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Mixed Evergreen Forests
Northern California Climate Change Vulnerability Assessment Synthesis

An Important Note About this Document: *This document represents an initial evaluation of vulnerability for mixed evergreen forests in northern California based on expert input and existing information. Specifically, the information presented below comprises vulnerability factors selected and scored by regional experts, relevant references from the literature, and peer-review comments and revisions (see end of document for a glossary of terms and brief overview of study methods). The aim of this document is to expand understanding of habitat vulnerability to changing climate conditions, and to provide a foundation for developing appropriate adaptation responses.*

Peer reviewers for this document included by Kimberly Baker (Environmental Protection Information Center), Dominick DellaSala (Geos Institute), Jeff Jones (U.S. Forest Service), and Chad Roberts (Tuleyome). Vulnerability scores were provided by Eureka workshop participants.

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Habitat Description

Within the northern California study region, mixed evergreen forests are distributed inland of the fog belt and associated coastal forests at relatively low elevations (Stuart & Stephens 2006; Sawyer 2007) between 150–1,200 m (500–4,000 ft; Bingham & Sawyer 1991; Vuln. Assessment Workshop, pers. comm., 2017). Forest structure and composition can vary widely depending on moisture balance, disturbance history (e.g., fire), and site conditions (e.g., aspect, slope, soil properties; (Whittaker 1960; Sawyer et al. 1977; Bingham & Sawyer 1991; Spies et al. 2006;

Sawyer 2007), which have historically created a mosaic of forest patches that result in high heterogeneity across the landscape (Bingham & Sawyer 1991; Spies et al. 2006). Douglas-fir (*Pseudotsuga menziesii*) is frequently the dominant or co-dominant species in the overstory (Whittaker 1960; Franklin & Dyrness 1973; Sawyer et al. 1977; Bingham & Sawyer 1991). Mixed-evergreen forests typically include a significant lower canopy composed largely of hardwood species, and an often-sparse shrub layer (Whittaker 1960; Franklin & Dyrness 1973; Sawyer et al. 1977; Bingham & Sawyer 1991). Tanoak (*Notholithocarpus densiflorus*) is generally the most abundant hardwood, and is a characteristic species in the mixed evergreen association (Sawyer et al. 1977). Other common hardwoods include Pacific madrone (*Arbutus menziesii*), canyon live oak (*Q. chrysolepis*), giant chinquapin (*Chrysolepis chrysophylla*), California bay (*Umbellularia californica*), California black oak (*Q. kelloggii*), Oregon white oak (*Q. garryana*), and bigleaf maple (*Acer macrophyllum*), among others (Franklin & Dyrness 1973; Sawyer et al. 1977; Bingham & Sawyer 1991). In some areas other conifers can occur, including ponderosa pine (*Pinus ponderosa*), sugar pine (*P. lambertiana*), incense cedar (*Calocedrus decurrens*), white fir (*Abies concolor*), and Port-Orford-cedar (*Chamaecyparis lawsoniana*; Franklin & Dyrness 1973; Sawyer et al. 1977; Bingham & Sawyer 1991; Spies et al. 2018a). With increasing elevation, as well as on some soils or slope aspects, mixed evergreen forests gradually lose the prominent hardwood component and assume characteristics of mixed conifer forest (Franklin & Dyrness 1973; Bingham & Sawyer 1991). Mixed-evergreen forests with open stand structures typically include substantial areas of shrublands, particularly on mafic substrates (Duren & Muir 2010).

Northern California mixed evergreen forests also include mature and old-growth stands (Bingham & Sawyer 1991; Spies et al. 2018a), which are characterized by large-diameter trees, varied tree sizes, multi-layered canopies, abundant snags, and dead wood (Sawyer et al. 1977; Old Growth Definition Task Group 1986; Bingham & Sawyer 1991). Because of the steep climate gradients and high landscape complexity of the region, mixed evergreen forests are highly diverse at multiple spatial scales and support many endemic species and unique vegetation communities (Whittaker 1960; Sawyer et al. 1977; DellaSala et al. 1999; Sawyer 2007). Hundreds of wildlife species utilize this forest type, including those dependent on old-growth habitat characteristics, such as the Pacific fisher (*Pekania pennant pacifica*), northern spotted owl (*Strix occidentalis caurina)*, and northern Goshawk (*Accipiter gentilis*; Raphael 1991; Zielinski et al. 2004; Spies et al. 2006; Keane 2008; Norgaard et al. 2016).

Many plant and wildlife species characteristic of mixed evergreen forests hold cultural value for northern California tribes (Lake 2007; Halpern 2016; Norgaard et al. 2016; Karuk Tribe 2019). Tanoak, in particular, is considered a cultural keystone species, providing acorns for food and holding considerable cultural and spiritual significance (Anderson 2005; Norgaard 2005; Bowcutt 2013; Halpern 2016; Norgaard et al. 2016; Karuk Tribe 2019). Mature tanoak stands also support the prized tanoak mushroom, more widely known as the American matsutake (*Tricholoma magnivelare*; Anderson & Lake 2013). Other forest species utilized by northern California tribes for food, fiber, and other materials include Pacific madrone, California bay, black oak, Oregon white oak, canyon live oak, giant chinquapin, ponderosa pine, sugar pine, California hazel (*Corylus cornuta californica*), evergreen huckleberry (*Vaccinium ovatum*),

salmonberry (*Rubus spectabilis*), manzanita (*Arctostaphylos* spp.),.), and many species of fungi (Schenck & Gifford 1952; Anderson 2005; Lake 2007; Norgaard et al. 2016). Culturally-valued wildlife species include the pileated woodpecker (*Dryocopus pileatus*), black-tailed deer (*Odocoileus hemionus columbianus*), Roosevelt elk (*Cervus canadensis roosevelti*), Pacific fisher, and porcupine (*Erethizon dorsatum*; Norgaard et al. 2016; Karuk Tribe 2019). Cultural burning practices are used to increase the productivity and predictability of plant and animal resources by maintaining forest patches in various successional stages (Voggesser et al. 2013; Norgaard et al. 2016; Karuk Tribe 2019).

Executive Summary

The relative vulnerability of mixed evergreen forests in northern California was evaluated as moderate-high by regional experts due to moderate-high sensitivity to climate and non-climate stressors, high exposure to projected future climate changes, and moderate adaptive capacity.

Mixed Evergreen Forests	Rank	Confidence
Sensitivity	Moderate-High	Moderate
Future Exposure	High	Moderate
Adaptive Capacity	Moderate	High
Vulnerability	**Moderate-High**	**Moderate**

Sensitivity & Exposure Summary	*Climate and climate-driven factors*: • Precipitation amount and timing, climatic water deficit, soil moisture, air temperature, drought *Disturbance regimes*: • Wildfire, disease, insects *Non-climate stressors*: • Fire suppression, timber harvest, roads/highways/trails *Other sensitivity factors*: • Mycorrhizal function

Mixed evergreen forests are sensitive to factors that increase water demand (e.g., air temperature) or decrease water availability (e.g., soil moisture, precipitation, drought). Changes in site water balance can alter patterns of tree growth and mortality, resulting in shifts in species composition, forest structure, and regeneration. Forest heterogeneity and productivity was historically maintained through frequent wildland fire, which included both natural and anthropogenic ignitions by northern California tribes. Changes in the frequency, timing, and/or intensity of disturbances (e.g., wildfire, disease, insect pests) may cause more extensive tree mortality, especially in stands where increased competition for soil moisture has decreased tree vigor. Historical logging following by decades of fire suppression has contributed to significant shifts in forest structure, reducing habitat heterogeneity and increasing vulnerability to disturbance-related mortality. In addition, stand-level shifts in species composition have resulted in the loss of culturally-valued tanoak groves and associated plant and fungi resources (e.g., tanoak mushrooms). Logging and road/highway/trail networks have also fragmented mixed evergreen forests, spread introduced pathogens across the landscape

(e.g., sudden oak death, Port-Orford-cedar root rot), and can also contribute to altered wildfire regimes by increasing human ignitions. Finally, mycorrhizal networks that facilitate post-disturbance conifer establishment and increase drought resistance in mixed evergreen forests may also be adversely affected by changes in wildfire intensity and patterns of disease.

Adaptive Capacity Summary	*Factors that enhance adaptive capacity*: + Extensive at low to middle elevations in the North Coast Range and Klamath Mountains + High landscape heterogeneity and forest structural diversity, which increases resistance and supports relatively rapid recovery from disturbances + Tanoak and many associated species highly valued by northern California tribes *Factors that undermine adaptive capacity*: − Declining structural integrity due to logging and fire suppression, resulting in reduced forest resistance to uncharacteristically large and/or severe disturbances − Limited public understanding of the ecosystem services provided by this habitat − Management efforts unlikely to stop the spread of sudden oak death

Mixed evergreen forests are relatively extensive at low to middle elevations in the North Coast Range and western Klamath Mountains. However, historical logging and associated road networks have fragmented and degraded this forest type, reducing structural integrity and increasing vulnerability to disturbances such as uncharacteristically severe wildfires and severe drought. Resistance to climate stressors and altered disturbance regimes is supported by high landscape heterogeneity, and high structural and species diversity may support shifts in forest composition towards species better adapted to warmer, drier conditions. However, climate changes may also slow forest recovery following disturbances. Despite its critical importance to northern California tribes, public understanding of the value of this habitat is limited. Societal support for management varies widely depending on location and scale considered, and the complexity of management challenges has made actively addressing the impacts of sudden oak death and altered fire regimes more difficult.

Sensitivity and Exposure

Mixed evergreen forests were evaluated by regional experts as having moderate-high overall sensitivity (moderate confidence in evaluation) and high overall future exposure (moderate confidence) to climate and climate-driven factors, changes in disturbance regimes, and non-climate stressors.

Climate projections suggest that much of the current mixed evergreen forest distribution in northern California will remain within climatically suitable areas by the end of the century (Thorne et al. 2016, 2017). Some studies predict range expansion into areas of forest currently dominated by conifers due to increased hardwood productivity in warmer temperatures (Lenihan et al. 2008), while others suggest that overall range contractions are likely for tanoak-dominated forests as the climate becomes warmer and drier (Ackerly et al. 2015; Serra-Diaz et al. 2016). Several component species within mixed evergreen forests have already shown some evidence of distributional shifts; for instance, tanoak has begun to shift upslope despite overall

range contractions (Serra-Diaz et al. 2016). However, the direction of these shifts is not consistent (Serra-Diaz et al. 2016). Due to the narrower range of climate conditions suitable for seedling establishment compared to adult persistence, an overall trend of declining tree recruitment is projected across all forest types in California, especially at the xeric edges of a species' range (Dobrowski et al. 2015). However, seedling recruitment in several species typical of mixed evergreen forests is projected to increase, including Pacific madrone, incense cedar, California bay, and Pacific dogwood (*Cornus nutallii*; Dobrowski et al. 2015). Variation in topography and forest

Potential Changes in Habitat Distribution
• Much of the current distribution is expected to remain climatically suitable by 2100 • Increased hardwood productivity may allow expansion into areas currently dominated by conifers, though some studies project a decline in tanoak-dominated forests as the climate becomes warmer and drier • Seedling recruitment in several component species is projected to increase, despite overall declines in seedling recruitment across all forest types
Source(s): Lenihan et al. 2008; Ackerly et al. 2015; Dobrowski et al. 2015; Thorne et al. 2016

structure may mediate the impact of changing climate factors on seedling recruitment by providing diverse microsites that facilitate species establishment (Dobrowski et al. 2015). Because patterns of regeneration and distributional shifts vary, there is potential for novel species assemblages under future climate conditions (Williams & Jackson 2007; Dobrowski et al. 2015; Serra-Diaz et al. 2016).

Potential refugia for mixed evergreen species may include north-facing slopes, valley bottoms and riparian areas, sites with relatively high precipitation, and areas surrounding seeps, springs, and perennial water bodies such as lakes (Dobrowski 2011; Olson et al. 2012; Flint et al. 2013). Stands of mature and old-growth forest with dense, closed canopies also influence light, temperature, humidity, and soil moisture, potentially providing microrefugia that could protect vulnerable species from thermal stress, increasingly dry conditions, and wildfire (Olson et al. 2012; Frey et al. 2016; Ellison et al. 2017).

Sensitivity and future exposure to climate and climate-driven factors

Regional experts evaluated mixed evergreen forests as having moderate-high sensitivity to climate and climate-driven factors (high confidence in evaluation), with an overall moderate-high future exposure to these factors within the study region (moderate confidence). Key climatic factors that affect mixed evergreen forests include precipitation amount and timing, climatic water deficit, soil moisture, air temperature, and drought.[1]

Precipitation amount and timing, soil moisture, and climatic water deficit
Water availability and corresponding moisture stress is an important driver of tree recruitment (Ackerly et al. 2015), growth (Restaino et al. 2016), and mortality (van Mantgem et al. 2009; Allen et al. 2010), ultimately playing a large part in determining forest structure and distribution across the western U.S. (Stephenson 1998; Lutz et al. 2010; Chmura et al. 2011; Anderegg et al.

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[1] All climate and climate-driven factors presented were ranked as having a moderate or higher impact on this habitat type.

2015b). Changes in the amount and/or timing of precipitation and associated changes in soil moisture are likely to alter patterns of moisture stress in mixed evergreen forests, potentially leading to shifts in species composition and habitat distribution along moisture gradients (Chmura et al. 2011; Restaino et al. 2016). One particularly useful way to measure moisture stress is climatic water deficit (CWD), which provides a "plant-relevant" metric to account for the interaction between water (precipitation) and energy (temperature; Stephenson 1998).[2] Site-specific factors such as soil depth and drainage significantly affect the water-holding capacity of soil; in combination with topographic features (e.g., north-facing slopes) that affect evaporative demand, these can mediate increases in CWD at a site-level scale (Stephenson 1998; Dobrowski 2011; Flint et al. 2013).[3] Differences in the drivers of water balance on a particular site contribute to distinct patterns of plant species composition and forest productivity (Stephenson 1998).

Although the direction and amount of change in future precipitation projections for California are highly uncertain, warmer temperatures and associated increases in evaporative demand mean that even areas where precipitation may increase are expected to see a rise in CWD (Thorne et al. 2015; Dobrowski et al. 2015; Restaino et al. 2016; Micheli et al. 2018). Increases in CWD have already reduced Douglas-fir growth rates, and water stress is likely to further limit growth over the coming century, particularly on drier sites and at the southern edge of a species' range (Restaino et al. 2016). However, CO_2 fertilization may compensate for moisture stress to a degree (Vuln. Assessment Reviewer, pers. comm., 2018). Increased CWD is associated with enhanced vulnerability to stressors such as drought (Young et al. 2017), insects (Fettig et al. 2007; Fettig 2012; Kolb et al. 2016), disease (Millar & Stephenson 2015; Kolb et al. 2016), and wildfire (van Mantgem et al. 2013). Low precipitation and high CWD have also been correlated with increased fire size and rate of spread, largely due to reduced fuel moisture (Littell et al. 2009; Abatzoglou & Kolden 2013; Parks et al. 2014; McKenzie & Littell 2017).

Climate-related changes in patterns of post-fire forest regeneration could result in shifts in species composition towards drought-tolerant hardwoods and shrubs (Lenihan et al. 2008; Welch et al. 2016; Vuln. Assessment Reviewer, pers. comm., 2018). However, topographical variation creates moist microsites that can support post-fire conifer seedling recruitment even in dry years (Donato et al. 2009a). Existing vegetation can also provide protected microsites where soil moisture is higher compared to surrounding, more exposed areas (Minore 1986), potentially facilitating conifer regeneration in areas burned at low- to moderate-severity (Fryer 2008; Irvine et al. 2009).

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[2] CWD, calculated as potential evapotranspiration (PET) minus actual evapotranspiration (AET), measures the degree to which the impact of local atmospheric conditions (particularly air temperature and relative humidity) on plants and soil exceeds available moisture (Stephenson 1998). The balance between water supply and demand in California shifts over the course of the year, with CWD increasing as soil moisture from the winter rains is depleted by late spring and evapotranspiration increases in warmer months (Stephenson 1998).

[3] Soil moisture is based on CWD (i.e., balance between water supply and water demand) and soil properties, including porosity, depth, and underlying geology. These properties determine the soil water-holding capacity (i.e., how much moisture can be stored and used for plant evapotranspiration; Stephenson 1998; Flint et al. 2013).

Increased precipitation could enhance forest productivity to some degree (North et al. 2016). However, this has the potential to increase fire severity due to a greater availability of potential fuels (e.g., brush and vegetation; (Parks et al. 2016). Increases in winter rainfall would likely also enhance spore production and transmission of the pathogen that causes sudden oak death, increasing infection risk and associated mortality (Davidson et al. 2005; DiLeo et al. 2014). By contrast, dry conditions and low soil moisture have a strong limiting effect on sudden oak death (Venette & Cohen 2006), and reduced duration and magnitude of winter precipitation could limit the spread and severity of this disease (Kliejunas 2011).

Regional Precipitation, Climatic Water Deficit (CWD), & Soil Moisture Trends[4]	
Historical & current trends: 7.2 cm (2.8 in) increase in mean annual precipitation and 0.4 cm (0.2 in) increase in average annual CWD between 1900 and 2009 for the Northwestern California ecoregion (Rapacciuolo et al. 2014) - No trends available for soil moisture	*Projected future trends:* 20% decrease to 34% increase in mean annual precipitation by 2100 (compared to 1951–1980) for the North Coast, Northern Coast Range, Northern Interior Coast Range, and Klamath Mountain ecoregions (Flint et al. 2013; Flint & Flint 2014)[5] - Increases in average annual CWD by 2100 (compared to 1951–1980; Flint et al. 2013; Flint & Flint 2014): 9–29% increase on the North Coast 7–24% increase in the Northern Coast Range 5–16% increase in the Northern Interior Coast Range 10–32% increase in the Klamath Mountains - Seasonal changes are projected to be more significant as the wet season becomes wetter and shorter (i.e., later onset of fall rains and earlier onset of summer drought) and the dry season becomes drier and longer (Pierce et al. 2018; Swain et al. 2018) - Overall, interannual variability is expected to increase (Pierce et al. 2018; Swain et al. 2018) - Increased CWD and decreased top-level soil moisture is likely even if precipitation increases

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[4] Trends in climate factors and natural disturbance regimes presented in this and subsequent summary tables are not habitat-specific; rather, they represent broad trends and future projections for the study region. The precipitation, temperature, climatic water deficit, and snowpack projections for this project are derived from the Basin Characterization Model, which uses modified Jepson ecoregions (Flint et al. 2013; Flint & Flint 2014). Projections for all other factors are based on a review of relevant studies in the scientific literature. For this project, exposure was evaluated by calculating the magnitude and direction of projected change within the modified Jepson ecoregions that include habitat distribution within the study geography.

[5] Projections for changes in annual and seasonal precipitation by ecoregion can be found in the full climate impacts table (https://bit.ly/2LHgZaG).

Regional Precipitation, Climatic Water Deficit (CWD), & Soil Moisture Trends[4]	
	due to temperature-related changes in evaporative demand (Thorne et al. 2015; Micheli et al. 2018; Pierce et al. 2018)
Summary of Potential Impacts on Habitat *(see text for citations)*	

- Reduced tree growth, especially at the southern edge of the species' range
- Increased vulnerability to disease outbreaks and insect attacks
- Increased risk of large-scale forest die-off following drought events
- Increased wildfire activity (e.g., fire size and rate of spread) due to reduced fuel moisture
- Changes in post-fire vegetation dynamics, including potential shifts in species composition

Air temperature

One of the primary ways that warming air temperatures impact mixed evergreen forests is by driving increased evaporative demand (Ackerly et al. 2015; Restaino et al. 2016). This increases water stress and is associated with growth declines (Irvine et al. 2009; Restaino et al. 2016), tree mortality (van Mantgem et al. 2009; Restaino et al. 2016) and shifts in species composition and forest distribution (Lenihan et al. 2008; Ackerly et al. 2015). However, warmer temperatures may also increase productivity and seedling recruitment in some hardwood species associated with mixed evergreen forests (e.g., Pacific madrone, California bay), enhancing competition with conifers and potentially leading to shifts in species composition (Lenihan et al. 2008; Dobrowski et al. 2015) and/or upslope expansion into areas typically dominated by coniferous forest (Lenihan et al. 2008).

Warmer temperatures may contribute to inland expansion of sudden oak death (Venette 2009), shifting forest structure and species composition as high rates of mortality occur in tanoak and several species of true oaks (e.g., black oak; Rizzo & Garbelotto 2003; McPherson et al. 2010; Metz et al. 2017). Warmer temperatures can also alter the timing and/or synchronicity of bark beetle life cycles (Bentz et al. 2010) and increase the vulnerability of mixed evergreen forests to mortality from drought and disease (Millar & Stephenson 2015). Finally, warmer temperatures are associated with reduced fuel moisture and increases in the amount of forest area burned by wildfire across the western U.S. (Abatzoglou & Williams 2016).

Dense, closed-canopy stands of mature and old-growth forest, particularly those on north-facing slopes, can provide cooler microclimates within the forest understory for insects, amphibians, and small mammals (Olson et al. 2012; Frey et al. 2016). Structural characteristics within these forests, such as tall canopies, high aboveground biomass, and vertical complexity, can reduce air temperatures by up to 2.5°C (4.5°F) near the ground, potentially allowing sensitive species to persist within these thermal refugia (Frey et al. 2016).

Regional Air Temperature Trends	
Historical & current trends:	*Projected future trends:*
• 0.2°C (0.4°F) increase in the average annual	• 2.2–5.4°C (4.0–9.7°F) increase in the average

Regional Air Temperature Trends	
temperature between 1900 and 2009 for the Northwestern California ecoregion (Rapacciuolo et al. 2014) ○ No seasonal temperature trends available	annual temperature by 2100 (compared to 1951–1980) for the North Coast, Northern Coast Range, Northern Interior Coast Range, and Klamath Mountain ecoregions (Flint et al. 2013; Flint & Flint 2014) ○ 1.9–4.8°C (3.4–8.6°F) increase in average winter minimum temperatures ○ 2.2–6.3°C (4.0–11.3°F) increase in average summer maximum temperatures

Summary of Potential Impacts on Habitat *(see text for citations)*

- Increased forest water stress due to greater evaporative demand, resulting in decreased tree growth and higher rates of tree mortality
- Shifts in species composition and forest distribution
- Increased hardwood productivity, potentially resulting in the displacement of coniferous forests with mixed evergreen
- Increased fuel aridity and associated increase in wildfire area burned

Drought

Drought events can cause significant tree mortality (Anderegg et al. 2013; Allen et al. 2015; McDowell & Allen 2015; Anderegg et al. 2015b; Young et al. 2017), especially in older stands dominated by large trees (McDowell & Allen 2015; Young et al. 2017) and in dry areas or sites with high tree density where competition for soil moisture resources is increased (Allen et al. 2010; Young et al. 2017). Mortality rates typically increase as drought progresses (Young et al. 2017), possibly due to the depletion of stored carbohydrate reserves and/or gradually declining photosynthesis as hydraulic damage occurs (Anderegg et al. 2015b, 2015a). During periods of drought, stressed trees are also more vulnerable to mortality from secondary causes, including wildfire, insect outbreaks, and disease (Allen et al. 2010; van Mantgem et al. 2013; McDowell & Allen 2015; Millar & Stephenson 2015; Kolb et al. 2016; Young et al. 2017).

Forests stressed by drought typically experience growth declines, which are greater in drier portions of a species' range and can persist for 2–4 years following a drought event (Anderegg et al. 2015a). Seedlings are more sensitive to drought compared to mature trees, and reduced recruitment during periods of drought could contribute to shifts in composition based on variable species responses to drought (Dobrowski et al. 2015). In general, species with intermediate drought and shade tolerance (e.g., Pacific madrone, incense cedar, tanoak, and chinquapin) show fewer projected recruitment declines compared to those at the high and low ends of the spectrum (Dobrowski et al. 2015).

Under future climate conditions, prolonged and/or severe droughts may cause large-scale dieback (Allen et al. 2015; McDowell & Allen 2015; Millar & Stephenson 2015), with higher rates of mortality occurring where drought-stressed trees are further impacted by climate-driven increases in insect outbreaks and disease (Allen et al. 2010, 2015; Millar & Stephenson

2015; Kolb et al. 2016). Additionally, wildfire activity increases during periods of drought (Whitlock et al. 2008; Colombaroli & Gavin 2010; Littell et al. 2016), largely due to decreases in vegetation moisture (Abatzoglou & Williams 2016; McKenzie & Littell 2017) and the presence of drought-killed trees that may drive larger and/or more severe fires (Vuln. Assessment Reviewer, pers. comm., 2018). However, studies of charcoal and pollen records in southwestern Oregon and northwestern California found that mixed evergreen forests in the Klamath-Siskiyou region have demonstrated high resilience to multi-decadal droughts and associated increases in severe fires over the past two millennia (Briles et al. 2008; Colombaroli & Gavin 2010).

Regional Drought Trends	
Historical & current trends: - Drought years have occurred twice as often over the last two decades compared to the previous century (Diffenbaugh et al. 2015) 2012–2014 drought set records for lowest precipitation, highest temperatures, and most extreme drought indicators on record (Griffin & Anchukaitis 2014; Diffenbaugh et al. 2015)	*Projected future trends:* - Drought years are twice as likely to occur over the next several decades due to increased co-occurrence of dry years with very warm years (Cook et al. 2015) 80% chance of multi-decadal drought by 2100 under a high-emissions scenario (Cook et al. 2015) - Severe droughts that now occur once every 20 years will occur once every 10 years by 2100 and once-in-a-century drought will occur once every 20 years (Pierce et al. 2018)
Summary of Potential Impacts on Habitat *(see text for citations)*	
- Reduced tree growth and increased mortality, especially on drier sites and in dense stands - Increased vulnerability to mortality from insects and disease in drought-stressed trees - Increased wildfire activity during periods of drought due to reductions in vegetation moisture	

Sensitivity and future exposure to changes in natural disturbance regimes

Regional experts evaluated mixed evergreen forests as having moderate sensitivity to changes in natural disturbance regimes (high confidence in evaluation), with an overall high future exposure to these stressors within the study region (high confidence). Key natural disturbance regimes that affect mixed evergreen forests include wildfire, disease, and insects.

Wildfire

Mixed evergreen forests historically experienced a mixed-severity fire regime, which included higher proportions of low- and moderate-severity fire interspersed with patchily distributed areas of high-severity fire (Odion et al. 2004, 2010; Donato et al. 2009b; Perry et al. 2011).[6] The

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[6] Definitions of mixed-severity fires vary somewhat, but typically describe a unique fire regime comprised of irregular patches of low-, moderate-, and high-severity fire. Patch sizes and the proportion of high-severity fire depend on forest type, as well as topography, weather, fuel, disturbance history, and other factors that influence fire behavior (Halofsky et al. 2011; Perry et al. 2011; Hessburg et al. 2016). Patches are generally described at an intermediate or stand-level spatial scale, as opposed to individual trees or landscape-scale (Halofsky et al. 2011; Perry et al. 2011).

complex patterns of fire that characterize mixed-severity fires are typical of the Klamath and North Coast regions where topographic and geologic complexity is high (Taylor & Skinner 2003; Skinner et al. 2006; Stuart & Stephens 2006; Halofsky et al. 2011; Perry et al. 2011; Odion et al. 2014), and the resulting mosaic of forest patches at varying stages of succession creates a diverse forest structure in the region (Odion et al. 2004; Stuart & Stephens 2006; Perry et al. 2011; Steel et al. 2015). Prior to Euro-American settlement in the 1850s, fire return intervals in mixed evergreen forests were roughly 15–80 years (Van de Water & Safford 2011; Safford & Van de Water 2014), with some sources suggesting that as little as 5% of that occurred within high-severity patches (Stephens et al. 2007; Steel et al. 2015). However, fire return intervals have increased significantly since the implementation of fire suppression in the early- to mid-1900s (Taylor & Skinner 1998; Safford & Van de Water 2014).[7] Natural lightning ignitions account for the majority of the area burned in the Klamath Mountains in the last several decades (Miller et al. 2012; Skinner et al. 2018), and lightning ignitions increase with elevation and distance from the coast (van Wagtendonk & Cayan 2008). At lower elevations where lightning ignitions are relatively rare, human ignitions dominate wildland fire regimes, particularly in areas with more roads and higher levels of human activity (van Wagtendonk & Cayan 2008; Syphard et al. 2017; Balch et al. 2017; Keeley & Syphard 2017).

Charcoal and sediment records from the Klamath-Siskiyou region suggest that the fire history of mixed evergreen forests has been highly episodic and was strongly associated with climatic variability, a pattern that mirrors study results for other forested landscapes in western North America (Whitlock et al. 2008; Colombaroli & Gavin 2010). Studies in southwestern Oregon indicate that prolonged periods without significant fire existed in the past 2,000 years, and these were punctuated by periods of recurrent fire at short intervals, including some larger and/or more severe fires that caused significant erosion (Colombaroli & Gavin 2010). Periods of increased fire activity were strongly associated with warmer, drier periods (Whitlock et al. 2008), and the most severe fires occurred during decadal-scale droughts (Colombaroli & Gavin 2010). However, the decline of tribal burning, the legacy of fire suppression, and contemporary human land use and management have altered historical fire-climate relationships (Taylor et al. 2016; Syphard et al. 2017), making it more difficult to predict future ecosystem dynamics based on historical patterns.

Impacts of low- and moderate-intensity fire
Prior to Euro-American settlement, cultural burning practices played an important role in maintaining habitat composition and structure in northern California mixed evergreen forests, which are adapted to frequent fire (Anderson 2005; Stuart & Stephens 2006; Lake 2007; Bowcutt 2013; Halpern 2016; Norgaard et al. 2016; Karuk Tribe 2019). Northern California tribes manage forests by burning areas at different frequencies to maximize growth and productivity of cultural resources, particularly at lower elevations (Anderson 2005; Stuart & Stephens 2006; Lake 2007; Bowcutt 2013; Halpern 2016; Norgaard et al. 2016; Karuk Tribe 2019). Low- and moderate-intensity fires, such as those that occur during cultural burning,

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[7] Refer to the section on non-climate stressors for a more complete discussion of the impacts of fire suppression on mixed evergreen forests.

primarily consume fuels at the ground level, although moderate levels of mortality can occur in the lower canopy (Uchytil 1990; Fryer 2008; Halofsky et al. 2011; Perry et al. 2011; Hessburg et al. 2016). In tanoak groves, frequent late summer and fall burning at low intensities helps maintain an open stand structure while also allowing mature acorn-producing tanoaks to survive (Bowcutt 2013; Halpern 2016; Norgaard et al. 2016). Low- to moderate-intensity fire also minimizes the risk of losing mature tanoak stands to severe fire (Bowcutt 2013; Norgaard et al. 2016), reduces insects that impact tree health and acorn production (Halpern 2016), and releases nutrients from the forest litter into the soil (Norgaard et al. 2016). Fire benefits wildlife by creating cavities in large-diameter trees that can be used as dens/nesting sites (Long et al. 2016; Norgaard et al. 2016) and produces high-quality understory forage for deer and elk (Hebblewhite et al. 2009; Bowcutt 2013). Thus, cultural burning practices support high species diversity and increased food security for the tribes (Norgaard 2005; Halpern 2016; Norgaard et al. 2016; Karuk Tribe 2019).

Impacts of high-intensity fire
Higher-intensity fires result in substantial modifications in the pre-fire forest structure and composition by killing a large proportion of existing vegetation and favoring the growth and establishment of hardwoods and shrubs that sprout following topkill or germinate from buried seedbanks (Stuart & Stephens 2006; Perry et al. 2011; Cocking et al. 2012, 2015; North et al. 2016). Following high-severity fires, mixed evergreen forests are often dominated by hardwood trees (e.g., tanoak, black oak) and fire-stimulated shrubs (e.g., *Arctostaphylos*, *Ceanothus*), with conifers such as Douglas-fir gradually overtopping the early successional species after several decades (Shatford et al. 2007; Fontaine et al. 2009; Donato et al. 2009a; Odion et al. 2010; Welch et al. 2016). The irregular size and shape of higher-intensity patches associated with mixed-severity fire regimes contribute to the high species diversity and complex structure characteristic of mixed evergreen forests, in combination with the highly variable substrate and climatic conditions in the region (Halofsky et al. 2011; Perry et al. 2011; Hessburg et al. 2016). For instance, high heterogeneity within fire patterns result in forests that include complex structural features, including a mixture of clumped trees and more open areas, heterogeneous multi-layered vertical structure (Swanson et al. 2011; Perry et al. 2011; Donato et al. 2012; DellaSala et al. 2014). These complex early-seral habitats increase biodiversity by supporting many species associated with early-successional shrubs and hardwoods (e.g., birds; Ralph et al. 1991; Fontaine et al. 2009; Swanson et al. 2011; Donato et al. 2012; DellaSala et al. 2014).

The primary drivers of post-fire vegetation dynamics in mixed evergreen forests include climatic water deficit, seed source proximity, interspecific competition, and abiotic site conditions (e.g., soil moisture and parent material, topography, and elevation; Donato et al. 2009a; Irvine et al. 2009; Welch et al. 2016; Tepley et al. 2017). Within a mixed-severity fire, conifer regeneration typically occurs in patches unevenly distributed across the landscape (Shatford et al. 2007; Welch et al. 2016), which contributes to the formation of complex early-seral habitats (Donato et al. 2012). However, because most conifers do not sprout following fire, regeneration declines with increasing distance from a seed source and conifer regeneration may be delayed within the interior of larger high-severity patches (Donato et al. 2009a; Welch et al. 2016; Tepley et al. 2017; Collins et al. 2017), especially in dry areas (Tepley et al. 2017). Under changing climate

conditions, higher propagule pressure may be required to establish conifer seedlings on drier sites, which may also experience slower initial conifer growth due to greater competition for soil moisture (Tepley et al. 2017; Serra-Diaz et al. 2018). As a result, shifts in species composition towards a greater proportion of shrubs and fire-adapted sprouting hardwoods may occur in some areas (Welch et al. 2016). Because early-successional vegetation is more likely to burn at higher severity in the region due to high productivity and the dominance of evergreen hardwoods (Odion et al. 2004, 2010; Thompson & Spies 2010), these vegetation communities are generally perpetuated by cycles of reburning (Odion et al. 2010).

The likelihood of repeated high-severity fire is influenced by post-fire vegetation dynamics, fuel structure, pre-disturbance forest structure, management history, and post-fire management practices (Thompson et al. 2007; Odion et al. 2010; Thompson & Spies 2010; Campbell et al. 2016). Studies have demonstrated that mixed evergreen forest patches that experience repeated high-severity burns at relatively short intervals (e.g., 15 years between the Silver and Biscuit Fires in southwestern Oregon) generally recover rapidly and support very diverse plant and bird communities (Donato et al. 2009b; Fontaine et al. 2009). However, repeated high-severity fire can also alter forest structure by converting stands of mature, large-diameter tanoak or black oak to shrubby, multi-stemmed growth forms (Norgaard et al. 2016; Hammett et al. 2017; Nemens et al. 2018) by preventing the development of larger trees with fire-resistant characteristics before subsequent burns (Long et al. 2016, 2017; Nemens et al. 2018). Warmer, drier conditions resulting from climate change may result in widespread delays or reductions in forest regeneration due to increased competition for soil moisture (Tepley et al. 2017; Serra-Diaz et al. 2018). Thus, the overall impacts of repeated high-severity burns are largely dependent on scale. That is, small patches that reburn can result in unique plant communities that support high biodiversity, but large high-severity patches may contribute to losses of valued cultural resources (Norgaard et al. 2016), reduced structural complexity (Hammett et al. 2017; Nemens et al. 2018), and possible type conversion where forest regeneration is delayed due to warmer, drier conditions (Tepley et al. 2017).

Climate-driven changes in the fire regime are likely to impact mixed evergreen forests to a greater degree where forest structure has been greatly altered by a combination of logging followed by several decades of fire suppression (Colombaroli & Gavin 2010; Sensenig et al. 2013; Odion et al. 2014). For instance, young, even-aged conifer plantations characterized by homogeneous forest structure are more likely to experience high-severity fire (Odion et al. 2004; Thompson et al. 2007; Zald & Dunn 2018). Post-fire logging can also have negative impacts on ecosystem functioning, particularly in the short-term, such as reducing soil integrity, increasing erosion, and spreading invasive species (Karr et al. 2004; Beschta et al. 2004; Hutto 2006; Lindenmayer & Noss 2006; Reeves et al. 2006; Thompson et al. 2007; Hebblewhite et al. 2009; Donato et al. 2013; Campbell et al. 2016). For instance, the loss of snags and downed wood in areas impacted by post-fire salvage logging also reduces wildlife habitat and food resources, including berries, nuts, and insects (Hutto 2006; Long et al. 2016; Norgaard et al. 2016). However, the longer-term impacts of post-fire management activities can be positive, depending on the goals of the management activity and how it is carried out. Post-fire logging and subsequent planting can benefit the forest over the long term by creating habitat

heterogeneity, fostering future wildlife habitat, and treating fuels that could increase the risk of large high-severity reburns (Spies et al. 2018a). Planting, in particular, can accelerate recovery in high-priority areas that might otherwise undergo type conversion (Welch et al. 2016; Tepley et al. 2017). Planting can also be used to shift forest composition towards species that are more tolerant of projected climate conditions and/or to increase the presence of drought- and disease-resistant strains (Potter et al. 2019; Vuln. Assessment Reviewer, pers. comm., 2019).

Regional Wildfire Trends

Historical & current trends:	*Projected future trends:*

• 85% of U.S. Forest Service lands in northern California are burning less frequently compared to pre-1850 fire return intervals, largely due to fire suppression (Safford & Van de Water 2014)
• Fire size and total area burned increased on U.S. Forest Service lands in northwestern California between 1910-2008, with the highest values occurring after 2000 (Miller et al. 2012)
• Changes in large fires (over 400 ha) in the inland northern California/Sierra Nevada region since the 1970s (Westerling 2016):
   ○ 184–274% increase in frequency
   ○ 270–492% increase in total area burned
   ○ 215% increase in length of the fire season
• Changes in fire size, area burned, and fire frequency over the past several decades remain well below historical tribally-influenced frequency and extent of burning in California (Stephens et al. 2007)
• No significant trends in the average areal proportion of high-severity fire were documented in northwestern CA from 1984–2008 (Miller et al. 2012; Parks et al. 2015; Law & Waring 2015; Keyser & Westerling 2017)
   ○ The relatively short period of record for fire severity data may obscure long-term trends
   ○ To date, there are no peer-reviewed studies on trends in northern California fire severity that include data from the last ten years
	• State-wide, up to 77% increase in mean annual area burned and 50% increase in the frequency of extremely large fires (>10,000 ha) by 2100 (Westerling 2018)
	   ○ Greatest increases in burned area (up to 400%) occur in montane forested areas in northern California (Westerling et al. 2011; Westerling 2018)
	   ○ Less significant increases or possible decrease along the North Coast (Westerling et al. 2011)
	• Little projected change in fire severity in northwestern California by 2050 in models based solely on historical fire-climate relationships (Parks et al. 2016)
	   ○ However, human activity and fuel buildup from decades of fire suppression have altered historical fire-climate relationships (Taylor et al. 2016; Syphard et al. 2017; Wahl et al. 2019), and projections that incorporate these factors suggest that more significant increases in fire severity and size may occur (Mann et al. 2016; Wahl et al. 2019)
	• The majority of impacts to natural and human ecosystems come from extreme fire events (i.e., fires that have a low probability of occurring in any given place and time), which are likely to increase over the coming century (Westerling 2018)
	   ○ Generally, these patterns are not well-represented in studies that evaluate indices of mean fire size, intensity/severity, etc.

Summary of Potential Impacts on Habitat *(see text for citations)*
• *Immediate*:   ○ Loss of mature trees that act as seed sources and provide food (e.g., fruit and nuts) for wildlife

Regional Wildfire Trends

and humans, particularly within high-severity patches
    ○ Increased availability of standing snags and logs that provide habitat for cavity-nesting birds and other wildlife
• *Short-term (~2-year)*:
    ○ Delayed conifer recovery under warmer, drier conditions, especially in large high- severity patches where fewer seed sources may be available
    ○ Regeneration of tanoak, black oak, chinquapin, and other sprouting species following low- to moderate-severity fire that eliminates competing conifers
• *Long-term*:
    ○ Increased forest heterogeneity following low- to moderate- and mixed-severity fires, including patches of early-successional vegetation and persistence of oaks and chinquapin
    ○ Possible shifts in species composition and forest structure
    ○ Little or no development of later-successional forest stages where fires are too frequent to allow seedling survival

Diseases and insects

At endemic levels, diseases and insects act as natural disturbance agents within mixed evergreen forests, contributing to a heterogeneous forest structure by creating canopy gaps, snags, and downed logs (Spies et al. 2006; Hawkins & Henkel 2011; Meigs et al. 2015; Fettig 2016). Heart rot fungi (*Phellinus* spp.) is common in large trees, causing stem breakage and increasing downed wood within old-growth stands (Spies et al. 2006). Other common diseases in conifers include Heterobasidion root disease (*Heterobasidion* spp.), black stain root disease (*Leptographium wageneri*), Armillaria root disease (*Armillaria* spp.), and Swiss needle cast (*Phaeocryptopus gaeumannii*; in Douglas-fir; Cannon et al. 2016). Dwarf mistletoes (*Arceuthobium* spp.) can also impact mixed evergreen species, acting as parasites that take water and nutrients from the host tree's vascular system (Hawksworth & Wiens 1998; Kliejunas 2011). Historically, fire was utilized to control some insects and diseases that impacted important cultural resources (Bowcutt 2013; Halpern 2016). For instance, late summer/fall prescribed burns reduce insect populations that infest tanoak acorns (Halpern 2016).

Extensive tree mortality at the landscape scale can significantly alter forest composition and structure (Raffa et al. 2008; Hicke et al. 2016), although insects must overcome a variety of thresholds at multiple spatial and temporal scales in order to reach outbreak population levels (Raffa et al. 2008). For instance, outbreaks of bark beetles (e.g., Douglas-fir engraver beetles [*Scolytus unispinosus*] and defoliating insects (e.g., Douglas-fir tussock moth [*Orgyia pseudotsugata*]) are more likely as the abundance of host species increases and/or forests become denser and more homogenous, largely because trees that are already stressed due to greater competition are more vulnerable to attack (Fettig et al. 2007, 2013; Bentz et al. 2010; Fettig 2012). Some studies have suggested that insect-related tree mortality could impact fire behavior (e.g., spread, intensity, severity) over the following 1–3 decades by adding insect-killed biomass to the forest floor, altering fuel structure and increasing fuel availability (Hicke et al. 2012; Stephens et al. 2018). However, no clear relationship between insect outbreaks and increased fire activity or severity has been demonstrated in western U.S. forests (Bond et al.

2009; Harvey et al. 2013; Donato et al. 2013; Meigs et al. 2015, 2016; Hart et al. 2015; Reilly & Spies 2016). It is possible that insect-related tree mortality can and does impact fire behavior at local scales, but the complex interactions between weather, disturbance history, and numerous other factors likely obscure any consistent relationship across broader spatial and temporal scales.

Changing climate conditions may impact the prevalence and severity of insect outbreaks and disease by directly influencing disease production and transmission and/or insect development and survival (Kliejunas 2011; Sturrock et al. 2011; Weed et al. 2013; Kolb et al. 2016; Agne et al. 2018). Climate change is also likely to alter tree defenses, host susceptibility, and community interactions (Raffa et al. 2008; Bentz et al. 2010; Kliejunas 2011; Sturrock et al. 2011; Weed et al. 2013; Kolb et al. 2016; Agne et al. 2018). The spread and establishment of exotic insects and pathogens, in particular, may increase with changes in climate such as warmer temperatures and altered patterns of precipitation (Kliejunas 2011; Sturrock et al. 2011; Weed et al. 2013; Kolb et al. 2016), as well as anthropogenic factors (e.g., fire suppression, timber harvest; Jules et al. 2015; Haavik et al. 2015). In mixed evergreen forests, introduced pathogens of concern include sudden oak death (caused by the pathogen *Phytophthora ramorum*; Kliejunas 2011) and Port-Orford-cedar root rot (*P. lateralis*; (Jules et al. 2015).

Warming temperatures may contribute to earlier insect emergence, more completed life cycles within a season, and expanding distributions for both insects and disease, especially those typically restricted to more southern ranges (Bentz et al. 2010; Kliejunas 2011; Sturrock et al. 2011; North et al. 2016). Changes in seasonal precipitation patterns are also likely to impact pathogens that require moisture (e.g., fungal diseases; Chmura et al. 2011; Kliejunas 2011; Fettig et al. 2013), though drought could reduce the prevalence of those that depend on moist conditions for spore production (Davidson et al. 2005; Kolb et al. 2016). However, drought stress can increase the risk of large-scale insect outbreaks (Bentz et al. 2010; Weed et al. 2013; Kolb et al. 2016), as water stress reduces tree vigor and limits the ability of trees to expel attacking insects (Bentz et al. 2010; Weed et al. 2013; Kolb et al. 2016; Slack et al. 2017). *Armillaria* and other root rot pathogens are also more likely to colonize drought-stressed trees, suggesting that these could become more severe and/or widespread under drought conditions (Sturrock et al. 2011; Kolb et al. 2016). Conversely, injury from insects and disease can also increase tree vulnerability to drought stress and associated mortality (Kolb et al. 2016).

Sudden oak death is one of the most significant threats to mixed evergreen forests and has caused extensive tree injury and mortality in coastal forests since it was identified in the 1990s (Rizzo & Garbelotto 2003; Meentemeyer et al. 2004; McPherson et al. 2010). The disease is currently distributed from coastal central California northwards into Humboldt County, with an additional infected site occurring in Curry County, Oregon (Meentemeyer et al. 2011; Filipe et al. 2012). Sudden oak death causes high rates of mortality in tanoak and many true oaks (*Quercus* spp.), although dozens of plant species within northern California can be affected by the disease (Kliejunas 2011). Many are culturally important and are utilized for food, medicine, and other purposes (Ortiz 2008a; Bowcutt 2013; Voggesser et al. 2013); these include tanoak,

black oak, California bay, Pacific madrone, California hazel, evergreen huckleberry, salmonberry, and common manzanita (*Arctostaphylos manzanita*; Ortiz 2008a, 2008b).

Transmission of sudden oak death
Spread of *P. ramorum* is facilitated by host plants that support spore production, including California bay and tanoak (Rizzo & Garbelotto 2003; Cobb et al. 2010). California bay has the highest spore production and does not experience mortality (Rizzo & Garbelotto 2003; Davidson et al. 2005), and sudden oak death infection risk increases with higher densities of California bay, suggesting that the presence of this species is a major driver in transmission of *P. ramorum* (Cobb et al. 2010, 2012; McPherson et al. 2010; Haas et al. 2016). Moisture is required for *P. ramorum* spore production and survival, with frequency, timing, and intensity of winter rain and storms playing a large role in inoculum production (Davidson et al. 2005; Venette & Cohen 2006; Kliejunas 2011). Spores are dependent on aerial transmission, which primarily occurs through rainsplash, being carried downstream, and through human-mediated channels (Davidson et al. 2005), including movement of contaminated soil and/or within the nursery and ornamental plant trade (Davidson et al. 2005; Kliejunas 2011).

Impacts of sudden oak death
Sudden oak death can cause a variety of impacts to vegetation, ranging from leaf blight and stem dieback to complete mortality following the development of trunk cankers (Rizzo & Garbelotto 2003; Kliejunas 2011; Metz et al. 2017). Particularly high rates of mortality occur in tanoak, which is more susceptible to infection than most other species (Rizzo et al. 2005; McPherson et al. 2010; Cobb et al. 2012; Metz et al. 2012). Affected trees may remain relatively asymptomatic for several years following initial infection, then die quickly after onset of visible symptoms (Kliejunas 2011). Large trees are more susceptible to infection and have higher mortality rates (McPherson et al. 2010; Cobb et al. 2012; Metz et al. 2012; Haas et al. 2016). Beetle attacks in infected trees can also speed mortality, reducing life expectancy by 65–70% (McPherson et al. 2010). However, stands with high species diversity have lower infection risk due to reduced host density (Cobb et al. 2012; Haas et al. 2016; Gaydos et al. 2017).

Because the impacts of sudden oak death are species-specific, patterns of mortality can significantly alter forest structure and species composition (Cobb et al. 2012; Metz et al. 2012; Haas et al. 2016). Significant loss of large tanoak trees is likely (McPherson et al. 2010; Cobb et al. 2012), which may shift overstory composition towards species with lower susceptibility to infection, such as Douglas-fir (Cobb et al. 2010). California bay may become increasingly dominant within the understory (Haas et al. 2016), potentially increasing transmission and infection risk due to greater host density (Cobb et al. 2012; Metz et al. 2012).

High rates of tree mortality and resulting shifts in species composition are likely to change patterns of fuel composition and availability, potentially altering the frequency and severity of wildfire (Metz et al. 2011, 2017; Forrestel et al. 2015; Varner et al. 2017). However, the relationship between disease-related mortality, fuel dynamics, and fire severity is complex, and depends on multiple factors including the length of time between infection and fire (Metz et al. 2011). For example, fire severity is likely to increase in stands that burn shortly after infection

due to dead leaves and twigs that remain on standing trees (Metz et al. 2011). As time since infection increases, fallen trees provide a greater proportion of surface fuels and crown fires become less common (Metz et al. 2011). Changes in forest structure and composition also alter fuel dynamics on longer time scales due to changes in litter flammability (Varner et al. 2017). Where Douglas-fir overtops tanoak in the overstory, litter flammability may decrease (Varner et al. 2017). Fire severity is unlikely to change significantly where hardwoods become increasingly dominant, but areas where litter from multiple species accumulates may experience an increase in fire severity by up to 17% (Varner et al. 2017).

Although it is unlikely that tanoak will completely disappear from the forest due to its sprouting ability, the presence of the *P. ramorum* pathogen may prevent tanoak from reaching the sub-canopy (Cobb et al. 2012; Metz et al. 2012), reducing populations to sprouts that are repeatedly killed before they produce acorns (Bowcutt 2014). Thus, sudden oak death has the potential to contribute to functional extirpation of this species across large parts of its range (Dillon et al. 2013), potentially leading to novel ecosystem conditions as other species are unlikely to fulfill the role of tanoak within the region (Metz et al. 2012). For instance, the loss of tanoak acorns is likely to have cascading effects on ecosystem food webs by reducing habitat quality and food resources for many northwestern California tribes and wildlife (birds and small mammals; (Monahan & Koenig 2006; Fryer 2008; Bowcutt 2014). Ecosystem processes (e.g., nutrient cycling) could also be impacted by the loss of the extensive mycorrhizal networks associated with tanoak (Bergemann et al. 2013; Cobb et al. 2013). Changes in soil nitrogen availability and litterfall dynamics tied to tanoak mortality also increase the risk of impacts to ecosystem processes (Rizzo & Garbelotto 2003; Cobb et al. 2013).

Interactions between climate changes and sudden oak death
Changing climate conditions may alter patterns of spore production, disease transmission, susceptibility, and mortality in forests impacted by sudden oak death (Meentemeyer et al. 2004; Venette 2009; Kliejunas 2011; Sturrock et al. 2011). Models predicting *P. ramorum* dispersal and infection risk based on host availability and weather conditions suggest that coastal areas from Mendocino County through southwestern Oregon are at very high risk of infection over the next several decades (Meentemeyer et al. 2011). Warmer winter temperatures and increased winter and spring precipitation would likely enhance spore production and increase infection risk (Davidson et al. 2008; Kliejunas 2011; Meentemeyer et al. 2011; DiLeo et al. 2014; Haas et al. 2016), with favorable weather conditions potentially doubling the rate of spread by 2030 (Meentemeyer et al. 2011). Conversely, drier summer conditions could reduce disease prevalence by limiting growth of *P. ramorum* (Davidson et al. 2005; Venette & Cohen 2006), although the most significant increases in drought stress are projected to occur in areas already considered climatically unsuitable for the spread of sudden oak death (Venette 2009).

Regional Disease & Insects Trends	
Historical & current trends:	*Projected future trends:*
• Northwards expansion of sudden oak death	• Increased spore production and infection risk

Regional Disease & Insects Trends	
into Humboldt County since the discovery of the disease in the 1990s (Meentemeyer et al. 2011)	in the North Coast and North Coast Range regions, resulting in range expansion of sudden oak death northwards and slightly inland (Venette 2009; Kliejunas 2011) • Suitable weather conditions (e.g., warmer temperatures, increased winter and spring precipitation) could double the rate of spread by 2030 (Meentemeyer et al. 2011)

Summary of Potential Impacts on Habitat *(see text for citations)*

- Increased tree mortality, especially during periods of drought and/or in response to introduced pathogens such as *P. ramorum*
- Shifts in species composition and forest structure following the loss of tanoak in the overstory
- Changes in fuel composition and availability due to the presence of dead and dying trees, potentially altering fire behavior
- Reduced food resources and habitat quality for wildlife, potentially with impacts to the food web
- Changes in ecosystem processes due to the loss of ectomycorrhizae associated with tanoak
- Loss of cultural resources including large tanoak stands, tanoak mushrooms, and other culturally-valuable species susceptible to sudden oak death

Sensitivity and current exposure to non-climate stressors

Regional experts evaluated mixed evergreen forests as having moderate sensitivity to non-climate stressors (high confidence in evaluation), with an overall moderate-high current exposure to these stressors within the study region (high confidence). Key non-climate stressors that affect mixed evergreen forests include fire suppression, timber harvest, and roads/highways/trails.[8]

Fire suppression
Since the introduction of widespread fire suppression in northwestern California in the mid-1900s (Strothmann & Roy 1984), the frequency of fire in mixed evergreen forests has been significantly reduced (Taylor & Skinner 1998; Skinner et al. 2006; Stuart & Stephens 2006; Safford & Van de Water 2014; Steel et al. 2015; Taylor et al. 2016). The exclusion of fire from mixed evergreen forests following the historical removal of large trees during timber harvest has contributed to a general trend toward denser, even-aged forests dominated by a greater proportion of small trees, particularly given the historical practice of plantation development (Strothmann & Roy 1984; Skinner 1995; Hunter 1997; Colombaroli & Gavin 2010; Sensenig et al. 2013; Odion et al. 2014; McIntyre et al. 2015). Overall, this has resulted in more homogenous forests on a landscape scale due to reduced structural complexity in older forest stands and reduced areas of high-integrity early- and mid-successional forest patches (Skinner 1995; Spies et al. 2006; Sensenig et al. 2013; McIntyre et al. 2015; Norgaard et al. 2016; Spies et al. 2018a).

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[8] Non-climate stressors presented are those ranked as having a moderate or higher impact on this habitat type; additional non-climate stressors that may influence the habitat to a lesser degree include invasive species.

In some areas, shifts in species composition have occurred in the absence of fire (Hunter 1997; Norgaard et al. 2016) due to changes in regeneration dynamics and patterns of forest succession (Hunter 1997; Sensenig et al. 2013; Spies et al. 2018b). For instance, continued establishment of shade-tolerant trees and shrubs in the forest understory has increased stem density (Skinner 1995; Hunter 1997; Stuart & Stephens 2006; Sensenig et al. 2013) and understory continuity (Sensenig et al. 2013). However, species-specific differences in shade tolerance, longevity, and other traits can impact observed changes in forest structure and composition (Hunter 1997). For instance, the longevity of Douglas-fir has allowed it to maintain canopy dominance even when regeneration rates are very low, while Pacific madrone has declined more dramatically (Hunter 1997). Over longer time scales, competitive exclusion in the absence of disturbance can lead to decreases in stem density and a trend toward increasing tree sizes in some areas (Hunter 1997; Sensenig et al. 2013).

Fire suppression activities (e.g., backburning, fire line construction, safety mitigation activities) can also damage forest resources directly, including mature trees with high cultural value (Lake 2007; Norgaard et al. 2016). For instance, practices such as backburning and the construction of fire lines can damage tanoak stands that may not have otherwise burned, injuring or killing large trees and disturbing soils and the myecelium net created by mycorrhizal relationships among fungi and plants (Lake 2007; Norgaard et al. 2016).

Currently, there is insufficient data to conclude whether fire suppression has resulted in an increased proportion of high-severity fire within mixed evergreen forests. The mixed-severity fire regime typical of this forest type can result in large patches that burn at high severity (Taylor & Skinner 2003; Halofsky et al. 2011; Odion et al. 2014). Longer fire return intervals in relatively productive forests also do not increase fuel accumulation as much compared to drier forest types (Agee 1996; Odion et al. 2004).

Timber harvest

The legacy of large-scale timber harvest in the late 19th and early 20th centuries has impacted most mixed evergreen forest areas in northwestern California (Strittholt et al. 2006; DellaSala et al. 2015), resulting in the fragmentation and loss of mature and old-growth forests across their range (Strittholt & DellaSala 2001; Staus et al. 2002; Strittholt et al. 2006). Following the end of World War II, the US Forest Service embraced a strategy intended to convert older stands of mixed evergreen forest on federal lands to plantations (Strothmann & Roy 1984). This strategy eventually resulted in the development of the Northwest Forest Plan to assure the long-term presence of mature and old-growth mixed evergreen forests and associated wildlife species (Spies et al. 2018b). By the mid-1990s, most clear-cutting on federal lands within the NWFP area had ceased, although continued demand for timber resulted in concurrent increases in clearcutting on private industrial timberlands (Phalan et al. 2019). Contemporary large-scale timber harvest (including large-tree logging and clear-cutting) are primarily limited to privately-owned lands, while federal forests are primarily impacted by the legacy of past overharvest (Phalan et al. 2019). Silvicultural treatments designed to enhance ecosystem recovery, such as

variable-density thinning, continue to be used on federal lands with the goal of accelerating the development of complex forest structure in young plantations (Phalan et al. 2019).

Overall, historical overharvest of timber resources in mixed evergreen forests has resulted in increased homogeneity on a landscape scale (Bradley et al. 2016; Zald & Dunn 2018) due to the loss of large trees, higher densities of small trees, and shifts towards a greater proportion of shade-tolerant species (Bingham & Sawyer 1991; Sensenig et al. 2013; McIntyre et al. 2015). Significant shifts in forest structure and composition have occurred in logged stands across the region, which have been exacerbated by many decades of fire suppression (Bingham & Sawyer 1991; Sensenig et al. 2013; McIntyre et al. 2015). Forests where large trees have been logged undergo simplified patterns of succession compared to stands that develop following a natural disturbance (e.g., mixed-severity fire) in forests with a preexisting complex structure (Swanson et al. 2011; Donato et al. 2012; DellaSala et al. 2014). The loss of forest canopy in heavily-logged and clear-cut stands can also alter the moisture balance within mixed evergreen forests, lowering relative humidity and increasing the risk of uncharacteristically large proportions of severe fire (Odion et al. 2014; Zald & Dunn 2018). By contrast, studies have found lower fire severity in dry forests protected from logging (Bradley et al. 2016). Logging and associated road networks also increase soil erosion in mixed evergreen forests, particularly in burned areas following heavy rainfall (Colombaroli & Gavin 2010).

Roads, highways, and trails
Roads, highways, and trails increase habitat fragmentation within mixed evergreen forests, dissecting large forest patches and inhibiting wildlife movement across the landscape (Gucinski et al. 2001; Coffin 2007). In northern California mixed evergreen forests, road construction associated with logging can further impact forests by significantly increasing soil erosion (Colombaroli & Gavin 2010). Roads can also spread invasive weeds that thrive on disturbed roadsides (Gucinski et al. 2001), potentially altering fire behavior by increasing fuel availability and continuity (Brooks et al. 2004). Human ignitions are also significantly correlated with proximity to roads, suggesting that the presence of roads may further contribute to altered fire regimes (Syphard & Keeley 2015; Vuln. Assessment Reviewer, pers. comm., 2018).

Transportation networks within mixed evergreen forest areas can contribute to the spread of diseases such as sudden oak death and Port-Orford-cedar root rot due to the movement of contaminated soil from infected sites (Hansen et al. 2000; Jules et al. 2002; Davidson et al. 2005; Swiecki et al. 2017). For example, one study that surveyed hikers on a 2.4 km (1.5 mi) trail in a Sonoma County mixed evergreen forest found that up to half of the hikers in a given year were carrying infested soil on their shoes (Davidson et al. 2005).

Sensitivity to other critical factors
Regional experts evaluated mixed evergreen forests as having high sensitivity to changes in mycorrhizal function (moderate confidence in evaluation).

Mycorrhizal function

Mycorrhizae mediate mutualistic relationships between fungi and plants, providing nutrients and water from the soil to plants in exchange for carbon and simple sugars (Smith & Read 2010). Among plant species with overlapping ranges, shared mycorrhizal communities can link individuals to one another, potentially facilitating the distribution of carbon, nutrients, and water (Amaranthus & Perry 1989, 1994; Perry et al. 1989). In this way, mycorrhizae can reduce competition between plant species (Perry et al. 1989) and increase resistance to stressors such as drought (Perry et al. 1990). Mycorrhizae associated with early-successional hardwoods and shrubs, including whiteleaf manzanita (*Arctostaphylos viscida*), Pacific madrone, tanoak, and canyon live oak, can also significantly enhance conifer establishment following disturbances such as fire and clear-cutting (Amaranthus & Perry 1989; Borchers & Perry 1990; Perry et al. 2011; Kennedy et al. 2012). For instance, Douglas-fir seedlings in southwest Oregon had higher rates of survival and growth when planted on a cleared site previously occupied by whiteleaf manzanita, and growth was enhanced even further following inoculation with unpasteurized soil from nearby Pacific madrone stands (Amaranthus & Perry 1989). Conversely, a lack of functioning mycorrhizal relationships can prevent tree establishment on disturbed sites, particularly when there is a temporal gap between plant communities such as occurs following clear-cutting (Perry et al. 1990). Legacy trees can act as refugia for mycorrhizal communities, allowing recolonization of newly established seedlings (Luoma et al. 2006).

Over the coming century, mycorrhizal community composition and biomass are likely to shift in response to changes in climate conditions, disturbance regimes, and plant species composition (Perry et al. 1990; Drigo et al. 2008; Vuln. Assessment Reviewer, pers. comm., 2018). In turn, changes in mycorrhizal relationships may alter fungal and soil microbial communities that regulate various ecosystem processes such as nutrient cycling (Drigo et al. 2008). These shifts could facilitate adaptation of existing plant species to changes in soil and warmer, drier climate conditions (Perry et al. 1990; Vuln. Assessment Reviewer, pers. comm., 2018). However, it is also possible that asynchronous shifts in plant species distribution and mycorrhizal community composition limit successful establishment of species within novel assemblages (Perry et al. 1990). Additionally, climate-driven changes in fire regimes and patterns of disease (e.g., sudden oak death) could result in direct or indirect (i.e., through the loss of host species) impacts to mycorrhizal communities, causing further disruption to the forest ecosystem (Bergemann et al. 2013; Norgaard et al. 2016).

Adaptive Capacity

Mixed evergreen forests were evaluated by regional experts as having moderate overall adaptive capacity (high confidence in evaluation).

Habitat extent, integrity, continuity, and permeability

Regional experts evaluated mixed evergreen forests as having a high geographic extent (high confidence in evaluation), moderate structural and functional integrity (moderate confidence), and moderate-high continuity (moderate confidence). Landscape permeability for mixed

evergreen forests was evaluated as low-moderate (moderate confidence). Timber harvest, topographic features, and roads/highways/trails were identified as the primary barriers to habitat continuity and dispersal across the study region.[9]

Mixed evergreen forests in northern California are extensive at low to middle elevations across the North Coast Range and western Klamath Mountains (Sawyer et al. 1977; Skinner et al. 2006; Stuart & Stephens 2006; Sawyer 2007; Fryer 2008; North et al. 2016). Historically, this region contained very large areas of mature and old-growth forests, but many of these have been lost over the past century to logging and secondary changes in forest structure (Strittholt et al. 2006; Olson et al. 2012). The Klamath-Siskiyou region of northern California and southwestern Oregon experienced fewer losses of mature and old-growth conifer forest compared to most other regions of the Pacific Northwest due to the rugged terrain (Strittholt et al. 2006). Due to increases in large wildfires, the area of mature and old-growth forest on federal lands has declined slightly over the past several decades despite the implementation of the Northwest Forest Plan (Phalan et al. 2019). However, the cessation of large-tree logging and clear-cutting on federal lands protected by the Northwest Forest Plan has slowed the rate of loss significantly compared to private industrial timberlands, and burned areas are creating more areas of diverse early-seral habitat on federal lands (Phalan et al. 2019).

Logging and the secondary impacts of timber management have been one of the primary causes of habitat fragmentation in mature and old-growth mixed evergreen forests (Spies et al. 2006, 2018a; Strittholt et al. 2006; Olson et al. 2012). Regenerating, fire-suppressed forests in logged areas are generally denser and more homogenous in structure and composition compared to forests that have not experienced overharvest (Swanson et al. 2011; DellaSala et al. 2014). This results in reduced habitat connectivity for wildlife dependent on old-growth forest characteristics and increased vulnerability to large-scale disturbances that could cause further fragmentation (Odion et al. 2004; Spies et al. 2018b). Roads can also fragment mixed evergreen forests, while topographic geologic features (e.g., mountain ranges, varying soil types) can result in patchiness in some areas (Vuln. Assessment Workshop, pers. comm., 2017).

Habitat diversity

Regional experts evaluated mixed evergreen forests as having moderate-high physical and topographical diversity (high confidence in evaluation), moderate-high component species diversity (high confidence), and moderate-high functional diversity (high confidence).

Mixed evergreen forests are typically dominated by Douglas-fir and one or more hardwoods, of which tanoak is the most abundant (Whittaker 1960; Franklin & Dyrness 1973; Sawyer et al. 1977; Bingham & Sawyer 1991). Species composition and stand structure in this habitat type can vary considerably over short distances depending on soil moisture, topography (e.g., slope, aspect), substrate, and disturbance history, among other factors (Whittaker 1960; Sawyer et al.

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[9] Barriers presented are those ranked as having a moderate or higher impact on this habitat type; additional barriers that may limit habitat continuity and dispersal to a lesser degree include land-use conversion and invasive species.

1977; Bingham & Sawyer 1991; Spies et al. 2006; Sawyer 2007). For instance, a rain shadow created by the North Coast Range results in different species assemblages on the eastern and western slopes of the range (Skinner et al. 2009; Vuln. Assessment Workshop, pers. comm., 2018). The resulting mosaic of forest patches and canopy openings creates high forest heterogeneity across the region (Sawyer et al. 1977; Bingham & Sawyer 1991; Spies et al. 2006). This heterogeneity is further maintained by a mixed-severity fire regime that creates variably-sized patches of vegetation in a range of successional stages from old-growth stands to early-seral communities (Agee 1996; Skinner et al. 2006; Stuart & Stephens 2006; Perry et al. 2011; Hessburg et al. 2016).

Stands with old-growth characteristics are more common within mixed evergreen forests compared to many other northern California forest types (Spies et al. 2006), and are characterized by large-diameter trees, a wide range of tree ages, a greater density of large snags and downed wood, and a complex vertical structure with shrubs, hardwoods, and conifers creating a multi-layered canopy (Old Growth Definition Task Group 1986; Bingham & Sawyer 1991; Spies et al. 2006; Sensenig et al. 2013). These habitat elements are critical for wildlife species such as Pacific fisher and northern spotted owl (Raphael 1991; Zielinski et al. 2004; Spies et al. 2006; Norgaard et al. 2016).

The high physical and topographic diversity in northwestern California is created by varied topography, multiple intersecting mountain ranges, steep temperature and precipitation gradients, and a variety of soil types, including serpentine soils (Sawyer 2007; North et al. 2016). These factors result in high forest structural and component species diversity, and forests support many endemic species and unique vegetation communities (DellaSala et al. 1999; Sawyer 2007; North et al. 2016). The region was an important refugium during past changes in climate and is likely to remain so (Olson et al. 2012; Schierenbeck 2017), potentially acting as a reservoir for genetic material that may allow shifts in species composition and/or genetic adaptation to future climate conditions (Schierenbeck 2017). However, reductions in species and structural diversity due to the legacy of past management practices (e.g., logging, road construction, fire suppression) reduce the potential for dynamic changes in habitat composition and structure in response to warmer, drier conditions (Skinner et al. 2006; Spies et al. 2006; Stewart et al. 2016).

Resistance and recovery

Regional experts evaluated mixed evergreen forests as having moderate resistance to climate stressors and natural disturbance regimes (moderate confidence in evaluation). Recovery potential was evaluated as moderate-high (high confidence).

Resistance to climate stressors and altered disturbance regimes in mixed evergreen forests is supported by high landscape heterogeneity, which provides varied microsites and areas of refugia from thermal stress, increasing climatic water deficit, and wildfire (Olson et al. 2012; Frey et al. 2016; Tepley et al. 2017). Where structural diversity remains relatively high, the impacts of disturbances are limited within a mosaic of forest patches at various successional

stages and recovery is relatively rapid (Skinner et al. 2006; Sawyer 2007; North et al. 2016). The combined impacts of warmer, drier conditions and increased frequency of high-severity fires are likely to alter successional patterns, potentially favoring the establishment of shrubland- and hardwood-dominated habitats with fewer conifers (Lenihan et al. 2008; Westerling et al. 2011; Welch et al. 2016; Tepley et al. 2017). However, many component species within mixed evergreen forests are characterized by tolerance to multiple stressors (e.g., shade, drought), and several of these are projected to see increased recruitment under future climate conditions, suggesting greater resilience within this forest type compared to many others (Dobrowski et al. 2015).

Over the past two millennia, mixed evergreen forests within the Klamath-Siskiyou region recovered from many periods of multi-decadal drought and severe wildfires (Whitlock et al. 2008; Briles et al. 2008, 2011; Colombaroli & Gavin 2010), and the forest is adapted to a wide range of fire frequencies and severities (Donato et al. 2009b; Skinner et al. 2009; Trouet et al. 2010; Colombaroli & Gavin 2010). However, a combination of historical logging and fire suppression have reduced the ability of the forest to recover from future stressors and large-scale disturbances (Spies et al. 2006; Millar & Stephenson 2015; Tepley et al. 2017). Climate changes are likely to result in further negative impacts to tribal resources (Long et al. 2016; Norgaard et al. 2016) and the persistence of sensitive species with limited distributions (e.g., northern spotted owl; California Department of Fish and Wildlife 2016).

Management potential

Public and societal value
Regional experts evaluated mixed evergreen forests as having moderate public and societal value (high confidence in evaluation).

The general public often does not appreciate the many ecosystem benefits provided by mixed evergreen forests, which includes commercial and non-commercial forest products, clean water, carbon storage and hydrologic functions, wildlife habitat and connectivity, and biodiversity, among others (Vuln. Assessment Workshop, pers. comm., 2017). Societal support for the management of mixed evergreen forests varies widely depending on the location and spatial scale (local to national) being considered (Vuln. Assessment Workshop, pers. comm., 2017). This is, in part, because management within this forest type is complex (Vuln. Assessment Workshop, pers. comm., 2017), necessitating the consideration of multiple resource objectives and interacting climate and non-climate stressors such as moisture stress, fire frequency and severity, the legacy of past management practices, and the continuing spread of sudden oak death (Spies et al. 2006; Norgaard et al. 2016; Hessburg et al. 2016; Metz et al. 2017). Additionally, there is frequently a lack of agreement on whether the most effective strategies involve more extensive management across the landscape or comprehensive protections that limit or prohibit activities such as thinning (Vuln. Assessment Workshop, pers. comm., 2017; Spies et al. 2018b).

The Northwest Forest Plan (NWFP), adopted in 1994, established a network of conservation reserves across a 10-million-hectare area spanning northwestern California, Oregon, and Washington that corresponds to the range of the threatened northern spotted owl (DellaSala et al. 2015). The intent of the NWFP is to assure that forest management in this region is directed toward maintaining or re-creating old-growth forest conditions (Franklin et al. 2002; Franklin & Johnson 2012; Spies et al. 2018a). As a result, the NWFP has helped to increase the level of protection and improve management strategies in mixed evergreen forests by shifting the focus from timber management to biodiversity conservation (DellaSala et al. 2015). Since its implementation, the NWFP has been successful in reducing losses of late-successional forest in the region to logging by 80% (Strittholt et al. 2006). However, the degree of protection provided by the NWFP and other regulations varies across the landscape (DellaSala et al. 1999; Strittholt et al. 2006). Additionally, some tribal members have noted that the NWFP protections limit the restoration of ecocultural conditions historically maintained by tribes (Long et al. 2018). Protected areas also remain vulnerable to uncharacteristically severe and/or large-scale disturbances that are likely to be exacerbated by climate change (e.g., wildfire (Spies et al. 2018a; Phalan et al. 2019).

Mixed evergreen forests are of critical importance to northern California tribes, who have carefully managed tanoak stands and associated cultural resources for centuries (Anderson 2005; Bowcutt 2013; Norgaard et al. 2016; Karuk Tribe 2019). Many of these plant and wildlife species have been significantly impacted by land use changes, human activity (e.g., over-harvesting of mushrooms), non-tribal management practices (e.g., fire suppression, logging, road building), and restrictions on forest access and resource gathering (Anderson & Lake 2013; Lake 2013; Voggesser et al. 2013; Norgaard 2014a, 2014b; Norgaard et al. 2016; Long & Lake 2018). Climate change is likely to further impact the distribution and abundance of cultural resources found within mixed evergreen forests (Voggesser et al. 2013). In addition to directly impacting culturally-important species, changes in climate and climate-driven factors and altered disturbance regimes may affect tribal access to forests and valued wildlife and plant species, as well as the ability to carry on traditional practices, including cultural burning (Norgaard 2014a; Whyte 2014; Norgaard et al. 2016). For instance, the spread of sudden oak death has resulted in the loss of over a million tanoak trees, including stands managed by northern California tribes for hundreds of years (Ortiz 2008a; Bowcutt 2014), and many cultural gathering activities have been restricted for fear of spreading the disease further (Voggesser et al. 2013).

Management capacity and ability to alleviate impacts[10]
Regional experts evaluated the potential for reducing climate impacts on mixed evergreen forests through management as low-moderate (moderate confidence in evaluation).

Changing climate conditions are likely to make the management of mixed evergreen forests more complex due to the increased potential for forests under stress to experience drastic

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[10] Further information on climate adaptation strategies and actions for northern California can be found on the project page (https://bit.ly/31AUGs5).

shifts in species composition and/or large-scale dieback (McIntyre et al. 2015; Tepley et al. 2017; Young et al. 2017). Because forests in this region are dynamic systems subject to frequent disturbances, management strategies will need to be highly responsive to site conditions, including vegetation composition, topography, and disturbance history (Halofsky et al. 2014). The scientific literature identifies several management approaches that may reduce the impact of climate stressors and climate-mediated changes in disturbance regimes; these are primarily focused on the promotion of spatial heterogeneity and structural complexity (Jewell & Vilsack 2014; Hessburg et al. 2016; Metlen et al. 2017) and the protection of existing forests from anthropogenic stressors (Strittholt et al. 2006; Olson et al. 2012; DellaSala et al. 2015).

Protecting existing late-successional forests and increasing landscape connectivity is vital, as these areas may be more resistant to changing climate conditions and could provide refugia for sensitive species (Strittholt et al. 2006; Olson et al. 2012; DellaSala et al. 2015). Protection efforts should include mid-seral and complex early-seral habitats that have high structural diversity and the potential to develop old-growth characteristics over time (Strittholt et al. 2006; Franklin & Johnson 2012; DellaSala et al. 2015). Maintaining a matrix of forest areas in multiple stages of succession and at varying elevations and slope aspects is important to promote landscape heterogeneity and high biodiversity (Hessburg et al. 2016; Spies et al. 2018b). Careful attention will be required in the future, however, to assure that landscape connectivity is maintained or enhanced with appropriate consideration for the functional needs of the species involved (Taylor et al. 2006).

In mixed evergreen forests that have been degraded and/or are threatened by climatic and anthropogenic stressors, management strategies often include reducing stem density (Latham & Tappeiner 2002; Dagley et al. 2017), reintroducing fire onto the landscape (Lake 2007; Bowcutt 2013; Jewell & Vilsack 2014; Halpern 2016; Long et al. 2016; Norgaard et al. 2016; Hessburg et al. 2016), and supporting forest regeneration (Welch et al. 2016). These strategies may enhance forest resilience by increasing spatial heterogeneity and structural complexity (Spies et al. 2006; Jewell & Vilsack 2014; Hessburg et al. 2016; Metlen et al. 2017; Young et al. 2017), which reduces vulnerability to large-scale disturbances such as uncharacteristically severe wildfire, insects, and disease (Odion et al. 2004; Norgaard et al. 2016; Hessburg et al. 2016). For instance, the use of ecologically-based legacy retention, prescribed fire, and thinning from below can increase tree vigor and encourage the development of complex mature forests with larger, more fire-resistant trees (Franklin et al. 2002; Latham & Tappeiner 2002; Youngblood et al. 2006; Franklin & Johnson 2012; Halofsky et al. 2014; Metlen et al. 2017; Spies et al. 2018a).

The U.S. Forest Service is currently involved in collaborative efforts with tribes to promote fire-resilient systems and reintroduce fire onto the landscape (Jewell & Vilsack 2014; Metlen et al. 2017; USDA Forest Service 2018). For instance, the Western Klamath Restoration Partnership has collaborated with the U.S. Forest Service on the Somes Bar Integrated Fire Management Project to implement a range of stewardship treatments within Six Rivers National Forest in accordance with traditional ecological knowledge (TEK) and tribal customs (USDA Forest Service 2018). Examples of management actions included in this project are prescribed burning and

thinning in dense areas to reduce ladder fuels and promote the development of larger, fire-resistant trees with high ecological and cultural value (USDA Forest Service 2018).

Managing mixed evergreen forests to mitigate the threat of introduced diseases such as sudden oak death and Port-Orford-cedar root rot is difficult at large scales (Hansen et al. 2000; Alexander & Lee 2010; Bowcutt 2013; Jules et al. 2015; Swiecki et al. 2017). Currently, slowing the spread of disease by preventing the movement of contaminated soil is a significant focus (Swiecki et al. 2017). For instance, management activities designed to reduce the rate of spread along roads and trails could include installing signage, closing roads and trails (especially during the wet season), posting preferred travel direction on trails, altering road surface materials, and providing additional instruction to road crews (Hansen et al. 2000; Bowcutt 2013; Jules et al. 2015; Swiecki et al. 2017).

Many communities, local governments, and regional/state-wide collaborations have also set up monitoring networks focused on early detection of sudden oak death in order to proactively manage tanoak infection (Alexander & Lee 2010). Tribal involvement in efforts to slow the spread of sudden oak death has helped to communicate the importance of this threat to local landowners, who may not place high value on tanoak (Alexander & Lee 2010). Other efforts have involved surveying the area to determine whether naturally-occurring resistance to sudden oak death can be leveraged to aid conservation benefits (Cobb et al. 2018). The protection of tanoak refuges in areas where environmental conditions are less conducive to the disease may allow the preservation of uninfected trees (Bueno et al. 2010; Dillon et al. 2013; Vuln. Assessment Reviewer, pers. comm., 2018), particularly those within cultural landscapes (Bowcutt 2013). Seed banks and living collections may also provide a way to preserve genetic diversity and allow for the future reintroduction of tanoak into infected areas (Bowcutt 2014).

Ecosystem services
Mixed evergreen forests play a particularly critical role in carbon storage and maintaining hydrological function across the landscape due to their high productivity (Law & Waring 2015; Ellison et al. 2017). Carbon sequestration is widely recognized as an important goal in forest management, and changes in forest management may promote carbon accumulation and storage in productive forests (Law & Waring 2015; Ellison et al. 2017; Law et al. 2018). Mixed evergreen forests also provide natural water collection, filtration, and delivery into streams, wet meadows, and aquifers and enhance flood control by absorbing vast amounts of water during major rain events and then releasing it slowly (Ellison et al. 2017). Other critical ecosystem services provided by mixed evergreen forests include:
- Provisioning of food, fiber, fuel, genetic resources, natural medicines, and ornamental resources;
- Regulation of air quality, climate/microenvironments (e.g., shade), pest/disease regulation, pollination, predator-prey dynamics, food web functions, aquatic ecosystem dynamics, and natural hazard reduction;
- Support of primary production, oxygen production, soil formation/retention, nutrient cycling, water cycling, and biodiversity;

- Cultural/tribal uses for spiritual/religious purposes, knowledge systems, educational values, sense of place, cultural heritage, inspiration, and recreation (Vuln. Assessment Workshop, pers. comm., 2017).

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Northern California Climate Adaptation Project:

Vulnerability Assessment Methods and Application

Defining Terms

Exposure: A measure of how much of a change in climate or climate-driven factors a resource is likely to experience (Glick et al. 2011).

Sensitivity: A measure of whether and how a resource is likely to be affected by a given change in climate or factors driven by climate (Glick et al. 2011).

Adaptive Capacity: The ability of a resource to accommodate or cope with climate change impacts with minimal disruption (Glick et al. 2011).

Vulnerability: A function of the sensitivity of a particular resource to climate changes, its exposure to those changes, and its capacity to adapt to those changes (IPCC 2007).

Vulnerability Assessment Model

The vulnerability assessment model applied in this process was developed by EcoAdapt (EcoAdapt 2014a; EcoAdapt 2014b; Kershner 2014; Hutto et al. 2015; Gregg 2018),[11] and includes evaluations of relative vulnerability by local and regional stakeholders who have detailed knowledge about and/or expertise in the ecology, management, and threats to focal habitats, species groups, and species, and the ecosystem services these resources provide. Stakeholders evaluated vulnerability for each resource by discussing and answering a series of questions for sensitivity and adaptive capacity. Exposure was evaluated by EcoAdapt using projected future climate changes from the scientific literature. Each vulnerability component (i.e., sensitivity, adaptive capacity, and exposure) was divided into specific elements. For example, habitats included three elements for assessing sensitivity and six elements for adaptive capacity. Elements for each vulnerability component are described in more detail below.

In-person workshops were held in Eureka, Redding, and Upper Lake between May and October 2017. Participants self-selected habitat and species group/species breakout groups and evaluated each resource's vulnerability. Participants were first asked to describe the habitat and/or list the species considered as part of the evaluation of an overarching species group. Due to limitations in workshop time and participant expertise, multiple resources were not assessed during these engagements. Evaluations for remaining habitats, species groups, and species were completed by contacting resource experts.[12]

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[11] Sensitivity and adaptive capacity elements were informed by Lawler 2010, Glick et al. 2011, and Manomet Center for Conservation Sciences 2012.

[12] Resources evaluated by experts included: coastal bluff/scrub habitats, coastal conifer hardwood forest, true fir forest, lakes/ponds, freshwater marshes, vernal pools, seeps/springs, native insect pollinators, native ungulates, salamanders, frogs, native mussels, marbled murrelet, and northwestern pond turtle.

Stakeholders assigned one of five rankings (High, Moderate-High, Moderate, Low-Moderate, or Low) for sensitivity and adaptive capacity; EcoAdapt assigned rankings for projected future climate exposure. Rankings for each component were then converted into scores (High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1) and the scores averaged (mean) to generate an overall score. For example, scores for each element of habitat sensitivity were averaged to generate an overall habitat sensitivity score. Scores for exposure were weighted less than scores for sensitivity and adaptive capacity; this was due to greater uncertainty about the magnitude and rate of future change. Sensitivity, adaptive capacity, and exposure scores were combined into an overall vulnerability score calculated as:

Vulnerability = [(Climate Exposure*0.5) x Sensitivity] - Adaptive Capacity

Elements for each component of vulnerability were also assigned one of three confidence rankings (High, Moderate, or Low). Confidence rankings were converted into scores (High-3, Moderate-2, or Low-1) and the scores averaged (mean) to generate an overall confidence score. These approximate confidence levels were based on the Manomet Center for Conservation Sciences (2012) 3-category scale, which collapsed the 5-category scale developed by Moss and Schneider (2000) for the IPCC Third Assessment Report. The vulnerability assessment model applied here assesses the confidence associated with individual element rankings, and also uses these rankings to estimate the overall level of confidence for each component of vulnerability as well as overall vulnerability.

Rankings and scores presented should be considered measures of relative vulnerability and confidence (i.e., comparing the level of vulnerability between the focal resources evaluated in this project).

Vulnerability Assessment Model Elements

Sensitivity & Exposure (Applies to Habitats, Species Groups, Species)

- **Climate and Climate-Driven Factors:** e.g., air temperature, precipitation, freshwater temperature, soil moisture, snowpack, extreme events: drought, altered streamflows, etc.
- **Disturbance Regimes:** e.g., wildfire, flooding, drought, insect and disease outbreaks, wind
- **Future Climate Exposure:** e.g., consideration of projected future climate changes (e.g., temperature and precipitation) as well as climate-driven changes (e.g., altered fire regimes, altered flow regimes, shifts in vegetation types)
- **Non-Climate Stressors:** e.g., residential or commercial development; agriculture and/or aquaculture; roads, highways, trails; dams and water diversions; invasive and other problematic species; livestock grazing; fire suppression; timber harvest; etc.

Sensitivity & Exposure (Applies to Species Groups and Species)

- **Dependencies:** e.g., dependencies on sensitive habitats, specific prey or forage species

Sensitivity & Exposure (Applies to Species ONLY)

- **Life History:** e.g., species reproductive strategy, average length of time to reproductive maturity

Adaptive Capacity (Applies to Habitats, Species Groups, Species)

- **Extent, Integrity, and Continuity/Connectivity:** e.g., resources that are widespread vs. limited, structural and functional integrity (e.g., degraded or pristine) of a habitat or health and functional integrity of species (e.g., endangered), isolated vs. continuous populations
- **Landscape Permeability:** e.g., barriers to dispersal and/or continuity (e.g., land-use conversion, energy production, roads, timber harvest, etc.)
- **Resistance and Recovery:** e.g., *resistance* refers to the stasis of a resource in the face of change, *recovery* refers to the ability to "bounce back" more quickly from stressors once they do occur
- **Management Potential:** e.g., ability to alter the adaptive capacity and resilience of a resource to climatic and non-climate stressors (societal value, ability to alleviate impacts, capacity to cope with impacts)
- **Ecosystem Services:** e.g., provisioning, regulating, supporting, and/or cultural services provided by the resource

Adaptive Capacity (Applies to Habitats ONLY)

- **Habitat Diversity:** e.g., diversity of physical/topographical characteristics, component native species and functional groups

Adaptive Capacity (Applies to Species Groups, Species)

- **Dispersal Ability**: i.e., ability of a species to shift across the landscape as the climate changes
- **Intraspecific/Life History Diversity:** e.g., life history diversity, genetic diversity, phenotypic and behavioral plasticity

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