FIA data (USDA FS 2017a) is 23.1 percent (± 18.9 percent) for yellow pine forests and 37.4 percent (± 23.9 percent) for mixed-conifer forests. FIA crews do not directly measure cover in the field; these estimates are modeled from the tree lists for the FIA plots using Forest Vegetation Simulator (FVS) equations in Crookston and Stage (1999), and are generally assumed to be underestimates of the true values (e.g., Fiala et al. 2006). Another important factor driving the low cover values in the modern FIA dataset is the fact that we do not include Douglas-fir-dominated stands in our analysis. Douglas-fir-dominated forest grows in moist, high-productivity sites and supports the highest canopy covers in the mixed-conifer forest type (mean = 60 percent cover in the FIA dataset).

Beardsley and Warbington (1996) used the same FVS equations to estimate canopy cover in uncut old-growth stands throughout the assessment area. They estimated average canopy cover for trees ≥12.7 cm dbh at 34, 42, and 49 percent, for Jeffrey pine, mixed conifer, and white fir, respectively. Canopy cover for trees ≤12.7 cm dbh was 14 percent for Jeffrey pine, 17 percent for white fir, and 28 percent for mixed conifer (Beardsley and Warbington 1996). These stands were considered “pristine” old growth in that they met certain old-growth criteria and did not experience human disturbance. It is likely, however, that fire suppression has affected these old-growth stands allowing the canopy cover, especially from smaller trees, to increase. It is important to note that the Beardsley and Warbington (1996) dataset included Douglas-fir-dominated stands in their mixed-conifer class.

We do not have assessment area-wide estimates of canopy cover for YPMC forest from historical or reference sources, but a number of studies have done direct local comparisons, and FVS equations can be used to make ballpark estimates from other local studies as well. For example, Hagmann et al. (2017) assessed the historical extent of suitable northern spotted owl habitat in an area of YPMC forest just northeast of the assessment area. They found that only about 5 percent of the study area had canopy cover ≥30 percent in 1914, but by 2014, the same study area supported ≥30 percent canopy cover over two-thirds of its area. Taylor

[1] Relative = uncounted overlapping canopies; max canopy cover = 100 percent.

(2010) reported size class distributions from a reference (uncut and with frequent 20th century fire) YPMC forest in the Ishi Wilderness on the Lassen National Forest, about 70 km south-southeast of Shasta Lake in the assessment area. Safford and Stevens (2017) used FVS equations to calculate a mean canopy cover from Taylor's (2010) data for these stands of about 25 percent. Additionally, Ritchie (2016) reported that 90 percent of observations made in a 1930s plot-based census in northeastern California found crown area to be low, ranging from 13 to 35 percent at the 1-ha scale.

Future—Assuming that current trends in forest densification and fire exclusion continue, canopy cover will likely continue to increase across YPMC forests. This will be offset to some extent if forest mortality agents, such as fire, insects, and disease, increase.

Forest gaps and tree clumps—

NRV and comparison to current—Openings in the forest canopy, or "forest gaps," are an important component of fine-grain heterogeneity in forest vegetation structure. The location and size of a forest gap is generally determined by environmental limitations, such as edaphic conditions that restrict the growth of trees in certain areas, or by tree fall, which may be caused by disturbances, such as wind, fire, or disease. Gaps are smaller features of the landscape that exist within a landscape-level mosaic of "patches." Fires that result in high-severity patches create forest openings on the landscape, the extent of which contributes to coarse-grain heterogeneity. Presettlement YPMC forests in the assessment area supported relatively low canopy cover and a high degree of fine-grained heterogeneity. The lack of clear age cohorts of trees in YPMC forest stands prior to fire suppression, coupled with the presence of stems in a wide range of age classes indicate that successful recruitment was largely driven by opportunities created by low- and moderate-severity fire, insect- and disease-driven mortality, and in some cases windthrow, and not by severe, coarse-grained disturbances such as high-severity fire (Leiberg 1900, Skinner et al. 2018, Stevens et al. 2016, Taylor and Skinner 2003).

The counterpart to forest gaps are tree clumps, an aggregation of individual trees within a stand. Tree clumps are often composed of trees that initiated after the same disturbance event, but can also be aggregates of multiaged trees that initiated at different times. Climate anomalies, in which a period of cool, wet conditions causes a hiatus in fire activity, create another possible pathway for patches of even-aged regeneration to establish in these forests, especially when such an anomaly aligns with good cone crops and lasts long enough for trees to grow large enough to survive subsequent fires (Brown 2006, Brown et al. 2008, Skinner et al. 2018). Larson and Churchill (2012) compiled information from dozens of studies to describe the characteristic spatial patterns of trees in frequent-fire forests throughout western North America. Three key elements were identified: forest gaps, tree clumps, and widely spaced single trees. This structural mosaic is typically present at the stand level (<0.4 ha), but in forests that exhibit a higher percentage of moderate-severity fire (i.e., moist mixed conifer) this can increase to 4 ha or more (Larson and Churchill 2012). Old-growth forests in the assessment area exhibit tree clumping at fine spatial scales and random spatial arrangements at coarser scales, according to a study by van Mantgem and Sarr (2015). Other studies in the assessment area have shown how stand development patterns of frequent-fire (FRI of 13 to 22 years), Douglas-fir-dominated forests are controlled by low- to moderate-severity fires (Agee 1991, Wills and Stuart 1994). Spatially complex fires historically created a fine-grained forest matrix with variably sized openings that helped dictate regeneration patterns (Taylor and Skinner 1998).

Table 8 in Safford and Stevens (2017) provides a summary of the mean sizes of forest gaps and tree clumps for historical and contemporary reference YPMC forests across much of the Southwestern United States (including California and central and eastern Oregon). On average, mean tree clump sizes ranged historically from 0.022 ha to 0.19 ha (includes regeneration patches), and average forest gaps ranged from 0.095 ha to 0.69 ha. These values are likely more relevant to the more eastern and southern portions of the assessment area in yellow pine and dry mixed-conifer forests, as the western and northern portions of the assessment area are wetter, include more Douglas-fir, and are thought to have included a higher proportion of moderate-severity fire. Such conditions would be expected to lead to somewhat coarser spatial patterns of mortality in fires, and hence somewhat larger gap sizes.

Overall, there was a 39 percent decrease in the area occupied by openings and it was observed that the forest canopy in stands adjacent to openings had also experienced an increase in density.

In a comparative study of the spatial characteristics in forest openings in the Klamath National Forest between 1944 and 1985, Skinner (1995) determined a mean gap size of 0.53 ha with a range of 0.09 to 297.8 ha for 1944. These forest openings were determined based on aerial photographs and included both ephemeral and more stable openings (those controlled by the physical environment rather than disturbance). The larger openings found in Skinner (1995) in the 1944 photos were noted to have had scattered groups of trees within them and varied greatly in their shape. The result of the comparison between 1944 and 1985 showed a decrease in the size of openings and increased distances between openings. It was noted that the reduced size of openings came partly from the closing off at the "fingers and lobes" of the bigger patches. Overall, there was a 39 percent decrease in the area occupied by openings and it was observed that the forest canopy in stands adjacent to openings had also experienced an increase in density (Skinner 1995).

In a report on the stand conditions in two YPMC stands in the early 20th century, Hasel (1932) describes the trees as being "irregularly distributed over the area in groups." He also noted that the distribution of trees after logging remained in a "grouped arrangement due to the character of the original stand," which indicated the strong grouping pattern. In a similar type of stand, Cooper and Kelleter (1907) also observed multiple forest openings that had been densely seeded. The presence of chaparral in openings was also noted, but the shrubs were apparently not dense enough to prevent pine seedlings from overtopping them.

Although information about historical spatial characteristics of YPMC forests throughout the assessment area is mostly qualitative, it can be inferred—based on knowledge of historical fire frequencies and stand densities as well as from studies outside of the assessment area—that current forests have fewer gaps and larger tree clumps than they did prior to the fire suppression era. Fine-grained heterogeneity has decreased as a result of increasing tree densities, and increases in large high-severity fires on the landscape have further promoted coarse-grained heterogeneity, creating a mosaic of larger, more defined patches across the landscape. Even in the early 20th century there was an understanding of the effect of eliminating fire from these forests. Show and Kotok (1924: 25) discussed how the virgin forest would respond to eliminating fire: "…in timber that will not be cut over for many years to come, the advance reproduction will fill in the blanks of the present forest.…Even in comparatively fully stocked stands under protection, gaps may thus be filled, as single mature or overmature individuals succumb."

Future—If current trends continue, forest gaps will continue to decrease in size and number, and tree clumps will continue to increase in size and density as forest densification proceeds. The reintroduction of low- to moderate-severity fire could help to mitigate this, yet the deficit of this type of fire across the landscape is so great that it will take much time to diminish the effects of fire suppression. Furthermore, the increase in large high-severity patches, which eliminate large swaths of overstory trees, is creating long-lasting, coarse-scale patches of early-seral vegetation on the landscape and further restricting opportunities to reintroduce fine-scale heterogeneity into these forests. Incorporation of fine-scale spatial heterogeneity during reforestation efforts may help to increase forest resistance to stand-replacing fire in the future (North et al. 2019).

Snags—

NRV and comparison to current—There is rather limited information available regarding historical densities and spatial distributions of snags (standing dead trees) in the assessment area. With the information that is available, it is clear that

In frequent-fire forests in which tree clumps were variably spaced across the landscape, fire-induced tree mortality was typically limited to younger trees, with large, older trees succumbing to insect and disease more often than fire.

densities were highly dependent upon the type of forest (i.e., dry vs. moist mixed conifer) and reigning ecological processes in that forest (Agee 2002). In frequent-fire forests in which tree clumps were variably spaced across the landscape, fire-induced tree mortality was typically limited to younger trees, with large, older trees succumbing to insect and disease more often than fire (Agee 2002). Instances of high-severity fire certainly also produced large snags, but in open yellow pine and dry mixed-conifer forests, higher severity fires were historically infrequent. In moist mixed-conifer forests, patches of high-severity fire were likely a more significant contributor to snag abundance on the landscape with the distribution driven mostly by topography and interactions with weather (Skinner 2002). Insect outbreaks also contributed to snags on the landscape, but these tended to be relatively isolated, minor events in yellow pine forests, according to Munger (1917), who suggested it was typical to see groups of "three to ten or more dead trees in a clump" at varying levels of decay.

In a paper discussing fuel and snag abundance in unlogged, unburned Jeffrey pine/mixed-conifer stands in northwestern Mexico, Stephens (2004: table 4) compared snag densities from pine-dominated conifer forests throughout western North America. Based on this comparison, it is apparent that modern fire-suppressed forests have, in almost all cases, substantially higher snag densities than forests with an active fire regime. While Stephens (2004: table 4) does not include information from the assessment area, similar trends can be seen in the fire-suppressed YPMC forests of northwestern California and southwestern Oregon. In an attempt to assess the effects of fire suppression on relict ponderosa pine, mixed-conifer stands in the southeastern Klamath Mountains, Leonzo and Keyes (2010) quantified snag densities and found an average of 35 snags/ha and a range of 0 to 713 snags/ha. This average, while on the lower end, is similar to those reported for fire-suppressed forests in Stephens (2004). Snag diameters in the southeastern Klamath Mountains ranged from 5 to 66 cm with a median of 10 cm. The low median size of these snags combined with their relatively high density is indicative of the effects of over a century of fire exclusion (Leonzo and Keyes 2010).

In assessing stand conditions in old-growth forests throughout the assessment area, Beardsley and Warbington (1996) quantified snag densities broken down by size class (small: 25–45 cm, medium: 50–70 cm, large: >70 cm) for widespread forest types in the region. For mixed conifer, the average density of small snags was 12.4 snags/ha, while the average density of medium and large snags combined was 7.4 snags/ha. Jeffrey pine stands had low snag densities of about 2.5 small snags/ha and a combined average of 4.9 snags/ha for medium and large snags. For interior ponderosa pine stands, the authors only provide a mean snag density for the two larger size classes (2.5 snags/ha) because of the limited number of plots that qualified as old growth. For white fir stands located in the Mendocino and Shasta-Trinity National Forests and the eastern Klamath National Forest, small snags averaged 14.8 snags/ha, while medium and large snags combined averaged about 17.3 snags/ha. White fir forests closer to the coast (in the Six Rivers National Forest and the western Klamath National Forest) had a small snag density average of 42 snags/ha and a combined medium and large density average of 22.3 snags/ha. These snag densities have likely been influenced by fire suppression, but are generally lower than would be expected based on values provided in Stephens (2004), with the exception of more coastal white fir forests.

Most other studies that provide estimates of historical snag abundance in YPMC forests are based on reference stands in the eastern Cascade Range, Sierra Nevada, or Sierra de San Pedro Mártir (a mountain range in northwest Mexico often used as a reference for historical YPMC forest conditions because logging and fire suppression have been absent until recently), of which discussion and summaries can be found in Safford and Stevens (2017). Average snag densities for snags >15 cm dbh ranged from 4.4 snags/ha in the reference forests in the Sierra de San Pedro Mártir (Stephens 2004) to 8 to 12 snags/ha in old-growth pine forests located in the eastern Cascade Range (Youngblood et al. 2004) and 14 to 36 snags/ha based on estimates made for presettlement yellow pine forests in eastern Washington, which did not account for fire consumption of dead trees (Harrod et al. 1998).

Current snag densities (all snags >15 cm dbh) summarized from forest inventory and analysis (FIA) plots show an average of 8.79 snags/ha (± 34.24 std dev) in yellow pine forests and 25.13 snags per/ha (± 40.16 std dev) in mixed conifer (USDA FS 2017a). These averages fall between the snag densities found by Beardsley and Warbington (1996) and Leonzo and Keyes (2010) in old-growth stands throughout the assessment area and are slightly higher than would be expected in presettlement stands. The prominence of more moderate-severity fire regimes in western portions of the assessment area may have led to higher snag densities historically, although the presence of more frequent fire would have also increased snag fall and consumption rates, and patches of mortality would have been at a finer scale than we are seeing with modern fires. In summary, it appears that snag densities in modern assessment area YPMC forests are similar to or higher than snag densities in reference forests.

Future—If we continue to see an increase in the area of high-severity fire and the size of large, high-severity patches in YPMC forests, not only will the abundance of snags increase, but their distribution will be skewed toward being concentrated in large, severely burned areas. Once these snags fall, however, we may begin to see an overall decrease over the long term. Increased drought conditions as a result of warming, combined with high tree densities, also make these forests vulnerable to insect outbreaks that are capable of killing large areas of drought-stressed trees. Although the assessment area has not experienced the same extent of beetle mortality as the central and southern Sierra Nevada, there have been increases in YPMC mortality associated with recent drought and beetle activity, and it is likely that densifying forests will continue to see elevated levels of mortality and thus, snag densities.

Coarse woody debris—

NRV and comparison to current—The presence and distribution of coarse woody debris (CWD) on the landscape is inherently connected to fire and snag dynamics on the landscape. CWD is typically defined as dead and down material with a diameter >7.6 cm (Brown 1974). Fire in a stand can increase the abundance of CWD on the ground by killing trees and destabilizing snags; however, fire also aids in the decomposition of woody debris, especially in drier areas where nonfire decomposition rates are slower. In frequent-fire forests historically, fire-induced mortality affected young conifers to a greater degree than large, older conifers, and frequent fire consumed ground fuels, limiting both the large snag component on the landscape and the CWD component on the ground (Agee 2002). In forests that historically experienced slightly less frequent fire, such as moist mixed-conifer forests, woody debris dynamics were more complex, owing to the larger moderate- and high-severity component of fire in these stands where pulses of CWD were more common (Agee 1993, 2002). In general, Agee (2002) classified low-severity fire regimes as having low, relatively stable levels of coarse woody biomass and fire regimes with a higher proportion of moderate- and high-severity fire as having somewhat higher levels with much more variation over time.

Information on the actual abundance and distribution of CWD historically is very limited. Results from studies assessing CWD in current YPMC reference sites in locations outside of the assessment area, such as the Sierra Nevada, the Sierra de San Pedro Mártir, and the Cascades, indicate a high level of variability (e.g., Dunbar-Irwin and Safford 2016, Lydersen and North 2012, Stephens 2004, Stephens et al. 2007a, Youngblood et al. 2004). Logs ≥50 cm in diameter averaged 10.3 logs/ha in a study located in the Sierra Nevada (Lydersen and North 2012), while logs ≥15 cm in diameter averaged 47 logs/ha with a range of 13.9 to 90.3 logs/ha in sites located in or just east of the Cascades (Youngblood et al. 2004) and averaged 47.8 logs/ha (+/− 8.7 SE) in the Sierra de San Pedro Mártir, with about 50 percent of the plots with no CWD ≥15 cm in diameter (Stephens et al. 2007a). Furthermore, Dunbar-Irwin and Safford (2016) reported a mean of 28.9 logs/ha for CWD >7.6 cm in the Sierra de San Pedro Mártir. Additional discussion of the above studies can be found in Safford and Stevens (2017). Table 8 provides a summary of these studies along with a comparison to modern pristine old-growth sites in the assessment area and current FIA data summaries.

Table 8—Density of coarse woody debris in reference sites outside of the assessment area (AA), in "pristine" old-growth sites within the AA, and current values from the Forest Service's Forest Inventory and Analysis (FIA) summary data from across the California portion of the AA

Forest type	Location	Diameter size class *cm*	Mean density *Logs/ha*	Study
Reference sites outside assessment area:				
Mixed conifer	Sierra Nevada Range	≥50	10.3	Lydersen and North 2012
Yellow pine-mixed conifer	Sierra San Pedro Mártir	>7	28.9	Dunbar-Irwin and Safford 2016
Jeffrey pine-mixed conifer	Sierra San Pedro Mártir	≥15	47.8	Stephens et al. 2007a
Ponderosa pine	Cascade Range	≥15	47	Youngblood et al. 2004
"Pristine" old-growth[a] within assessment area:				
Mixed conifer	California portion of assessment area	≥25; ≥50	39.5; 17.3	Beardsley and Warbington 1996
Jeffrey pine (ultramafic)	California portion of assessment area	≥25; ≥50	24.7; 9.9	Beardsley and Warbington 1996
White fir	Mendocino, Shasta-Trinity, and eastern Klamath NFs	≥25; ≥50	49.4; 19.8	Beardsley and Warbington 1996
White fir	Six Rivers and western Klamath NFs	≥25; ≥50	66.7; 24.7	Beardsley and Warbington 1996
Current condition based on FIA data:				
Mixed conifer	California portion of assessment area	≥25	39.8	USDA FS 2017a
Yellow pine	California portion of assessment area	≥25	19.2	USDA FS 2017a

[a]Stands that met certain old-growth criteria according to Beardsley and Warbington (1996) and did not experience any human disturbance other than the indirect effects of fire suppression.

Beardsley and Warbington (1996) reported CWD densities for several forest types in a study looking at pristine old-growth forests in the assessment area that had not experienced logging, but were likely influenced by fire suppression. In white fir stands located in the Shasta-Trinity and Mendocino National Forests, and the eastern Klamath National Forest, the authors reported about 20 logs/ha for logs ≥50 cm. This is slightly less than the white fir type located in the Six Rivers National Forest and western Klamath National Forest, which was reported to have about 25 logs/ha for the same size class. Mixed-conifer forests were reported to have slightly less than the drier white fir forests at about 17 logs/ha for logs ≥50 cm. The majority of CWD for both the white fir types and the mixed-conifer type was composed of Douglas-fir and white fir. Lastly, Jeffrey pine forests had the lowest CWD density with 10 logs/ha, three-quarters of which were Jeffrey pine logs. Small-diameter logs (about 25–46 cm) ranged from 15 logs/ha in Jeffrey pine stands to 42 logs/ha in moist, white fir stands. The current average density for logs ≥25 cm in diameter according to FIA data is 19.2 logs/ha for yellow pine and 39.8 logs/ha for mixed conifer (USDA FS 2017a), which is comparable to the values reported for old-growth stands, but possibly less than would be expected historically.

Although they provide a starting point for understanding presettlement conditions, most of the studies mentioned above and in table 8 were located outside of the assessment area, and those within the assessment have experienced a century or more of fire suppression. Based on what we do know, it is safe to say that there was substantial variation in the biomass and density of logs on the landscape and that drier areas with a more frequent-fire regime probably had less CWD than wetter areas that naturally experienced longer fire-free periods. It is suggested that forests that experienced a combination of low-, moderate-, and high-severity fire, such as Douglas-fir forests and moist mixed-conifer forests in the assessment area, would have seen much more variation in CWD densities and pockets of substantially higher CWD biomass throughout the landscape (Agee 2002). In a paper assessing how fire affects dead woody material throughout the assessment areas, Skinner (2002) suggested that the variability in fire regime characteristics driven by topography across the landscape contributed considerably to the variability in the spatial distribution of dead woody material. The frequency of mostly low- to moderate-severity fires in the assessment area likely made it rare for dead woody material to decay fully without being consumed by fire, reducing its residence time on the landscape (Knapp et al. 2005, Skinner 2002, Uzoh and Skinner 2009).

The variability in fire regime characteristics driven by topography across the landscape contributed considerably to the variability in the spatial distribution of dead woody material.

Future—Future trends toward more CWD seems likely, especially if nonfire forest mortality agents increase substantially. Increases in fire frequency will increase snags and CWD initially, but will reduce them as areas are reburned.

Fine fuel, litter, and duff—

NRV and comparison to current—Fine fuel is dead plant material, including grass, herbs, leaves, bark, needles, and small twigs, that is less than ¼ inch in diameter. Litter is any dead plant material that is small, such as leaves, bark, and needles, but does not have a specified size class. Combined, fine fuels and litter are considered surface fuels, while duff, defined as litter that is in various stages of decomposition, is considered a ground fuel. Few historical documents make reference to surface and ground fuels in the assessment area, but those that do often remark on the role they played in fire behavior within a given stand or forest type. Leiberg (1900), for example, described the cover of "humus consisting entirely of decaying pine needles" in yellow pine forests west of the Cascades to be more conspicuous than on the drier east side, but remarked that "it is never more than a fraction of an inch in thickness, just enough to supply the requisite material for the spread of forest fires." Munger (1917) writes, "Light, slowly spreading fires that form a blaze not more than 2 or 3 feet high and that burn chiefly the dry grass, needles, and underbrush start freely in yellow-pine forests, because for several months each summer the surface litter is dry enough to burn readily." In areas that tend to have more productive vegetation in the understory, such as in moist mixed-conifer forests or in drainages and on north-facing slopes, litter may have been more abundant. Unfortunately, we have no way of actually quantifying the natural range of variation of litter and fine fuels for the assessment area, especially due to the direct impact excluding fire has had on litter and fine fuel accumulation.

Although intensive grazing practices early on may have caused a reduction in herbaceous cover in open, low-elevation YPMC forests, potentially reducing litter and fine fuels on the landscape in the early part of the 20th century (Borman 2005, Fry and Stephens 2006, Pinchot 1905, Riegel et al. 2018, Skinner et al. 2009), the removal of an active fire regime has allowed for litter and fine fuels to accumulate on the forest floor. Modern fuelbed measurements from FIA plots give an average depth of about 3.7 cm for yellow pine forests and 5.8 cm for mixed-conifer forests for the combined litter and duff layers (USDA FS 2017a). Although we do not have reference sites within the assessment area, according to Safford and Stevens (2017), modern average combined litter and duff measurements in contemporary reference sites (undegraded sites that were not logged and have experienced some 20th and 21st century fire) for YPMC forests range from about 1.5 to 2.1 cm in depth.

Assuming that historical conditions were similar in the assessment area to these contemporary reference sites due to similar species compositions and fire regimes, current litter and fine fuel levels are comparatively higher today than they were historically. In a study conducted in the Klamath Mountains, DiMario et al. (2018) found that litter and duff loading was positively correlated with sugar pine tree size, possibly indicating that greater deposition rates of large trees may be creating an even larger deviation from NRV for litter and duff layers in stands with large trees. This could also suggest that the widespread presence of large trees historically may have caused a more rapid accumulation of needles, yet, with lower tree densities in general and frequent consumption by fire, it was likely that litter and duff loading were still much lower than they are presently.

Future—Unless an active fire regime is returned to forests adapted to frequent fire, surface and ground fuels will continue to accumulate in assessment area forests.

Forest Understory and Nonforest Vegetation

Tree seedlings and saplings—

NRV—Early records of tree regeneration dynamics in YPMC forests often describe it as patchy and somewhat limited, especially for pine species (Cooper and Kelleter 1907, Leiberg 1900, Munger 1917). Leiberg (1900) wrote that the yellow pine type was very open with little growth in the understory and few seedlings and saplings. He stated that the lack of regeneration was due to fire and that "freedom from fires insures a good and abundant reproduction of the forest type, whether east or west of the [Cascade] range." Historically, spatially complex fires created a fine-grained forest matrix with variably sized forest openings that strongly influenced regeneration patterns (Taylor and Skinner 1998). As discussed in the section on forest structure, regeneration in YPMC forest often occurred in newly created gaps or canopy openings, thus forming small patches or clumps of young conifers. The size of these regeneration clumps was highly dependent upon the severity and spatial characteristics of the disturbance that facilitated their establishment.

Munger (1917), in his report on western yellow pine in Oregon, referred to field studies that indicated yellow pine seedlings were most abundant in forest openings with bare mineral soil: "…yellow-pine reproduction is extremely patchy in the virgin forest; here there will be almost a thicket of young trees, and nearby, under seemingly similar conditions, there will be little or no reproduction (Munger 1917: 8). With regard to the effects of fire on tree regeneration, Munger (1917: 11) noted the following:

> …[F]requent fires prevent the stand from having the proper number of young trees. If this process is continued long enough, it will annihilate the yellow pine by gradually killing off the old trees and at the same time preventing the survival and maturity of any young ones. This very thing has happened in places in the Siskiyou Mountains and southern Cascades. Here areas once covered by fine stands of yellow-pine timber are now tree-less wastes, covered only by brush and mock chaparral.

It is hard to determine whether these "tree-less wastes" described by Munger (1917) were a result of fires caused by early Euro-American settlers or not. Other early accounts remark on the shift in fire frequency and size associated with the onset of the settlement period (Leiberg 1900: 277):

> The aspect of the forest, its composition…indicate without any doubt the prevalence of widespread fires throughout this region long before the

> coming of the white man. But, on the other hand, the great diversity in the age of such stands as show the clear origin as reforestations after fires, proves that the fires during the Indian occupancy were not of such frequent occurrence nor of such magnitude as they have been since the advent of the white man... The fires were more numerous and devastated much larger areas in the early days of the settlements than they have done in later years. Much the larger percentage of what may be classed as modern burns date back twenty-five to forty years. 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…

Leiberg (1900: 251) also described the spatial dynamics of sugar pine and incense cedar and their ability to regenerate in YPMC forests west of the Cascades in southwestern Oregon:

> Of the other elements which constitute the yellow-pine type the most prominent are the sugar pine and incense cedar. They rarely form any considerable groups or aggregations together or singly, being found mostly as scattered trees among the other species. The reproductive capacities of the two species appear to be much inferior to those of the other conifers that make up the yellow-pine forest type, which partly accounts for their relative scarcity....

The lack of clustering and the "inferior" reproductive capacities of these two species compared to yellow pine is likely due to their relatively lower fire tolerance as young trees.

In the following passage, Cooper and Kelleter (1907: 6) described regeneration patterns of a YPMC forest located in Siskiyou County in 1904:

> A heavy seeding about fifty-five years ago brought a large number of trees into the forest, which are at present from 12 to 18 inches in diameter. Besides the young trees scattered among the old ones, several openings were seeded up to dense stands.…Another series of heavy seedlings began from ten to fifteen years ago, and resulted in a large number of dense thickets in the openings of the forest. Some of these are of considerable size, one covering more than a section.

Although this account is from 1904, it is stated at the beginning of this same passage that fire had done "little damage" to the virgin timber in the stand for 60 years prior to the study, likely as a result of the removal of American Indian burning and prevention of any other fire in the area when possible. Although it is hard to say for certain, this early fire suppression was potentially already biasing the abundance of regeneration observed, especially because there was an apparent flush of regeneration 55 years prior that still remained within the stand and was presumably untouched by fire.

Show and Kotok (1924: 24) made similar observations:

> Within the past 15 or 20 years, or since fire protection has been attempted, conditions in the virgin forest have changed radically from the time when fires ran unchecked. The most outstanding of these changes is the enormous number of young forest trees that have appeared as individuals and in groups, or, in the more open virgin stand, as a veritable blanket under the mature timber.…The general occurrence of young growth or advance reproduction in the virgin forest today is the effort of nature, in response to fire protection, to utilize the full growing power of the land, and to restore the broken and understocked forest to a more normal condition.

Comparison to current—In current old-growth stands located in the assessment area, the lack of fire over the past century has led to much higher tree densities, especially for shade-tolerant species. In one study, the most common seedlings and saplings found were white fir, despite white fir making up an average of only 14 percent of the large (≥76 cm dbh) overstory trees (Beardsley and Warbington 1996). Leonzo and Keyes (2010) found a relatively even-aged cohort of young trees that emerged 60 years prior to sampling in their study of fire-excluded relict ponderosa pine forests. In addition to the substantial increase in tree density, the authors also observed a shift in composition. While 76 percent of the relict component was made up of fire-tolerant pines, pines accounted for only 17 percent of the younger trees, while white fir accounted for about one-third. In addition to the more closed-canopy conditions favoring shade-tolerant tree species, this shift in composition may also be attributed to the lack of bare mineral soil and the reduced survival and establishment of shade-intolerant species in areas with thick litter and duff layers.

While we may be seeing an overabundance of conifer regeneration within long-unburned forests, we are also seeing a lack of natural regeneration following severe burning in recent wildfires, especially in more arid locations.

At the same time, as we continue to see larger contiguous areas on the landscape burn (and reburn) at high severity, the forest's ability to recover after severe disturbance has been reduced. Large, high-severity patches lead to a reduction in seed source availability and create harsh conditions that reduce the chances of seedling survival and can drive changes in future forest composition and overall sustainability (Collins et al. 2017, Donato et al. 2009, Shive et al. 2018, Tepley et al. 2017, Welch et al. 2016). Therefore, while we may be seeing an overabundance of conifer regeneration within long-unburned forests, we are also seeing a lack of natural regeneration following severe burning in recent wildfires, especially in more arid locations (Tepley et al. 2017). Furthermore, the potential for the regeneration that does occur to reach maturity is strongly diminished because of the likelihood of reburns as the climate continues to warm.

Modern seedling densities based on FIA plot data average about 642 seedlings/ha (± 1,420 std dev) for yellow pine forests and 1,215 seedlings/ha (± 2,222 std dev) for mixed-conifer forests (USDA FS 2017a). The high standard deviations (coefficients of variation are 2.2 for yellow pine and 1.8 for mixed conifer) indicate that there is a high level of variation in seedling densities among plots. Although we lack quantitative information on presettlement tree regeneration, combining early descriptions of regeneration patterns with what we know about spatial characteristics of presettlement forest stands and present day regeneration patterns allows us to conclude that modern seedling and sapling densities are higher than average historical densities. Additionally, we can conclude that—in an overall sense—regeneration responds differently to modern fires that contain high proportions of high-severity fire in large patches than it did to the mostly low- to moderate-severity fires that burned in presettlement times.

Future—If current trends continue, seedling and sapling densities in unburned forest will continue to increase and shade-tolerant species will become increasingly dominant. The reintroduction of fire (possibly in combination with thinning where young trees are too large to be removed by prescribed fire alone) can help reduce regeneration densities and may help shift regeneration compositions back toward more shade-intolerant species. Supplemental planting may be necessary to shift forest composition in many places where tree canopy dominance has shifted. In the wake of large fires that have large stand-replacing patches, conifers struggle to reestablish because of a combination of a lack of seed source and highly competitive, fire-stimulated shrub species (Tepley et al. 2017) accompanied by an increased likelihood of reburning. In these situations, it may be necessary to assist in the recovery of these forests through planting and fuels management (Coppoletta et al. 2016, Welch et al. 2016).

Shrubs—

NRV—Shrubs were noted by a number of early travelers in the assessment area, usually with regard to their impenetrability. It is clear from early accounts (e.g., Cooper and Kelleter 1907, Leiberg 1900, Munger 1917) that shrub cover was very heterogeneous across the landscape. Munger (1917) describes pure yellow pine forests throughout Oregon as "fairly free from underbrush and debris" and easy to travel through on foot or horseback. He goes on to note that, "On the north slopes, in draws, or in places where mixed with other species, the yellow-pine forests are usually denser, more brushy, and therefore harder to traverse." Munger also writes that in the southern Cascades and Siskiyou Mountains, where mixed conifer becomes more prevalent and the forests are more productive, shrubs become much more abundant and "…the more open the woods the greater the amount of brush." General Land Office surveys conducted in the northern portion of the assessment area also mentioned "dense undergrowth" in the understory of some of the mixed-conifer stands (Hickman and Christy 2011).

Frequent fire in YPMC forests played a key role in influencing shrub cover in the understory. This took the form of stimulating shrub growth by creating openings in the canopy and scarifying the seeds of fire-stimulated species, as well as limiting shrub growth by exhausting the ability of shrubs to recover through frequent burning. Cooper and Kelleter (1907) state that fire's "chief effect upon the forest is the destruction of brush and litter" in the understory. They go on to discuss openings created by fire and the persistence of shrubs in these areas, particularly when fire returns: "Openings thus made in the forest are effectually prevented by subsequent fires from coming up to young growth, while chaparral, which sprouts from the roots and is not permanently eliminated, even though completely burned back, takes possession of the ground."

Leiberg (1900) noted that it was common for the growth of shrub and hardwood species, such as ceanothus, manzanita, and madrone, to increase after fire. He described extensive brushfields resulting from fire on west- and south-facing slopes previously covered by Douglas-fir stands, and he wrote that fires in some areas "…instead of being followed by reforestations, give rise to enormously dense brush growths. Tracts of this sort are found in scattered patches everywhere along the middle elevations on the western side of the Cascades, and throughout the Siskiyou range…" In a summary of General Land Office surveys for southwest Oregon, Hickman and Christy (2011) noted that shrubs were no doubt an important component on the landscape, but the extensive shrubfields described in the early 20th century were most likely the result of early Euro-American settlers as they were not documented in the mid-19th century General Land Office surveys.

Aside from fire influencing the presence or absence of shrubs, shrub cover patterns are largely driven by interactions between moisture availability and substrate.

Aside from fire influencing the presence or absence of shrubs, shrub cover patterns are largely driven by interactions between moisture availability and substrate. According to Whittaker (1960), in low-elevation gabbro and diorite sites, average shrub cover increased from 10–12 percent in moister sites to 20–32 percent in xeric sites. Whittaker (1960) stated that "It is consequently possible to stand on one hillside in the gabbro area and look through the canopy to the soil on another, nearby hillside…" On serpentine soils, shrub cover followed a unimodal trend with lower cover (20–50 percent) in mesic sites, high cover (50–90 percent) in semi-mesic sites, dropping back down (20–50 percent) in semi-xeric sites, and finally to its lowest level (0–20 percent) in xeric sites (Whittaker 1960).

Comparison to current—Currently, there are a couple of different trends in shrub cover that are apparent in YPMC forests. The long-term lack of fire in these forests has led to the infilling by trees of forest openings, leading to a decrease in the area of smaller stands of chaparral that often grow in forest gaps (Duren et al. 2012, Knapp et al. 2013, Nagel and Taylor 2005, Skinner 1995). Knapp et al. (2013), for example, found an 11-fold decline in shrub cover between 1929 and 2008 related to the increase in forest density and infilling of small gaps where shrubs were most commonly found. Similarly, in a 420-year chronosequence study conducted in the Siskiyou Mountains, shrub cover was found to be highest in the youngest stands and then declined as forest stands moved into the canopy closure phase, and remained low in the oldest forest stands. The occurrence of large, high-severity fires, on the other hand, is leading to large areas of forest being converted to shrublands because of the lack of nearby seed sources and the inability of the forest to reestablish successfully, especially in more xeric areas (Tepley et al. 2017, Welch et al. 2016). In both cases, fine-scale heterogeneity across the landscape is being replaced by coarse-grained heterogeneity (Skinner 1995).

According to a summary of modern studies by Safford and Stevens (2017) in contemporary reference sites in the southern Cascades and Sierra Nevada, average relative shrub cover ranges from about 15 to 25 percent, but is highly variable. These numbers seem broadly appropriate for the assessment area, with the caveat that moister sites (which are more common in the assessment area than in the Safford and Stevens study region) tend to support higher shrub cover after disturbance, but more rapid regrowth of tree canopy in such sites can rapidly attenuate shrub cover as well. From the FIA data, modern shrub cover in the assessment area averages 19.2 percent (± 17.4 percent std dev) for yellow pine forests and 17.7 percent (± 20.4 percent std dev) for mixed conifer, with 12.6 percent shrub cover in dry mixed-conifer forests and 20.4 percent shrub cover in moist mixed-conifer forests. Based on what we know, the overall cover of shrubs on the landscapes is probably broadly similar to historical conditions, but the spatial distribution of shrub cover is likely different. Specifically, there is likely lower shrub cover within intact forest—due to increasing tree density and tree cover under fire suppression—while the extent of large shrubland patches resulting from high-severity fire is increasing, much like it did in the early settlement years (see Mallek et al. 2013, Safford and Stevens 2017, Tepley et al. 2017).

Future—Under current trends, we will continue to see increasing forest density and reduction in the area occupied by forest gaps and forest-floor shrubs. At the same time, we will see large increases in the area of contiguous shrub patches that occupy large, high-severity patches in wildfires (Tepley et al. 2017).

Grasses and forbs—

NRV and comparison to current—Information on historical herbaceous cover in the understory of YPMC forests is extremely limited. Most early accounts focus on trees and shrubs, usually only mentioning herbaceous plants with regard to forage for their horses or livestock (Leiberg 1900, Munger 1917). In general, herbaceous cover, especially grass cover, was described as patchy in YPMC forests prior to Euro-American colonial settlement (Hasel 1932, Leiberg 1900, Munger 1917), although given what we know about how grasses and forbs respond to fire, herbaceous cover was likely higher than described in these early accounts. Historically, forage was promoted in forest understories through the use of fire by American Indians (and later by early settlers) (Anderson 2005, LaLande 1980). Based on our knowledge of trends in mixed-conifer forests of the Sierra Nevada and southern Cascades (e.g., Laudenslayer and Darr 1990, McKelvey and Johnston 1992, Safford and Stevens 2017), it is likely that a combination of logging, fire suppression, and heavy grazing in the assessment area during the first half of the 20th century reduced herbaceous cover and allowed for woody plants (i.e., shrubs and trees) to expand.

On upper slopes and summits in the Cascades and Siskiyous, Leiberg (1900) noted that it was common for fires to promote grass-covered tracts where forest had previously grown. In areas of higher moisture, the cover was sometimes described as being more or less continuous, but more often than not it was scattered in limited patches. Leiberg (1900) also noted that "…if suffered to remain undisturbed by further fires they will return to forest cover." Most of the areas that Leiberg described were at higher elevations where lodgepole pine is more abundant, and these patterns may not necessarily apply to mid- and lower-montane areas (Leiberg 1900).

Munger (1917: 31) stated the following in his description of western yellow pine in Oregon:

> In the yellow-pine forests of Oregon (except those on both slopes of the Cascades south of Crater Lake and those on the Siskiyou Mountains in southern Oregon and on some of the pumice-stone land toward the head of the Deschutes River) the trees are so open-grown and the woods are so free of underbrush that a good herbaceous vegetation suitable for forage springs up each year. The character of the vegetation depends upon the region, but it usually consists in part of a variety of grasses and in part of 'weeds' (annual flowering plants).…Nearly all yellow-pine land in the State which is not too brushy or too sandy is grazed by one or the other of these classes of stock [cattle and sheep].

Munger (1917) made a point to note that yellow pine forests in the areas of Oregon included in the assessment area were not open enough to support herbaceous cover suitable for grazing every year. However, his statement that yellow pine land that is "not too brushy or sandy" is able to support grazing of cattle and sheep, permits the inference that areas in the assessment area that had openings maintained by fire or edaphic conditions likely supported higher herbaceous cover. But even in more open forests, herbaceous cover may still have been rather low. Figures 8 and 9 show some of the variability in understory abundance in the assessment area across a moisture gradient, with the drier, open yellow pine forest exhibiting patchy grass and forb cover (fig. 8), and the moister, mixed-conifer forest exhibiting a more continuous herbaceous layer in openings (which are rare in undisturbed forest) (fig. 9).

In a study assessing the effects of fire suppression on a Jeffrey pine savanna on serpentine soil, Sahara et al. (2015) found evidence of grass-dominated areas declining substantially since the 1940s (from 75 ha in 1942 to 21 ha in 2009; last fire was recorded in 1940). Similarly, Damschen et al. (2010) found considerable declines in herbaceous cover in the Siskiyou Mountains between 1950 and 2007 on both serpentine and diorite soils, with the largest differences seen in forb cover. The authors of this latter study acknowledge the possible effect that fire suppression may have had on herbaceous cover but mostly attribute this decline to a changing climate because the strongest reductions in cover were associated with species with a northern biogeographic affinity.

In a study conducted in a Douglas-fir-dominated forest in the Siskiyou Mountains, Jules et al. (2008) compared cover and richness of understory plants species between stands ranging from 7 to 427 years old (a 420-year chronosequence). The authors found graminoid cover to be negatively correlated with stand age, while herb cover, in contrast, was positively correlated with stand age, reaching its peak in the oldest stands. In a study looking at the impact of reforestation (planting plus removal of competing vegetation) and time since high-severity fire on plant species richness and cover in Sierra Nevada mixed-conifer forests, Bohlman et al. (2016) found that average herbaceous cover was nearly 70 percent in the youngest sites (10 years postfire) but dropped substantially to 2.5 percent in the oldest sites (41 years postfire). This drop in herbaceous cover occurred while average conifer cover increased from 4 percent in the youngest sites to more than 70 percent in the oldest sites. This same study showed that paired untreated sites (sites that had no reforestation or control of shrubs) had significantly lower herbaceous cover at 10 and 22 years postfire (27.5 and 3.1 percent, respectively) but not at 41 years postfire (likely due to the high levels of conifer or shrub cover in these older stands [Bohlman et al. 2016]).

Based on FIA plot summaries, modern forb and grass cover averages about 4.7 percent (± 7.7 percent std dev) for yellow pine forests and 3.8 percent (± 8.1 percent std dev) for mixed-conifer forests (USDA FS 2017a). It is difficult to assess whether these current values are within NRV because we lack sufficient presettlement data, but it is likely that, similar to shrub cover, herbaceous cover is lower within forested stands than it was historically but possibly higher in severely or repeatedly disturbed areas, except where cover of fire-stimulated shrub remains high (Bohlman et al. 2016).

Modern herbaceous cover in undisturbed mixed-conifer forests is likely lower today than during presettlement times due to much higher tree densities.

Based on a systematic review conducted by Abella and Springer (2015) on the effects of tree cutting and fire on understory vegetation in western, mixed-conifer forests (excluding yellow pine forests), it is clear that both disturbances, alone or in combination, usually increase the abundance of grasses and forbs, especially over the long term (>4 years after treatment). All of the longest term studies included in the review that exhibited increases in herbaceous plant cover or richness also all experienced a substantial and persistent reduction in tree canopy cover. This positive response to a sustained reduction in overstory tree density further supports the idea that modern herbaceous cover in undisturbed mixed-conifer forests is likely lower today than during presettlement times due to much higher tree densities.

Future—It is likely that within forested areas there will be a continued reduction of herbaceous cover due to fire suppression. Yet as fire activity increases, grasses and forbs may be promoted, especially in more arid areas where woody species struggle to reestablish.

Composition

Forest Landscape Composition

NRV and comparison to current—

YPMC forests throughout the assessment area vary a great deal in their composition and stand structure across the landscape. Multiple early accounts described forest patterns on the landscape, frequently attributing them to observed climatic and edaphic conditions. From these accounts it is clear that open, yellow pine and dry mixed-conifer forests were most common at lower elevations, on south- and west-facing slopes, and on less productive soils. Similar to today, moist mixed-conifer forests were more abundant at higher elevations, on north- and east-facing slopes, and in sites with higher soil moisture, such as in drainages. Further to the north and to the west in the assessment area, Douglas-fir becomes the dominant conifer species; however, mixed-conifer forests are still present on warmer slopes and on thin or nutrient-poor soils, such as serpentine.

According to observations made by Merriam (1899) in the central eastern portion of the assessment area, yellow pine formed a continuous open forest on both the south and west sides of Mount Shasta reaching up to about 5,500 ft (1680 m) in elevation. Merriam described ponderosa pine as very widespread, spanning across adjacent foothills and covering the lower portions of the Scott Mountains to the west, reaching across Scott Valley into the Salmon Mountains where it mixed with Douglas-fir, incense cedar, white fir, and sugar pine to form mixed-conifer forests.

Show and Kotok (1929) provided estimates of acreage for each major vegetation type across three of the national forests (Klamath, Shasta-Trinity, and Mendocino) in the assessment area. In the 1920s, 24.8 percent of the landscape on these three forests was estimated to be yellow pine and 22.1 percent was estimated to be mixed conifer. Based on current estimates of vegetation cover using the presettlement fire regime types developed by Van de Water and Safford (2011), yellow pine forests in the same area cover 11.7 percent, which is less than half the area estimated by Show and Kotok (1929). Mixed conifer on the other hand, covers approximately 42 percent of this landscape, almost double the estimate of Show and Kotok from the early 1900s. This current estimate includes both dry and moist mixed conifer, which, on their own, cover 19 and 23 percent, respectively.

According to Hickman and Christy (2011), who summarized and mapped General Land Office survey data from the mid-19th century for an area in southwestern Oregon, areas with less than 100 cm of annual precipitation were largely dominated by mixed-conifer/hardwood forests. These forests were reported to include Douglas-fir, yellow pine, sugar pine, cedar, madrone, and white and black oak, with the relative dominance of Douglas-fir being dependent on environmental variables (i.e., elevation, aspect, soil) as well as disturbance history. Duren et al. (2012) compared the same 1850s General Land Office vegetation reconstruction maps with present-day aerial orthoimages in southwest Oregon to determine trends in vegetation distribution over time and the factors contributing to these observed trends. The authors found an increase in conifer-dominated sites (both yellow pine-dominated and Douglas-fir-dominated sites with ≥25 percent conifer cover and <10 percent hardwood cover) and a decrease in mixed-conifer/hardwood sites (both conifer and hardwood cover ≥10 percent in more or less equal ratios), despite a substantial conversion of savanna sites to mixed-conifer/hardwood. The decrease in mixed-conifer/hardwood sites was, in part, attributed to the encroachment of conifers shifting the composition to favor conifers as a result of fire suppression, yet some of these areas also transitioned to hardwoods, losing their conifer component (Duren et al. 2012).

In a comparison between historical (1930s) Forest Service plots (from the Vegetation Type Mapping [VTM] project) and contemporary (2000s) FIA plots, McIntyre et al. (2015) found an overall increase in the oak-to-pine ratio throughout California. Overall, the authors attributed this trend to increased temperatures and water stress, but timber harvest, land use, and successional changes were also locally important (McIntyre et al. 2015). While pines showed a significant decrease in the assessment area, the reported increase in oaks was not statistically significant, which may be due to less extreme shifts in climatic water deficit compared to other parts of the state, or simply the relatively low number of VTM plots in the assessment area (see fig. 3 in McIntyre et al. 2015).

Future—
While conifers will continue to invade areas previously dominated by hardwoods and shrubs in the absence of fire, projections of increased fire activity (especially increased risk of high-severity fire) and increasing temperatures will likely accelerate the already ongoing shift toward more hardwood and shrub domination in disturbed areas, as a result of their ability to resprout after disturbance, as well as the relatively high tolerance for drought conditions of some of the key hardwood taxa (especially oaks).

Forest Composition and Species Diversity

Overstory—
In this section we focus on quantitative and qualitative descriptions of forest composition prior to Euro-American colonial settlement and discuss current departures along with projections for future trends. Refer to the "Ecological Setting" section in the introduction as well as the many references cited in that section for a more indepth description of current forest composition patterns.

NRV and comparison to current—Both historically and presently, the presence and dominance of different tree species in YPMC forests in the assessment area is strongly driven by climatic and edaphic conditions as well as the frequency and intensity of disturbance. The assessment area is also known for being a refuge for plant populations that were otherwise extirpated by "glaciation, submergence of coastal plains, climatic desiccation, and in the West, the great lava flows of the Interior" (Whittaker 1960). Its unique history and diverse climatic conditions across the landscape have made the Klamath region a hotspot for conifer diversity, which can be seen most prominently in the famous "enriched mixed conifer" forests (Sawyer et al. 2009). Early descriptions, however, focused largely on the more dominant species that were present across most of the landscape.

Information provided by Leiberg (1900) for a handful of individual stands located in the yellow pine type permits a summary of the range of percentages for each of the key species as follows: 60–70 percent yellow pine; 5–35 percent Douglas-fir; 0–8 percent sugar pine; and 0–25 percent for oak, madrone, and incense cedar. Douglas-fir and white fir were said to form higher percentages in moister areas among the predominantly yellow pine forests, limiting yellow pine to only about 5–10 percent of the stand at times and in some cases eliminating yellow pine altogether. Leiberg noted that yellow pine was capable of maintaining its dominance over Douglas-fir in these sorts of areas where fires had been frequent. At the lower elevational range of yellow pine, oaks were dominant, forming about 40–60 percent of the stand with the balance made up of pines, Douglas-fir, white fir, madrone, and other broadleaf species.

Leiberg (1900) emphasized the relative importance of broad-leaved trees on the west side of the Cascades as compared to the east, forming about 6 percent of the forest on the west side, with oaks, madrone, and chinquapin being the dominant taxa. While yellow pine and Douglas-fir were identified by Leiberg as the predominant conifers west of the Cascades in his study region, he also noted a high level of variation in the ratios of different tree species. Leiberg (1900: 237) gave the following relative percentages for conifer species that made up the forest on the western slope of the Cascades in southern Oregon: 44 percent Douglas-fir, 27.5 percent yellow pine, 6.5 percent mountain hemlock, 6.3 percent lodgepole, 5.8 percent noble fir, 5.4 percent white fir, 2.8 percent sugar pine, 1 percent western hemlock, 0.6 percent Engelmann spruce, 0.41 percent incense cedar, 0.36 percent white pine, 0.2 percent alpine fir, and 0.03 percent whitebark pine.

Although climatic conditions are a strong driver of forest composition, there are many cases in the assessment area in which edaphic conditions are the driving force.

Although climatic conditions are a strong driver of forest composition, there are many cases in the assessment area in which edaphic conditions are the driving force. A study done in 1949–1951 by Whittaker (1960) in the central Siskiyou mountains assessed tree distributions across moisture gradients (mesic sites in ravines and on north-facing slopes, to xeric south/southwest-facing slopes), elevation gradients (460 to 2140 m), and on three different parent materials (quartz diorite, olivine gabbro, and ultramafic/serpentine rock). Whittaker found that, in general, edaphic conditions were much stronger drivers of species compositions and densities than climate in his study area. Pine dominance was higher on gabbro sites than diorite sites at low elevations (610 to 915 m), with Douglas-fir dominating on diorite. In montane sites, mixed-conifer stands of white fir, Douglas-fir, western white pine, Jeffrey pine, and cedar were all present with a total conifer cover ranging from 40 to 70 percent (Whittaker 1960). Low-elevation, serpentine sites in Whittaker's study were dominated by sparse pines and lacked the tree forms of almost all deciduous broad-leaved and sclerophyllous species. Many species present on nonserpentine also appeared on serpentine, but as dwarfed individuals. Semixeric to xeric serpentine sites hosted most of the same conifers present in montane sites, but with much lower tree cover (typically less than 50 percent) dominated primarily by Jeffrey pine.

In dry sites located on the eastern Klamath National Forest, Hasel (1932) described the composition of two YPMC stands prior to logging in 1910. The plots had very similar species compositions and averaged 61.2 percent ponderosa pine, 18.5 percent incense cedar, 14.1 percent Douglas-fir, 5.8 percent white fir, and 0.5 percent sugar pine, with black and white oaks mentioned as part of a sparse "brush layer." Hasel noted that between 1911 (the year after selective cutting) and 1930 (final measurements taken), ponderosa pine and incense cedar percentages had dropped, while Douglas-fir and white fir had increased.

In a study conducted by Hagmann et al. (2017), stand exam data from 2014 were compared to a 1914–1924 timber inventory for an area managed as northern spotted owl habitat under the Northwest Forest Plan in the foothills of the Cascade Range. Hagmann et al. (2017) not only found an overall increase in canopy cover and basal area, they also found a shift in forest composition associated with selective logging and fire suppression. Large areas that were previously dominated by open, yellow pine or dry mixed-conifer forests had been historically maintained by frequent fire. Long-term lack of fire and the harvest of the large pines from the system has made these forests more similar in density and composition to a moist mixed-conifer forest, with a substantially greater Douglas-fir and grand/white fir component (Hagmann et al. 2017). Results from the study underline the human-driven shifts in forest composition that have occurred throughout much of the assessment area over the past 150 years.

In their study of relict YPMC forests in Whiskeytown National Recreation Area, Leonzo and Keyes (2010) also found shifts in forest composition associated with fire suppression. The overstory (their "relict" component) was dominated by pines (76 percent of individuals), with 9 percent white fir and 7 percent Douglas-fir. The "encroachment" trees (younger trees that recruited after fire suppression) on the other hand were 17 percent fire-adapted pines and 64 percent fire sensitive species, predominantly white fir. One of the most dramatic shifts in composition reported was at a site that was previously pine dominated, where the remaining pines are present as fire-killed snags and are surrounded by heavily recruiting tanoak and Douglas-fir (Leonzo and Keyes 2010).

Another compositional shift occurring in YPMC forests is in high-severity burn areas where knobcone pine is present. Knobcone pine is a serotinous conifer that is endemic to parts of California and southwestern Oregon and is usually found growing in chaparral (Keeley 1999, Reilly et al. 2019). It can also be found in small stands within YPMC forests and has the ability to regenerate prolifically after high-severity fire, often expanding into adjacent areas (Donato et al. 2009, Reilly et al. 2019). Reilly et al. (2019) found an overall expansion of knobcone pine in response to recent fires throughout its range. One example of the expanding extent of knobcone pine in the assessment area is in the Lamoine area of the Sacramento River watershed. In 1985, the Delta Fire burned through mixed-conifer stands with small pockets of knobcone pine present. The fire was largely high severity in this area, and knobcone pine was able to capture most of the area, even on north slopes where it was able to share dominance with rapidly resprouting black oaks. The area subsequently reburned in another severe fire in 2018 (also called Delta). This second fire released the seeds stored from the young knobcone pines that had established since the 1985 fire and helped expand the area occupied by knobcone pine. Once a site is captured by knobcone pine, it is extremely difficult to convert it back to mixed-conifer forest.

Current relative densities in YPMC forests based on FIA plot summaries in the California portion of the assessment area show some distinct differences between dry and wet sites (USDA FS 2017a). In dry sites, Douglas-fir and ponderosa pine have the highest relative densities (25 and 20.3 percent, respectively) followed by canyon live oak (15.2 percent), white fir (11.4 percent), black oak (6.4 percent), and incense cedar (6 percent). The remaining 15.7 percent is composed of sugar pine, white oak, Jeffrey pine, madrone, and a number of other tree species, each with less than 4 percent average relative density. In wet sites, white fir and ponderosa pine have the highest relative densities (27.6 and 15.6 percent, respectively) followed by Douglas-fir (15.6 percent), incense cedar (11.9 percent), canyon live oak (4.3 percent), and black oak (3.4 percent). The remaining 21.7 percent is composed of tanoak, Jeffrey pine, lodgepole pine, sugar pine, red fir, and a number of other tree species, each with less than 3 percent average relative density.

Future—The majority of understory trees in modern YPMC forests are shade-tolerant and relatively less fire-tolerant species. These species outcompete pines in shady, undisturbed conditions. Under current management and climate trends, it seems clear that future forests will support higher and higher densities of such species at least until they burn, further decreasing populations of fire- and drought-tolerant species and decreasing ecosystem resilience to future fire and drought. In areas that burn at high severity, there is the potential for type conversion to species that are capable of responding vigorously to severe fire (i.e., chaparral, knobcone pine, or resprouting oaks).

Floristic diversity and endemism are extremely high in the assessment area.

Understory—

***NRV and comparison to current*—**Floristic diversity and endemism are extremely high in the assessment area (Whittaker 1960) —and as with overstory species—the presence and abundance of understory species varies along the strong climatic and edaphic gradients in the area. Safford and Miller (2020) demonstrated that levels of plant endemism on serpentine substrates (both serpentinite and peridotite) in the assessment area are notably higher than anywhere else in California; most serpentine endemics in California and the assessment area are herbaceous species.

Merriam (1899) described how climatically driven changes in vegetation near Mount Shasta "occur in such small compass that they may be studied to excellent advantage." Species characteristic of more arid areas, such as *Artemisia tridentata* Nutt. ssp. *arbuscula* (Nutt.) H.M. Hall & Clem., *Purshia tridentata* (Pursh) DC., *Arctostaphylos patula*, *Ericameria nauseosa*, *Rhus trilobata* Nutt. var. *arenaria* (Greene) F.A. Barkley, *Garrya fremontii* Torr., and *Prunus subcordata* Benth., can be found near areas with moister soils that host species such as *Acer glabrum* Torr., *Cornus nuttallii* Audubon ex Torr. & A. Gray, *Rubus parviflorus* Nutt., and *Spiraea douglasii* Hook. (Merriam 1899). Merriam (1899) was one of the few individuals to develop a robust list of understory species prior to the 20th century within the assessment area. Despite the detail with which Merriam (1899) described the relative abundance of each species, his work was qualitative and was not linked to plot measurements, which does not allow for directional comparisons of species abundances or distributions.

Leiberg (1900), who provided detailed descriptions of tree species distributions and abundances throughout the northern portion of the assessment area, largely ignored the understory plant communities, only referring to them briefly with respect to their response to fire-induced openings in the forest. He noted that in low-elevation, dry, yellow pine forests, fires can cause an increased growth of ceanothus and manzanita, and at higher elevations, fire tends to lead to increases in grasses and sedges (Leiberg 1900). Observations made by Leiberg (1900) are in line with current knowledge and observations surrounding the effects of fire and reduced tree densities on the understory plant community in and around the assessment area (e.g., Abella and Springer 2015, Martin and Sapsis 1992, Odion et al. 2010, Whittaker 1960). It can be assumed that, owing to the modern lack of frequent fire throughout most of the assessment area, current understory plant communities are outside their NRV when it comes to composition and relative densities. A study conducted by Stevens et al. (2015) in eastern California found that richness of species from more southerly and more xeric biogeographic areas (i.e., *Arctostaphylos* spp., *Ceanothus* spp., *Epilobium* spp.) increased with fire severity, while richness of species with moister, more northerly centers of distribution (i.e., *Amelanchier* spp., *Symphoricarpos* spp., *Chrysolepis sempervirens*) showed an inverse relationship with fire severity. This effect was greatest in more productive mixed-conifer forests. When considering stand-level plant diversity, peak diversity occurred where fires burned at intermediate severities (low- to moderate-severity fire burning that was promoted by prefire fuels treatments) as taxa from both biogeographic groups could coexist. The results of this study further support the idea that reduced fire frequencies have likely had a large impact on assessment area understory plant communities by shifting their composition more toward plants with northern temperate affinities.

Jules et al. (2008), in a 420-year chronosequence study that compared cover and richness of understory plants species between stands in the Siskiyou Mountains, found that understory richness showed a steady decline as young stands entered the canopy closure stage and then increased in older stands as light increased. Stands around 55 years old were estimated to have the lowest understory richness levels. In a study in which the authors were able to compare modern understory richness and composition with data from 1929, herbaceous species richness in 2008 was not significantly different from 1929 (prelogging), but was significantly lower than in 1931 (2 years postlogging) (Knapp et al. 2013). The authors also noted a shift in composition from species typically found in higher light environments and associated with disturbance to species typical of shaded environments with deep duff layers. Furthermore, in areas that burn at high severity, shrub cover and associated competition for resources can be a major driver in reducing understory species richness. Bohlman et al. (2016) found that postfire sites with high shrub cover had significantly lower species richness than paired sites that were treated to reestablish conifer cover and reduce shrub cover. In both cases, however, understory richness decreased with time postfire as shrub or conifer cover increased (Bohlman et al. 2016).

Overall, the evidence suggests that there have been notable shifts in local composition and dominance patterns among forest understory species. However, while disturbed areas are susceptible to invasion by nonnative species (Dix et al. 2010), and the number of these invaders is constantly increasing (Baldwin et al. 2012), we know of no evidence suggesting that overall native species richness across the assessment area has changed.

We know of no evidence suggesting that overall native species richness across the assessment area has changed.

***Future*—**Though the understories of unburned YPMC forests may continue to provide habitat for shade-tolerant, moisture-loving species, warming temperatures will likely drive such species to cooler slopes, higher elevations, and more northerly latitudes over time (as demonstrated by Damschen et al. 2010). Altered understory microclimates associated with fire or management actions, such as thinning, will likely accelerate plant community shifts toward species from warmer regions (Stevens et al. 2015).

Summary of Probable Deviations From NRV and Conclusion

Based on our understanding of YPMC forest systems in the assessment area and an extensive literature review of the best available science, we have attempted to draw conclusions with regard to whether key ecosystem variables are currently within or outside of the NRV. Table 9 summarizes our conclusions and directs the reader to the areas of this report that discuss the ecosystem elements in question.

The following general conclusions can be made about the current status of YPMC forests in the assessment area:

- Ecosystem structure in YPMC forests is considerably different than the structure that existed prior to Euro-American colonial settlement in a number of ways:
    - Mean tree densities have increased in the assessment area primarily because of fire-suppression policies. Exceptions to this rule may be found in pedologically stressful environments, but taken as a whole, modern forests contain much higher stem densities than they did during the presettlement period.
    - Tree seedling densities are similarly much higher in the modern forest, and they are dominated by shade-tolerant species that are usually less fire tolerant, especially when young, than yellow pines.
    - Average tree size is much smaller in modern forests than in the presettlement forests.
    - The number of small trees has increased tremendously over the past century or more, with many areas showing two to five times higher small-tree densities, and the number of large trees has decreased greatly.
    - The overall distribution of tree size classes vs. density has shifted from a relatively flat curve (relatively small differences in density between tree size classes) to a very steep curve dropping from high numbers of small trees to low numbers of large trees.
    - Modern basal areas appear to be higher than they were in presettlement forests owing to the marked increase in smaller diameter trees, which to this point has more than compensated for declines in basal area as a result of losses in large-diameter trees.
    - Modern canopy cover in assessment area YPMC forests is probably higher on average than under presettlement conditions, but our confidence in this statement is low due to limited quantitative information from early forests.

    - Current forests have fewer gaps and larger tree clumps than they did prior to the fire-suppression era. Fine-grained heterogeneity has decreased as a result of increasing tree densities, and increases in large, high-severity patches of fire on the landscape have further promoted coarse-grained heterogeneity, creating a mosaic of larger, more defined patches across the landscape.
    - Most of the evidence suggests that, on average, snag densities are somewhat higher today than in presettlement forests.
    - Overall fuel loading is higher today than it was during presettlement times. Although coarse woody debris is highly variable across the landscape, evidence suggests that current averages are either somewhat higher than historical averages or are within the NRV. Litter and fine fuel loadings are higher today than they were historically.
    - Total shrub cover in modern forests is most likely similar to presettlement conditions, but the spatial distribution of shrub cover differs. Modern forests are more likely to support large areas of contiguous shrub fields but relatively low shrub cover within forest stands (owing to more closed forests), whereas presettlement forests supported higher cover of shrubs present in a patchy matrix within stands, as light incidence at the soil surface was much higher.
- The increasing density of forest stands and the distribution of dense shrub stands has likely caused dramatic reduction in the understory cover of grasses and forbs.
- With regard to ecosystem composition of YPMC forests, although overall plant species diversity across the assessment area has likely experienced minimal changes (except for the addition of nonnative species), there has been a major shift over the past century from dominance by shade-intolerant/fire-tolerant species to dominance by shade-tolerant/less fire-tolerant species. This has happened in both the forest overstory and understory.
- With regard to ecosystem function, the major change in YPMC forests has been in the role and behavior of fire in the following ways:
    - More than 85 percent of Forest Service lands in the California portion of the assessment area is burning much less frequently now than under presettlement conditions.
    - Fires have shifted from being a frequently recurring ecological process on the landscape (median FRI of <20 years with some exceptions) to being an extremely rare event. Most of the landscape has not seen a fire in the past 100+ years, even with the occurrence of recent large fires.

Table 9—Summary of probable deviations from the natural range of variation (NRV) for yellow pine and mixed-conifer forests in the assessment area

Ecosystem attribute	Indicator group	Indicator	Variable	Within NRV[a]	Confidence	See discussion on page:	Notes
Function	Disturbance	Extreme climatic events	Drought	Yes	Moderate	23	Likely within NRV, but projected future drought conditions may move outside of the NRV.
Function	Disturbance	Extreme climatic events	Extreme precipitation events	Yes	Moderate	26	Likely within NRV, but projected future precipitation patterns may move outside of the NRV.
Function	Disturbance	Fire	Fire regime	No	High	28	There has been an overall shift from Fire Regime I to Fire Regimes III and IV. Multiple components of the current fire regime are outside of NRV (see below).
Function	Disturbance	Fire	Fire frequency	No	High	31	Fire frequency is currently lower than NRV; some exceptions include higher elevation white fir forests and Jeffrey pine forests on ultramafic sites.
Function	Disturbance	Fire	Fire rotation	No	High	35	Fire rotation on the landscape is currently longer than NRV.
Function	Disturbance	Fire	Fire severity	No (yes)	Moderate	37	Area of fires burning at high severity is higher than NRV, as is proportion of area burning at high severity; annual exceptions to this can occur when lightning bursts cause large areas to burn during periods of moderate fire weather.
Function	Disturbance	Fire	Fire size	No	High	45	There is a deficit of area burned by small fires and an excess of area burned by large fires.
Function	Disturbance	Fire	High-severity patch size	No	High	44	There are larger contiguous areas burning at high severity in current fires.
Function	Disturbance	Fire	Annual area burned	No	Moderate	45	In some recent years, annual area burned has approached presettlement levels, but overall average is still lower than NRV due to fire suppression efforts.
Function	Disturbance	Fire	Fire season	No	Moderate	50	Although overall patterns are similar, fire season has been getting longer with warmer spring and early summer temperatures in addition to a lower snow-to-rain ratio.
Function	Disturbance	Insect outbreaks	Tree mortality from insects	Unknown	Low	53, 61	The lack of presettlement information on insect outbreaks makes it difficult to conclude whether current insect activity is within the NRV or not.
Function	Disturbance	Disease	Tree mortality from disease	No	Low	53, 61	Impacts from disease are likely higher due to the increased spread of preexisting and newly introduced diseases.

Ecosystem attribute	Indicator group	Indicator	Variable	Within NRV[a]	Confidence	See discussion on page:	Notes
Structure	Physiognomy	Functional groups/ growth forms	Proportion early-/middle-/ late-seral forest	No	Moderate	64	There are fewer small patches and more, large, landscape-level patches of early-seral forest (or shrubfields) than there were historically based on what we know about presettlement disturbance regimes and descriptions from early settlers. There is an overabundance of middle-seral forest due to fire suppression and an under abundance of late-seral forests due to logging.
Structure	Physiognomy	Gap size distribution	Gap size	No	High	70, 85	Gap sizes are generally decreasing (in undisturbed forests), but also increasing in disturbed forests because of more severe disturbance.
Structure	Physiognomy	Overstory density	Number of trees per unit area	No	High	74	Modern forests have a higher density of trees.
Structure	Physiognomy	Overstory density	Number of large trees per unit area	No	High	78	There are currently fewer large trees per unit area.
Structure	Physiognomy	Snag density	Number of snags per unit area	No	Low	88	Slightly higher snag densities today than under NRV.
Structure	Physiognomy	Tree size class distribution	Tree size class distribution	No	High	78	Overabundance of small-diameter trees and fewer large-diameter trees.
Structure	Physiognomy	Average tree size	Mean diameter at breast height	No	High	78	The increase in small-diameter trees and the decrease in large-diameter trees has led to a drop in mean tree size.
Structure	Physiognomy	Canopy cover	Percentage of cover	No	Low	83	Based on the limited information available, canopy cover appears to be higher today, although confidence in this assertion is low.
Structure	Physiognomy	Shrub cover	Percentage of cover	Yes (no)	Moderate	97	Shrub cover is probably within NRV, yet we lack quantitative information on presettlement shrub cover. The distribution of shrub cover on the landscape is likely different than it was historically, with less shrub cover in forested areas and larger contiguous shrub patches due to recent high-severity fires.
Structure	Physiognomy	Grass and forb cover	Percentage of cover	No	Low	100	Herbaceous cover is likely lower.

Table 9—Summary of probable deviations from the natural range of variation (NRV) for yellow pine and mixed-conifer forests in the assessment area (continued)

Ecosystem attribute	Indicator group	Indicator	Variable	Within NRV[a]	Confidence	See discussion on page:	Notes
Structure	Physiognomy	Coarse woody debris	Pieces of coarse woody debris per unit area	No	Low	90	Although coarse woody debris is highly variable across the landscape, evidence suggests that current averages are either somewhat higher than historical averages or at the higher end of the NRV and include much more material in advanced stages of decay.
Structure	Physiognomy	Litter and fine fuels	Depth of litter and fine fuels	No	Low	92	Based on inference from reference sites outside of the assessment area, litter and fine fuel depths are higher than they were historically.
Structure	Productivity	Tree basal area	Basal area	No	Moderate	82	Modern basal area appears to be higher in modern forests than it was in presettlement forests due to the strong increase in smaller diameter trees, which more than compensates for declines in basal area as a result of losses in large-diameter trees. Lack of historical data makes it difficult to be confident.
Composition	Functional diversity	Functional groups/ growth forms	Proportion of shade tolerant vs. shade intolerant spp.	No	High	104	Shade-tolerant species have become more abundant.
Composition	Species diversity	Species richness	Plant species richness	Yes	Moderate	104, 108	Information is lacking, but it is likely that overall woody species richness levels in modern forests are similar to presettlement forests, while distributions of certain species/ communities have shifted. It is likely that herbaceous vegetation as a component of yellow pine and mixed-conifer forests has changed dramatically.

NRV presettlement reference period is assumed to refer to 1500/1600 to 1850, unless otherwise indicated in notes. NRV for most indicators/variables also includes information from contemporary reference sites.

[a] As defined as the range of means from multiple sources.

- Changes in forest structure, fuels, and climate have altered the way in which fire behaves in modern forests. The proportional area of fires burning at high severity today (areas where most trees are killed) is generally much greater than in average presettlement period fires. Modern fires that burn under moderate conditions (i.e., absence of extreme fire weather), however, tend to burn with proportions of low, moderate, and high severity that are similar to historical fires, especially when temperature inversions are present. However, such fires are usually extinguished soon after ignition.
- Overall, the primary role of fire has changed from one of forest maintenance (of relatively open-canopy, fuel-limited conditions with dominance primarily by fire-tolerant species) to one of forest transformation, where dense stands of less fire-tolerant species and heavy fuel accumulations are more likely to burn at high-severity, resulting in major ecosystem changes.

The conclusions provided above and in table 9 should be applied to specific management areas only when using the guidance of local expertise. Within each section of this document we have done our best to provide the range of conditions that exist throughout the assessment area, including major exceptions. We recommend that specific sections be referred to directly for guidance on how a specific NRV or departure may or may not be relevant to a given area.

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English Equivalents

When you have:	Multiply by:	To get:
Centimeters (cm)	0.394	Inches
Meters (m)	3.28	Feet
Kilometers (km)	0.62	Miles
Hectares (ha)	2.47	Acres
Square kilometers (km^2)	0.386	Square miles
Square meters per hectare (m^2/ha)	4.357	Square feet per acre
Degrees Celsius (°C)	1.8 °C + 32	Degrees Fahrenheit

Metric Equivalents

When you have:	Multiply by:	To get:
Inches (in)	2.54	Centimeters
Feet (ft)	0.305	Meters
Miles (mi)	1.609	Kilometers
Acres (ac)	0.405	Hectares
Square miles (mi^2)	2.59	Square kilometers
Square feet per acre (ft^2/ac)	0.229	Square meters per hectare

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