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Meeting the challenges of climate change

Mixed Conifer and Ponderosa Forests
Northern California Climate Change Vulnerability Assessment Synthesis

An Important Note About this Document: *This document represents an initial evaluation of vulnerability for mixed conifer and ponderosa 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 scientific 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 Gabrielle Bohlman (U.S. Forest Service), Laura Brodhead (Bureau of Land Management), Frank Lake (U.S. Forest Service), and Chad Roberts (Tuleyome). Vulnerability scores were provided by Redding workshop participants. Upper Lake workshop participants provided additional comments on the climate change vulnerability of this habitat.

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Habitat Description

Mixed conifer and ponderosa forests are characterized by a combination of conifer species, including ponderosa pine (*Pinus ponderosa*), Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), sugar pine (*P. lambertiana*), Jeffrey pine (*P. jeffreyi*), and incense cedar (*Calocedrus decurrens*) as dominant or co-dominant species (Sawyer 2007; Evans et al. 2011; CNPS 2019). These forest types are extensive in northern California, covering many low- to mid-elevation montane areas between 300 and 1,800 m (1,000–6,000 ft) in elevation (North et al.

2016; CNPS 2019). In the southern Cascades region, more open forests dominated by ponderosa pine are particularly common (Skinner & Taylor 2006; Safford & Stevens 2017). Within the Klamath Mountains and North Coast region, Douglas-fir is a characteristic dominant species, often with a more diverse mix of conifers (Taylor & Skinner 2003; Skinner et al. 2006; Sawyer 2007; Spies et al. 2018). Throughout the region, black oak (*Quercus kelloggii*) is typically the primary hardwood species, and can dominate stands maintained by frequent fire (Long et al. 2016; North et al. 2016). The lower canopy of mixed conifer forests often contains many other hardwoods as well, including canyon live oak (*Q. chrysolepis*), giant chinquapin (*Chrysolepis chrysophylla*), Pacific madrone (*Arbutus menziesii*), bigleaf maple (*Acer macrophyllum*), and Oregon white oak (*Q. garryana*), among others (Skinner et al. 2006; Sawyer 2007; Norgaard et al. 2016; CNPS 2019). Within the understory, mixed conifer and ponderosa forests support shrubs, ferns, grasses, forbs, mushrooms, and lichens (Norgaard et al. 2016; North et al. 2016). Serpentine outcrops are typically unforested, or may contain shrubby vegetation overtopped by open stands of Jeffrey pine and foothill pine (*P. sabiniana*, also known as gray pine; Skinner et al. 2006).

Species composition and forest structure can change dramatically within short distances, depending on disturbance history, parent geology and soil type, topography (e.g., elevation, slope, aspect), temperature, and precipitation, among other factors (Taylor & Skinner 2003; Skinner et al. 2006; Sawyer 2007; North et al. 2016; Safford & Stevens 2017). Both dry and moist forest types occur within this habitat category, with species composition and disturbance regimes generally based on site characteristics that affect water balance (Evans et al. 2011; North et al. 2016; Safford & Stevens 2017). Moist mixed conifer forests are typically found in areas where annual precipitation is relatively high (generally over 100 cm [39.4 in]), and at higher elevations (Safford & Stevens 2017). Old-growth characteristics can be found in mature and late-seral stands across the region (Old Growth Definition Task Group 1986; Leonzo & Keyes 2010; Spies et al. 2018), and can arise through a variety of developmental sequences that are primarily determined by disturbance timing, extent, and severity (Spies et al. 2006, 2018; Reilly & Spies 2015).

Mixed conifer and ponderosa forests support hundreds of wildlife species, including many that depend on old-growth habitat characteristics (e.g., large cavities for denning/nesting), such as the Pacific fisher (*Pekania pennant pacifica*), northern spotted owl (*Strix occidentalis caurina)*, and northern Goshawk (*Accipiter gentilis*; Spies et al. 2006, 2018; Keane 2008; Norgaard et al. 2016; North et al. 2016). Many plant and wildlife species within mixed conifer and ponderosa forests also hold cultural value for northern California tribes, including black oak, sugar pine, giant chinquapin, California hazel (*Corylus cornuta californica*), evergreen huckleberry (*Vaccinium ovatum*), gooseberry (*Ribes californicum*), beargrass (*Xerophyllum tenax*), Pacific fisher, and black-tailed deer (*Odocoileus hemionus columbianus*), among others (Lake 2007; Norgaard et al. 2016; Karuk Tribe 2019). Northern California tribes depend on cultural burning practices to increase the productivity and predictability of valued resources by maintaining a series of resource patches in various successional stages (Anderson 2005; Lake 2013; Voggesser et al. 2013; Long et al. 2016; Norgaard et al. 2016; Karuk Tribe 2019).

Executive Summary

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

Mixed Conifer and Ponderosa Forests	Rank	Confidence
Sensitivity	Moderate-High	High
Future Exposure	Moderate	Moderate
Adaptive Capacity	Moderate-High	High
Vulnerability	**Moderate**	**High**

Sensitivity & Exposure Summary	*Climate and climate-driven factors*: • Climatic water deficit, soil moisture, air temperature, precipitation amount and timing, earlier timing of snowmelt and runoff, drought *Disturbance regimes*: • Wildfire, insects *Non-climate stressors*: • Fire suppression, human infrastructure (residential and commercial development, roads/highways/trails)

Mixed conifer and ponderosa forests are primarily vulnerable to climate stressors that increase water demand (e.g., warmer air temperatures) or decrease water availability (e.g., changes in precipitation amount and timing, earlier timing of snowmelt/runoff, reduced soil moisture, drought). Enhanced moisture stress also intensifies competition for resources and increases wildfire potential by reducing fuel moisture. This results in shifts in species composition and forest structure, increases in tree mortality, and reduced or delayed forest recovery following disturbances. Fire is a primary driver of forest succession within mixed conifer and ponderosa forests, and ecosystem fire dynamics prior to the suppression era included substantially more fire, including both wildfire and prescribed cultural burns. More frequent and/or more severe disturbances, including altered wildfire regimes and insect outbreaks, may cause extensive tree mortality, especially where high forest density and water stress have reduced tree vigor. Fire suppression is the primary non-climate stressor that impacts mixed conifer and ponderosa forests in northern California, and associated increases in tree density over the past century has resulted in greater competition and moisture stress. Competition for resources has reduced tree vigor, leading to altered species composition and further changes in forest structure. These changes increase forest vulnerability to high-severity fire, widespread insect attack, and drought-related mortality. Human infrastructure (e.g., development and roads/highways) can also contribute to altered wildfire regimes by increasing human ignitions and limiting opportunities for fire management, especially within the wildland-urban interface (WUI).

Adaptive Capacity Summary	*Factors that enhance adaptive capacity*: + One of the most extensive habitat types in northern California + High topographic complexity in the region increases resistance to large-scale disturbances + High species diversity may allow shifts in response to changing climate conditions + Management practices focused on reducing forest density and restoring low- to moderate-severity fire regimes may alleviate some climate impacts (e.g., drought, more frequent high-severity wildfire) *Factors that undermine adaptive capacity*: − Limited habitat continuity and species dispersal due to large high-severity wildfires − Structural diversity and functional integrity reduced by fire suppression − Delays and/or declines in forest recovery are likely under increasingly dry conditions, especially in areas where trees are already close to their tolerance limits

Mixed conifer and ponderosa forests in northern California are widely distributed and habitat continuity is generally high, especially in more remote locations. However, large patches of high-severity fire can reduce landscape permeability and limit species dispersal, especially where post-fire conifer regeneration is limited, such as on dry sites. Warmer, drier future conditions may further delay forest recovery in these areas, leading to potential longer-term shifts in species composition and/or conversion to non-forested vegetation. Structural diversity in mixed conifer and ponderosa forests has declined following a century of selective logging and fire suppression, resulting in higher densities of small trees and associated increases in vulnerability to climate stressors (e.g., drought) and disturbances (e.g., fire, insect attacks). However, the high species diversity in this habitat type increases adaptive capacity by potentially allowing shifts towards species better suited to future conditions (e.g., drought-tolerant and/or fire-adapted). High public and societal value increases support for management within this habitat type, and the scientific literature suggests that many management actions focused on restoring heterogeneous forest landscapes (e.g., reducing forest density, restoring frequent fire) may successfully decrease the impact of climate stressors and climate-mediated changes in disturbance regimes.

Sensitivity and Exposure

Mixed conifer and ponderosa forests were evaluated by regional experts as having moderate-high overall sensitivity (high confidence in evaluation) and moderate 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 conifer and ponderosa forest distribution in northern California will remain within climatically suitable areas by the end of the century, although conditions at lower elevations within the inland Coast Range, Klamath Mountains, and southern Cascades may become unsuitable under a hot/dry high-emissions future scenario (Thorne et al. 2016, 2017; Reilly et al. 2018). However, even within otherwise climatically suitable areas, range contractions may occur due to projected expansion of oak woodlands and mixed evergreen forests (Lenihan et al. 2008). Type conversion following high-

severity wildfire may also result in range contraction of mixed conifer and ponderosa forests (Lenihan et al. 2008). In addition, the narrower climatic tolerance range of seedlings may limit recruitment, especially at the xeric edges of a species' range (Dobrowski et al. 2015). However, variation in topography and forest structure may mediate some climate impacts by providing microsites that facilitate seedling establishment (Dobrowski et al. 2015).

Potential areas of refugia for this habitat type may include north-facing slopes, valley bottoms with deep soils, 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

Potential Changes in Habitat Distribution
• Reduced climatic suitability for lower-elevation mixed conifer forests by 2100 within inland areas of northern California • Likely range contraction of mixed conifer forests due to the expansion of hardwoods and/or type conversion following high-severity wildfire • Potential for limited seedling recruitment in otherwise climatically-suitable areas due to narrower seedling tolerance ranges • Potential areas of refugia include north-facing slopes, valley bottoms, and perennial water bodies
Source(s): Lenihan et al. 2008; Flint et al. 2013; Dobrowski et al. 2015; Thorne et al. 2016

forest with dense, closed canopies, particularly those on north-facing slopes, may also provide important microrefugia that protect species sensitive to thermal stress, increasingly dry conditions, and wildfire (Olson et al. 2012; Frey et al. 2016).

Sensitivity and future exposure to climate and climate-driven factors

Regional experts evaluated mixed conifer and ponderosa forests as having moderate 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 conifer and ponderosa forests include climatic water deficit, soil moisture, air temperature, precipitation amount and timing, timing of snowmelt/runoff, and drought.[1]

Climatic water deficit and soil moisture
Water availability and corresponding moisture stress is a primary driver of tree recruitment (Ackerly et al. 2015), growth (Restaino et al. 2016; Vernon et al. 2018), and mortality (van Mantgem & Stephenson 2007; van Mantgem et al. 2009; Das et al. 2013), ultimately playing a large role in determining forest distribution across the western U.S. (Stephenson 1998; Lutz et al. 2010; Ackerly et al. 2015; Anderegg et al. 2015b; Reilly et al. 2018). Climatic water deficit (CWD) is a "plant-relevant" measurement of moisture stress that takes into account the interaction between water (precipitation) and energy (temperature; Stephenson 1998).[2] The

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[1] Climate and climate-driven factors presented are those ranked as having a moderate or higher impact on this habitat type; additional climate and climate-driven factors that may influence the habitat to a lesser degree include storms.

[2] CWD, which is 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).

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). Due to increased evaporative demand as temperatures rise, even areas where precipitation may increase are expected to see a rise in CWD under future climate conditions (Thorne et al. 2015; Dobrowski et al. 2015; Restaino et al. 2016; Micheli et al. 2018). However, factors such as soil depth and drainage significantly affect the water-holding capacity of soil, and topographic features such as north-facing slopes can mediate CWD at a site-level scale (Dobrowski 2011; Flint et al. 2013; Reilly et al. 2018).[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).

Increases in CWD over the past century have reduced tree growth in several species associated with mixed conifer and ponderosa forests, including ponderosa pine (McDowell et al. 2009) and Douglas-fir (Restaino et al. 2016). Growth declines tend to be greater on dry sites within a species' distribution (e.g., low elevations, southern edge of range), where water is an important limiting factor (McDowell et al. 2009; Restaino et al. 2016). On more mesic sites, trees may experience shifts in the timing of growth-limiting factors as CWD increases, with water stress limiting growth earlier in the year (Restaino et al. 2016).

Biotic factors (e.g., insects, disease) are the dominant causes of background tree mortality in mixed conifer and ponderosa forests, and reduced vigor within water-stressed trees enhances vulnerability to tree mortality from insects, disease, and wildfire, particularly on dry sites (van Mantgem & Stephenson 2007; van Mantgem et al. 2009, 2013; Das et al. 2016). For instance, high CWD is associated with decreased resin duct size in sugar pine, impacting the effectiveness of defense systems that protect against bark beetle attack (Slack et al. 2017). High levels of CWD also increase the risk of broad-scale forest mortality during meteorological drought events (Young et al. 2017), and moisture stress is strongly associated with increased wildfire activity (e.g., number of fires, fire size, and area burned) in northern California forests (Abatzoglou & Kolden 2013; Abatzoglou & Williams 2016; Mann et al. 2016; Westerling 2016), likely due in large part to reduced fuel moisture (Abatzoglou & Williams 2016). Post-fire conifer regeneration may be impacted as CWD continues to increase over the coming century, particularly on dry sites where trees are already near the edge of their tolerance (Tepley et al. 2017). In dry forest types, delayed conifer regeneration may result in type conversion to non-forested vegetation (e.g., chaparral; Lenihan et al. 2008; Stevens-Rumann et al. 2017; Tepley et al. 2017).

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[3] Soil moisture is based on CWD and soil properties (e.g., porosity, depth, underlying geology), which determine the soil water-holding capacity.

Regional Climatic Water Deficit (CWD) & Soil Moisture Trends[4]	
Historical & current trends:	*Projected future trends:*

• 1.1 cm (0.4 in) decrease to 0.4 cm (0.2 in) increase in average annual CWD between 1900 and 2009 for the Northwestern California and Southern Cascade ecoregions (Rapacciuolo et al. 2014)
 • No trends available for soil moisture
	• Increases in average annual CWD by 2100 (compared to 1951–1980; Flint et al. 2013; Flint & Flint 2014):
	   ○ 7–24% increase in the Northern Coast Range
	   ○ 5–16% increase in the Northern Interior Coast Range
	   ○ 10–32% increase in the Klamath Mountains
	   ○ 16–43% increase in the Southern Cascades
	 • Increased CWD and decreased top-level soil moisture is likely even if precipitation increases 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 lower elevations and at the southern edge of a species' range
- Increased vulnerability to disease outbreaks and insect attacks, driving higher rates of background tree mortality
- Increased risk of broad-scale forest die-off during drought events
- Increased wildfire activity, including potential increases in fire frequency and size
- Reduced post-fire conifer regeneration, leading to shifts in species composition and/or type conversion to non-forested vegetation

Air temperature

Warmer air temperatures primarily impact mixed conifer and ponderosa forests by increasing evaporative demand (Restaino et al. 2016), which enhances water stress and associated growth declines and tree mortality (van Mantgem & Stephenson 2007; Restaino et al. 2016). Due to elevational temperature gradients, forests at higher elevations experience less water stress compared to those at lower elevations (van Mantgem & Stephenson 2007; Chmura et al. 2011). Warmer temperatures are also reducing snowpack and contributing to earlier timing of snowmelt (Hamlet et al. 2005; Stewart et al. 2005; Gergel et al. 2017), decreasing stored soil moisture and lengthening the period of summer drought (Chmura et al. 2011). Warmer temperatures can also contribute to shifts in species composition and forest distribution (Lenihan et al. 2008).

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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.

Spring and summer temperature is significantly correlated with area burned in mid- to high-elevation forests (Keeley & Syphard 2015; Westerling 2016), likely due in part to reductions in fuel moisture (Abatzoglou & Williams 2016). Warmer air temperatures can also increase rates of growth and reproduction in forest pests (e.g., bark beetles) and pathogens (e.g., fungal diseases; Hicke et al. 2006; Bentz et al. 2010; Kliejunas 2011). For instance, reduced cold-related mortality in bark beetles increases the likelihood of large-scale insect outbreaks that can cause extensive mortality in mixed conifer and ponderosa forests (Hicke et al. 2006; Scheller et al. 2018).

Regional Air Temperature Trends	
Historical & current trends: 0.03°C (0.05°F) decrease to 0.2°C (0.4°F) increase in the average annual temperature between 1900 and 2009 for the Northwestern California and Southern Cascade ecoregions (Rapacciuolo et al. 2014) - No seasonal temperature trends available	*Projected future trends:* 2.2–5.5°C (4.0–9.9°F) increase in the average annual temperature by 2100 (compared to 1951–1980) for the Northern Coast Range, Northern Interior Coast Range, Klamath Mountain, and Southern Cascade ecoregions (Flint et al. 2013; Flint & Flint 2014) 2.0–5.8°C (3.6–10.4°F) increase in average winter minimum temperatures 2.2–6.7°C (4.0–12.1°F) increase in average summer maximum temperatures
Summary of Potential Impacts on Habitat *(see text for citations)*	
- Increased forest water stress due to enhanced evaporative demand and temperature-driven changes in snowpack and the timing of snowmelt - Decreased tree growth and increased tree mortality - Shifts in species composition and forest distribution - Increase in annual area burned, likely due to reductions in fuel moisture - Increased rates of growth and reproduction in insect pests and pathogens	

Precipitation amount and timing

Because water is the limiting factor for growth in most low- to mid-elevation forests in northwestern California (Das et al. 2013; Restaino et al. 2016), changes in the amount and/or timing of precipitation would alter seasonal patterns of moisture stress, impacting tree growth and mortality and potentially leading to shifts in species composition along moisture gradients (Chmura et al. 2011; Restaino et al. 2016; Reilly et al. 2018). Species sensitivity to precipitation patterns varies; for instance, ponderosa pine and Jeffrey pine are well-adapted to xeric sites and annual periods of summer drought (Legras et al. 2010; Welch et al. 2016), while Douglas-fir and white fir tend to occupy more mesic sites (Sawyer 2007; North et al. 2016).

Changes in the timing and/or amount of precipitation may increase the vulnerability of forests to severe wildfires and other disturbances (Millar & Stephenson 2015; Gergel et al. 2017). This is particularly likely if precipitation decreases or a greater proportion of annual precipitation occurs as winter rain instead of snow, effectively increasing the length of the summer dry

season (Flint et al. 2013; Flint & Flint 2014; Gergel et al. 2017). Low precipitation leading up to and during the current fire season is correlated with increased fire activity, largely due to reductions in fuel moisture that enhance fire size and rate of spread (Miller et al. 2009; Littell et al. 2009; Abatzoglou & Kolden 2013; Parks et al. 2014). Patterns of post-fire conifer recovery are also strongly influenced by the region's high interannual precipitation variability, with pulses of regeneration occurring in wet years and reduced rates of recovery occurring during very dry years (Welch et al. 2016; Tepley et al. 2017). Under changing climate conditions, fewer cool/wet years could increase post-fire conifer regeneration delays or failures, especially on dry sites (Tepley et al. 2017). However, topographical variation can create microsites with higher levels of soil moisture where seedling recruitment can remain relatively high even in dry years, as long as seed sources remain available (Donato et al. 2009a; Malone et al. 2018).

Regional Precipitation Trends	
Historical & current trends:	*Projected future trends:*
• 7.2–9.4 cm (2.8–3.7 in) increase in mean annual precipitation between 1900 and 2009 for the Northwestern California and Southern Cascade ecoregions (Rapacciuolo et al. 2014)	• 20% decrease to 34% increase in mean annual precipitation by 2100 (compared to 1951–1980) for the Northern Coast Range, Northern Interior Coast Range, Klamath Mountain, and Southern Cascade ecoregions (Flint et al. 2013; Flint & Flint 2014)[5] • 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)
Summary of Potential Impacts on Habitat *(see text for citations)*	
• Altered patterns of tree growth and mortality • Potential shifts in species composition along moisture gradients • Increased fire activity due to low fuel moisture • Reduced and/or delayed post-fire conifer recovery as the frequency of cool/wet years decreases	

Timing of snowmelt and runoff

Earlier timing of snowmelt and runoff is likely to increase water stress in mixed conifer and ponderosa forests by accelerating the loss of stored water at high elevations, reducing growing season soil moisture and leading to longer periods of summer drought (Bales et al. 2011; Gergel et al. 2017). The timing of snowmelt is closely associated with early seasonal growth at higher-elevation sites, and earlier snowmelt may increase forest productivity in these stands (Royce & Barbour 2001; Chmura et al. 2011). However, earlier onset of the summer drought period could

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

negate any potential gains in the length of the growing season (Royce & Barbour 2001; Chmura et al. 2011).

Longer snow-free periods will also reduce fuel moisture in the summer and fall, increasing wildfire potential in most northern California forests where fire has become increasingly limited by climate conditions rather than fuel availability since Euro-American settlement (Taylor et al. 2016; Gergel et al. 2017).

Regional Snowmelt Trends	
Historical & current trends: • 15–40-day shift towards earlier date of 90% snowmelt across the western U.S. since 1915 (Hamlet et al. 2005) • 10–30-day shift towards earlier timing of snowmelt-driven runoff across the western U.S. since 1948 (Stewart et al. 2005)	*Projected future trends:* • Likely 5–15-day shift towards earlier timing of snowmelt-driven runoff in northern California by 2100 (up to 60-day shift across the western U.S.; Stewart et al. 2004; Rauscher et al. 2008)
Summary of Potential Impacts on Habitat *(see text for citations)*	
• Reduced soil moisture and longer periods of summer drought, increasing forest water stress • Reduced fuel moisture in the summer and fall	

Drought

More frequent, longer, and/or more severe drought conditions are likely to increase tree mortality across all forest and woodland types (van Mantgem et al. 2009; Allen et al. 2010, 2015; McDowell & Allen 2015; Millar & Stephenson 2015; Young et al. 2017; Reilly et al. 2018; Stephens et al. 2018), particularly in drier areas (McDowell et al. 2009; Young et al. 2017). During the record-breaking drought in 2012–2016, tree mortality rates across California were relatively stable for the first several years but rose dramatically during the fourth year, likely demonstrating a lagged mortality response to drought (Young et al. 2017). This may be due to stored carbohydrates that mitigate the impacts of reduced soil moisture until reserves are depleted and/or a gradual decline in photosynthesis as hydraulic damage (i.e., reduced ability to move water from roots to leaves) occurs (Anderegg et al. 2015b, 2015a). Delayed mortality in drought-stressed trees may also be due to increased vulnerability to wildfire, insect outbreaks, and disease (McDowell & Allen 2015; Millar & Stephenson 2015; Kolb et al. 2016; Young et al. 2017; Stephens et al. 2018). Trees often grow at a reduced rate for several years following a drought event (Anderegg et al. 2015a; Vernon et al. 2018), with the largest growth reductions occurring in dry forests and among species vulnerable to hydraulic failure (Anderegg et al. 2015a).

Longer, hotter droughts may lead to significant increases in tree mortality (Allen et al. 2015; Millar & Stephenson 2015; Young et al. 2017). Where fire suppression has increased tree density, competition for soil moisture is higher and trees are more vulnerable to drought-related mortality (van Mantgem et al. 2016; Young et al. 2017; Vernon et al. 2018). Additionally, wildfire activity increases during and after drought periods as fuel moisture declines

(Abatzoglou & Williams 2016; McKenzie & Littell 2017; Stephens et al. 2018). Dead and dying trees can alter fuel structure and patterns of availability (i.e., canopy vs. surface fuels) for several decades, potentially leading to changes in fire behavior and severity (Stephens et al. 2018). While higher moisture levels can moderate fire spread and intensity in closed-canopy forests in historically productive areas, these forests may experience significant shifts in structure as they dry out, resulting in more intense wildfires and subsequent shifts towards even-aged stands that perpetuate a cycle of more severe fires (Odion et al. 2004). Finally, because seedlings are disproportionately affected by drought (Dobrowski et al. 2015), post-fire conifer recovery may be delayed or reduced during periods of drought (Tepley et al. 2017).

Regional Drought Trends	
Historical & current trends:	*Projected future 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)	- 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 insect outbreaks and disease in drought-stressed trees - Reduced fuel moisture leading to increased fire activity - Increased fuel availability and changes in fuel structure due to dead and dying trees - Decreased rates of forest recovery following disturbances such as wildfire	

Sensitivity and future exposure to changes in natural disturbance regimes

Regional experts evaluated mixed conifer and ponderosa forests as having moderate-high sensitivity to changes in natural disturbance regimes (high confidence in evaluation), with an overall moderate future exposure to these stressors within the study region (high confidence). Key natural disturbance regimes that affect mixed conifer and ponderosa forests include wildfire and insects.[6]

Wildfire
Altered wildfire regimes may cause significant shifts in the species composition, forest structure, and distribution of mixed conifer and ponderosa forests in northern California over

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[6] Disturbance regimes presented are those ranked as having a moderate or higher impact on this habitat type; additional changes in disturbance regimes that may influence the habitat to a lesser degree include disease.

the coming century (Skinner et al. 2006; Lenihan et al. 2008; Norgaard et al. 2016; Tepley et al. 2017). Fire is the primary driver of natural forest succession in this habitat type, which historically experienced low- to moderate-severity fire regimes in drier ponderosa and mixed conifer forest types (Skinner & Taylor 2006; Safford & Stevens 2017). In moist mixed conifer forests, fires were typically of mixed severity,[7] especially within the Klamath region where topographic and geologic complexity is high (Taylor & Skinner 2003; Skinner et al. 2006; Halofsky et al. 2011; Perry et al. 2011; Odion et al. 2014). The complex patterns of fire that characterize northern California forests contribute to the high landscape heterogeneity typical of the region by creating diverse forest structure within patches of forests in varying stages of succession (Odion et al. 2004; Skinner et al. 2006; Skinner & Taylor 2006; Perry et al. 2011; Reilly & Spies 2015, 2016).

The composition and structure of mixed conifer and ponderosa forests in northern California is highly adapted to fire (Taylor & Skinner 2003; Skinner et al. 2006, 2009; Reilly et al. 2018), and fires were relatively frequent fires (mean intervals of ~2–20 years) before fire suppression became widespread in the early 1900s (Taylor & Skinner 2003; Fry & Stephens 2006; Skinner et al. 2009; Van de Water & Safford 2011; Perry et al. 2011; Lake 2013; Safford & Van de Water 2014; Steel et al. 2015; Reilly et al. 2018).[8] Cultural burning has been practiced by tribes within the region for centuries, and these practices help maintain this fire-adapted forest type (Fry & Stephens 2006; Lake 2007, 2013; Lake & Long 2014; Norgaard et al. 2016; Karuk Tribe 2019), with burning of black oak, giant chinquapin, gooseberry, and beargrass resource patches occurring every 5–10 years (Hummel et al. 2012; Norgaard et al. 2016). Routine burning also benefits wildlife species by maintaining cavities as dens/nesting sites, areas of early-successional vegetation as forage for black-tailed deer, and habitat for migratory birds and prey (Long et al. 2016; Norgaard et al. 2016; Karuk Tribe 2019).

Fire suppression has contributed to the development of dense stands and shifted species composition toward species that are fire-sensitive at smaller sizes (Leonzo & Keyes 2010; Safford & Stevens 2017), thereby increasing susceptibility to high-severity wildfires (Taylor & Skinner 2003; Safford & Stevens 2017). Prior to the implementation of fire suppression, dry mixed conifer and ponderosa forests were fuel-limited, with frequent fires keeping fuel loads relatively small and preventing the development of large areas that burned at high severity (Taylor & Skinner 2003; Fry & Stephens 2006; Safford & Stevens 2017). Fuel-limited forests had a generally open structure where large, more fire-resistant trees and a sparse understory were interspersed with patches of mixed chaparral (Safford & Stevens 2017). Moister mixed conifer forests were historically limited by a combination of climate conditions and fuel availability (Taylor et al. 2016; McKenzie & Littell 2017). However, since Euro-American settlement in the

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[7] Definitions of mixed-severity fires vary somewhat, but typically describe 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 (Hessburg et al. 2007, 2016; Halofsky et al. 2011; Perry et al. 2011). 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).
[8] Refer to the section on non-climate stressors for a more complete discussion of the impacts of fire suppression on mixed conifer forest habitats.

1800s, fire occurrences in all mixed conifer and ponderosa forests have become increasingly limited by climate conditions rather than fuel availability (Skinner et al. 2009; Halofsky et al. 2011; Steel et al. 2015; Taylor et al. 2016; McKenzie & Littell 2017).

Although mixed conifer and ponderosa forests as a whole are highly fire-adapted, individual species have varying sensitivity to fire (Skinner et al. 2006; North et al. 2016). Ponderosa pine, Jeffrey pine, sugar pine, and large Douglas-fir are more fire-resistant than other species (McCune 1988; Safford & Stevens 2017), and ponderosa pine seedling establishment is enhanced on patches of bare mineral soil in burned areas (Bailey & Covington 2002; Malone et al. 2018). Black oak, canyon live oak, and giant chinquapin are able to resprout following topkill, and relatively frequent fire decreases competition for light, nutrients, and moisture by reducing encroaching conifers in patches dominated by hardwoods (Donato et al. 2009b; Cocking et al. 2012; Crotteau et al. 2013; Norgaard et al. 2016). In general, shade-tolerant species (e.g., white fir) are more sensitive to fire, while shade-intolerant species (e.g., pine, oak) are less sensitive to fire (Skinner et al. 2006). However, the relatively rapid understory growth of some shade-tolerant species, such as Douglas-fir, allows them to develop thick fire-resistant bark (Taylor & Skinner 2003; van Mantgem & Schwartz 2003). Greater fire resistance in mature trees due to their thick bark and high crown base heights limits ladder fuels and reduces the risk of crown fires (van Mantgem & Schwartz 2003; Safford & Stevens 2017; Malone et al. 2018). By contrast, seedling mortality is high for all species during even low- to moderate-severity wildfires (Safford & Stevens 2017).

Following severe fires, mixed conifer and ponderosa forests are often dominated by hardwood trees and fire-stimulated shrubs (e.g., *Arctostaphylos*, *Ceanothus*), with conifers filling in and gradually overtopping the early-successional species (Donato et al. 2009a; Odion et al. 2010; Thompson & Spies 2010; Knapp et al. 2012; Crotteau et al. 2013). Climatic water deficit and the proximity of seed sources are important drivers of patterns of post-fire conifer regeneration (Welch et al. 2016; Stevens-Rumann et al. 2017; Tepley et al. 2017; Malone et al. 2018), in addition to interspecific competition (Welch et al. 2016) and abiotic site conditions (e.g., soil type, topography; Donato et al. 2009a; Irvine et al. 2009). Conifer regeneration may be greater where topographic variations are associated with greater moisture availability, such as at high elevations and on north-facing slopes (Malone et al. 2018). However, because most conifers are dependent on seed sources, regeneration declines with increasing distance to a seed source (Welch et al. 2016; Tepley et al. 2017). Thus, regeneration may be delayed within the interior of larger high-severity patches (Welch et al. 2016; Tepley et al. 2017), especially in dry areas (Tepley et al. 2017). Higher propagule pressure may be required to maintain a given seedling density on drier sites under changing climate conditions, and these sites may also experience slower growth and greater growth reductions due to competition for soil moisture (Stevens-Rumann et al. 2017; Tepley et al. 2017).

Post-fire vegetation composition and fuel structure can increase the likelihood of re-burning at high severities (Odion et al. 2010; Thompson & Spies 2010; Coppoletta et al. 2016). Shifts in fuel dynamics associated with changes in species composition and forest structure could further alter fire behavior, size, and intensity (Parks et al. 2016; McKenzie & Littell 2017; Stephens et al.

2018). Repeated high-severity fires could drive a larger-scale transition to non-forested vegetation (e.g., shrublands, savannah) in some areas (Lenihan et al. 2008; Odion et al. 2010; Thompson & Spies 2010; Coop et al. 2016; Coppoletta et al. 2016; Tepley et al. 2017), especially where climatic water deficit increases and/or seed sources are lost (Tepley et al. 2017). Although patches of fire-tolerant shrubs were a greater part of the forest landscape before fire suppression became widespread (Skinner 1995; Odion et al. 2010), shifts toward warmer, drier conditions and increased vulnerability to disturbance-related mortality (e.g., insects, drought) (van Mantgem & Stephenson 2007; van Mantgem et al. 2013; Allen et al. 2015; McDowell & Allen 2015; Stephens et al. 2018) could perpetuate a cycle of severe fire and reduced rates of forest recovery that result in a more significant departure from historical forest structure, composition, and distribution (Lenihan et al. 2008; Welch et al. 2016; Airey Lauvaux et al. 2016; Coppoletta et al. 2016; Tepley et al. 2017). While vegetation shifts such as these have occurred repeatedly within the geological history of the region (Whitlock et al. 2008; Briles et al. 2011), the combination of climate changes and anthropogenic land use is likely to result in 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; CDFW 2016).

Regional Wildfire Trends

Historical & current 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):
    o 184–274% increase in frequency
    o 270–492% increase in total area burned
    o 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)
	Projected future trends:
	 • 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)
	    o Greatest increases in burned area (up to 400%) occur in montane forested areas in northern California (Westerling et al. 2011; Westerling 2018)
	    o 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)
	    o 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

Regional Wildfire Trends	
o The relatively short period of record for fire severity data may obscure long-term trends o To date, there are no peer-reviewed studies on trends in northern California fire severity that include data from the last ten years	(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) o 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*:
  - o Increased mortality in high-severity patches, resulting in the loss of mature trees that act as seed sources and/or provide food (e.g., fruit and nuts) and wildlife denning habitat
- *Short-term (~2-year)*:
  - o Increased dominance of black oak, chinquapin, and other sprouting species following low- to moderate-severity fire that eliminates competing conifers
  - o Delayed conifer recovery under warmer, drier conditions, especially in high-severity patches
  - o Changes in wildlife habitat quality and availability in larger high-severity patches, with benefits for some species (e.g., deer, woodpeckers) and negative impacts for others (e.g., northern spotted owl)
- *Long-term*:
  - o Increased forest heterogeneity following low- to moderate- and mixed-severity fires due to patches of early-successional vegetation and persistence of oaks and chinquapin
  - o Shifts in species composition and forest structure following high-severity fire
  - o Possible increases in forest homogeneity following large high-severity fires
  - o Potential type conversion to non-forested vegetation (e.g., shrublands, savannah) following severe fires, particularly on dry sites and/or where fires are too frequent to allow forest recovery

Insects

Insects are one of the primary causes of biotic mortality in mixed conifer and ponderosa forests (van Mantgem & Stephenson 2007; Das et al. 2016). At endemic levels, insects act as important natural disturbances, contributing to a heterogeneous forest structure by creating canopy gaps, snags, and downed logs (Spies et al. 2006; Meigs et al. 2015; Fettig 2016). Insects must overcome a variety of thresholds at multiple spatial and temporal scales in order to cause a large-scale outbreak (Raffa et al. 2008), but when these occur extensive tree mortality at a landscape scale can significantly alter forest composition and structure (Raffa et al. 2008; Hicke et al. 2016; Scheller et al. 2018). Changing climate conditions may impact the prevalence and severity of insect outbreaks by directly impacting insect development and survival, as well as by affecting 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). For instance, bark beetle outbreaks are more likely to occur when the abundance of host species increases and/or forests become denser and more homogenous, largely because competition for water and other resources is greater in these areas (Fettig et al. 2007; Bentz et al. 2010). Water stress reduces tree vigor and impairs defenses by limiting the ability of individual trees to

expel attacking insects (Bentz et al. 2010; Weed et al. 2013; Kolb et al. 2016; Slack et al. 2017). During periods of drought, insect populations can rapidly increase and move across the landscape attacking stressed trees (Bentz et al. 2010; Weed et al. 2013; 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).

Warmer temperatures may contribute to earlier bark beetle emergence, the completion of additional life cycles with a single year, reduced cold-related mortality, and expanding distributions for species typically restricted to more southern ranges (Powell & Logan 2005; Hicke et al. 2006; Bentz et al. 2010). Large-scale mountain pine beetle (*Dendroctonus ponderosae*, outbreaks are likely to occur when warmer temperatures allow a combination of timing (i.e., emergence at the appropriate time of the season) and synchrony (i.e., life cycle completed in one year) in beetle development (Powell & Logan 2005; Bentz et al. 2010). The influence of temperature is non-linear and mountain pine beetles may become more or less synchronous as temperatures increase, resulting in sudden large outbreaks where previously unfavorable conditions become benign (Powell & Logan 2005; Bentz et al. 2010). Western pine beetles (*D. brevicomis*), which attack all pine species, produce 2–4 broods per year in California (DeMars & Roettgering 1982; Owens 2003). Anecdotal reports suggest that western pine beetles have been able to complete an additional generation in California because of increased temperatures resulting from climate change, which increases the species' ability to initiate a large-scale attack across the landscape (Vuln. Assessment Reviewer, pers. comm., 2018).

Because most insects are genus- or species-specific, the high tree species diversity within northern California mixed conifer and ponderosa forests can help limit widespread forest mortality (Vuln. Assessment Workshop, pers. comm., 2017). However, high rates of mortality in key forest species (e.g., ponderosa pine) can potentially shift species composition, favoring shade-tolerant conifers (e.g., incense cedar) within the understory (Stephens et al. 2018; Scheller et al. 2018). Hardwoods such as black oak may also benefit as light availability is increased, particularly in the absence of the fire-stimulated shrubs that normally accompany the loss of overstory trees (Stephens et al. 2018). Although some studies have suggested that insect-related tree mortality could impact fire behavior (e.g., spread, intensity, severity) by altering the proportion of large woody fuels to surface fuels (Hicke et al. 2012; Stephens et al. 2018), no consistent relationship between insect outbreaks and fire behavior 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. Landscape-scale outbreaks may also result in reduced forest structural and species diversity, further decreasing forest resilience to climate-related stressors and disturbance (Lake 2007; Stephens et al. 2018; Vuln. Assessment Workshop, pers. comm., 2017).

Regional Insect Outbreak Trends	
Historical & current trends:	*Projected future trends:*
• Bark beetle outbreaks occurred in most western U.S. forests from 1997–2012, with significant increases in beetle-related mortality after 2000 (peaking from 2008–2012; Hicke et al. 2016) • Compared to other forested areas in the western U.S., northern California experienced relatively low beetle-related mortality during the past two decades (Hicke et al. 2016)	• More severe insect outbreaks are likely as temperatures increase and periods of drought become more frequent (Hicke et al. 2006; Bentz et al. 2010; Kolb et al. 2016) • The average elevation of mountain pine beetle outbreaks is expected to shift upslope, resulting in an overall decrease in the area of forest vulnerable to outbreaks by the end of the century (Williams & Liebhold 2002; Hicke et al. 2006)
Summary of Potential Impacts on Habitat *(see text for citations)*	
• Increased rates of tree mortality, especially during and after periods of drought and in dense areas • Shifts in forest composition and structure, reducing landscape-scale heterogeneity	

Sensitivity and current exposure to non-climate stressors

Regional experts evaluated mixed conifer and ponderosa forests as having moderate sensitivity to non-climate stressors (high confidence in evaluation), with an overall moderate current exposure to these stressors within the study region (high confidence). Key non-climate stressors that affect mixed conifer and ponderosa forests include fire suppression and human infrastructure (e.g., residential/commercial development, roads/highways/trails).[9]

Fire suppression

Following European settlement and subsequent logging, fire suppression has occurred in most northern California forests since the early 1900s (Skinner et al. 2006; Steel et al. 2015; Safford & Stevens 2017). Over the past century, fire exclusion has increased the density of small and medium-sized trees, simplifying forest structure and composition at the landscape level and increasing forest vulnerability to tree mortality and large-scale disturbance events (e.g., wildfire, insect outbreaks) in dense stands (Taylor & Skinner 2003; Leonzo & Keyes 2010; Perry et al. 2011; Safford & Stevens 2017). Because regeneration of pines and oaks is inhibited in dense stands due to resource competition (Gray et al. 2005; Long et al. 2016; van Mantgem et al. 2016; Young et al. 2017), fire suppression has contributed to shifts in species composition toward shade-tolerant species, including Douglas-fir and white fir (Taylor & Skinner 2003; Leonzo & Keyes 2010; Steel et al. 2015; McIntyre et al. 2015; Welch et al. 2016; Safford & Stevens 2017). Decreased tree vigor in dense stands can also reduce resistance to disturbances, predisposing existing trees to mortality from drought, insects, and disease (van Mantgem et al. 2004; Leonzo & Keyes 2010; Fettig et al. 2013; Safford & Stevens 2017; Slack et al. 2017). Fuel availability has increased dramatically in the absence of frequent fires that limit surface fuels,

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[9] 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 timber harvest (private and federal).

and fires that escape containment are more likely to be severe and spread quickly over large areas (Taylor & Skinner 2003; Spies et al. 2006; Halofsky et al. 2014; Steel et al. 2015; North et al. 2016). Fire suppression has also resulted in the loss of culturally-valued species that require relatively frequent fire for persistence and/or to maintain high production of nuts and berries (e.g., black oak, giant chinquapin; Lake 2007; Lake & Long 2014; Long et al. 2016; Norgaard et al. 2016).

Human infrastructure (e.g., residential and commercial development, roads/highways/trails)
The impacts of human infrastructure on mixed conifer and ponderosa forests is typically localized around population centers and within the WUI (Vuln. Assessment Workshop, pers. comm., 2017). Increased human activity within these areas is associated with more wildfire ignitions, especially within the WUI and popular recreation areas (Syphard et al. 2007, 2009; Mann et al. 2016). Although fuel in these areas is typically plentiful, the risk to human communities limits opportunities to manage wildfire for ecological benefit, and most activities are focused on fire prevention and suppression (Schoennagel et al. 2017).

Adaptive Capacity

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

Habitat extent, integrity, continuity, and permeability

Regional experts evaluated mixed conifer and ponderosa forests as having a high geographic extent (high confidence in evaluation), low-moderate structural and functional integrity (moderate confidence), and high continuity (high confidence). Landscape permeability for mixed conifer and ponderosa forests was evaluated as moderate-high (moderate confidence). High-severity wildfire was identified as the primary barrier to habitat continuity and species dispersal across the study region.[10]

Mixed conifer and ponderosa forests are widely distributed throughout northern California (Sawyer 2007; North et al. 2016; CNPS 2019). At their lower elevational limit, mixed conifer and ponderosa forests may border mixed chaparral shrublands and oak savannas or woodlands (North et al. 2016). At the upper limit, they gradually transition to true fir forests or chaparral (CNPS 2019). Enriched stands may occur on moist sites at higher elevations (Sawyer 2007), and these endemic and highly site-specific stands are considered areas of concern (Vuln. Assessment Workshop, pers. comm., 2017).

Due to the legacy of past management practices (e.g., fire suppression, logging) and ongoing non-climate stressors, the structural and functional integrity of northern California mixed conifer and ponderosa forests has declined over the past 150 years (Skinner & Taylor 2006; Vuln. Assessment Workshop, pers. comm., 2017). For instance, changes in habitat structure and

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[10] All barriers presented were ranked as having a moderate or higher impact on this habitat type.

composition, such as the loss of old, fire-resistant trees and more dense, homogenous stands dominated by small trees, have resulted in reduced resilience to disturbances such as severe wildfire (Skinner & Taylor 2006; Spies et al. 2018). In general, remote areas have typically experienced fewer anthropogenic impacts, although most have still been impacted by many decades of fire exclusion (Leonzo & Keyes 2010).

Northern California mixed conifer and ponderosa forests generally have few barriers to continuity and dispersal, especially in more remote areas (e.g., federal lands with wilderness characteristics (Strittholt & Dellasala 2001; Vuln. Assessment Workshop, pers. comm., 2017). However, large patches of high-severity fire can reduce forest continuity for a period of time, especially where conifer regeneration is delayed on dry sites that lack seed sources (Olson et al. 2012). In some cases, type conversion to non-forested vegetation (e.g., shrublands) can occur (Lenihan et al. 2008; Odion et al. 2010; Airey Lauvaux et al. 2016). To a lesser degree, land-use conversion may fragment and degrade habitats that border existing agricultural and developed areas (Vuln. Assessment Workshop, pers. comm., 2017). The presence of serpentine soils can also act as a barrier to habitat continuity in limited areas (Vuln. Assessment Workshop, pers. comm., 2017). Many mixed conifer and ponderosa stands with old-growth characteristics have been fragmented due to timber management and development over the past century, and further losses of old-growth conditions in northern California forests may result from increasing drought stress and more extensive areas of high-severity wildfire (Spies et al. 2006, 2018; Halofsky et al. 2014).

Habitat diversity

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

Mixed conifer forests are typically characterized by their diverse mix of species, while ponderosa pine is considered a keystone species in ponderosa forests (Vuln. Assessment Workshop, pers. comm., 2017). Conifer dominance is largely determined by site conditions, including disturbance history, topography, and water balance (Taylor & Skinner 2003; Skinner et al. 2006; Sawyer 2007; North et al. 2016; Safford & Stevens 2017). For instance, ponderosa pine typically dominates warm, dry sites across the region; by contrast, white fir is sensitive to even low-intensity fires and tends to occur on cool, moist sites at higher elevations (Skinner et al. 2006; Skinner & Taylor 2006; Sawyer 2007; Safford & Stevens 2017). Other conifer species that may be dominant within mixed conifer forests include Douglas-fir, sugar pine, Jeffrey pine (primarily at higher elevations or on serpentine soils), and incense cedar (in drier forests; Sawyer 2007; Reilly & Spies 2015; North et al. 2016; CNPS 2018; Spies et al. 2018). The distribution of these species is often limited by cold temperatures at high elevations and dry conditions at lower elevations (North et al. 2016; Safford & Stevens 2017). Dry and moist mixed conifer forest types can be differentiated based on the relative proportion of pines to other conifers (e.g., fir), and these two types follow different successional pathways (Halofsky et al. 2014; Safford & Stevens 2017). For instance, ponderosa pine dominates later-successional dry

forests but occurs primarily as an early seral species on moister, more productive sites where it later becomes co-dominant with Douglas-fir and white fir (Halofsky et al. 2014; Reilly & Spies 2015, 2016).

Physical and topographic diversity in northwestern California is extremely high, 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 contribute to a heterogeneous landscape maintained by frequent disturbances at multiple spatial scales, which results in high forest structural diversity and component species diversity and supports many endemic species and unique vegetation communities (Sawyer 2007; North et al. 2016). Montane species from both the Pacific Northwest and the Southwest can be found within northern California, which shares some characteristics in common with both regions (North et al. 2016). Endemic species (e.g., Brewer spruce [*Picea breweriana*]) and some conifers that occur in distinct populations well away from the rest of their distribution (foxtail pine [*P. balfouriana*], Engelmann spruce [*P. engelmannii*], subalpine fir [*A. lasiocarpa*]) also contribute to the high conifer diversity found in northwestern California (Sawyer 2007). At higher elevations in the Klamath Mountains, several enriched stands occur with up to 17 conifer species in a small area, compared with 5–7 species within a typical stand (see Box 1; DellaSala et al. 1999; Cheng 2004; Sawyer 2007).

Box 1. Climate change vulnerability of enriched mixed conifer stands

Localized areas of extraordinarily high species richness occur in the Klamath-Siskiyou region, where high topographic diversity and a wide range of substrates (including serpentine soils) support many relict species and those at or near the edges of their range (Cheng 2004; Sawyer 2007). The richest known area occurs within one square mile of the Russian Wilderness in the Klamath National Forest and includes 17 conifer species and hundreds of understory plants within a small area (Cheng 2004).

The climate change vulnerability of enriched mixed conifer forests is moderate-high, and is driven by a combination of species-specific traits and the condition of the species assemblage as a whole.

- The interaction of drought, fire, insects, and disease could result in many component species reaching a tipping point under future climate conditions, making it difficult to maintain the species assemblage if a significant number of component species are impacted.
- Species-specific habitat requirements, tolerance to changing climate conditions, and dispersal capability are highly variable across the species assemblage, which is dependent on a combination of specific site conditions (i.e., geologic substrate, elevational gradient, soil moisture, etc.) that can support a diverse range of species.
- The adaptive capacity of these stands is supported by very high collective genetic and life history diversity within the species assemblage.
- The geographic extent of these unique species assemblages is very low and most relict populations are isolated, increasing vulnerability to large, severe fires that could result in the near-complete mortality and likely loss of many relict species in regenerating forests.

The high species richness and genetic diversity within the Klamath-Siskiyou region suggests that the region was an important refugium during past changes in climate (Whitlock et al. 2008; Briles et al. 2011; Olson et al. 2012; Schierenbeck 2017), and it may remain so, acting as a

reservoir for genetic material that may allow shifts in species composition and/or adaptation to future climate conditions (Schierenbeck 2017). Distributional shifts towards higher elevations are possible, although range contractions may be more likely for species already at the limits of their tolerance (e.g., ponderosa pine in dry, low-elevation sites; Rapacciuolo et al. 2014; Vuln. Assessment Workshop, pers. comm., 2017). For species with limited distributions (e.g., foxtail pine), reduced climatic suitability at the edges of their range may impact species persistence on the landscape to a greater degree (Schierenbeck 2017). Reductions in species occurrence due to the legacy of management practices over the past century (e.g., fire suppression, selective logging) may further reduce the potential for dynamic changes in habitat composition and structure in response to warmer, drier conditions (Skinner & Taylor 2006; Spies et al. 2006; Leonzo & Keyes 2010; Stewart et al. 2016; Safford & Stevens 2017).

Resistance and recovery

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

The topographic complexity of the region increases the resistance of mixed conifer and ponderosa forest to climatic stressors by providing varied microsites and areas of refugia from increasing temperatures, water stress, and wildfire (Olson et al. 2012; Frey et al. 2016; Tepley et al. 2017). High structural diversity within the forest also increases resilience to disturbances by limiting impacts within a mosaic of successional stages and promoting relatively rapid recovery (Skinner et al. 2006; Sawyer 2007; North et al. 2016). In addition, the diverse mix of species found within mixed conifer and ponderosa forests can allow shifts in species composition in response to changing conditions, increasing overall habitat resistance and recovery (Vuln. Assessment Workshop, pers. comm., 2017).

Fire suppression has led to denser, less heterogeneous mixed conifer and ponderosa forests across the region (Skinner & Taylor 2006). These areas are significantly more vulnerable to insect outbreaks and high-severity wildfire, and may be less able to recover following large-scale disturbances (Spies et al. 2006, 2018; Millar & Stephenson 2015; Tepley et al. 2017). Reducing tree density, such as through the use of mechanical forest treatments (e.g., thinning) and prescribed fire, likely increases forest resistance to the effects of drought by decreasing the risk of mortality in the remaining trees (van Mantgem et al. 2016; Young et al. 2017; Bradford & Bell 2017; Stephens et al. 2018; Vernon et al. 2018). Reducing forest density may also decrease vulnerability to the effects of severe wildfires that are outside of the natural range of variability in this forest type (Hessburg et al. 2016; Schoennagel et al. 2017; Krofcheck et al. 2018). Natural disturbances (e.g., insect outbreaks) can also result in lower forest densities in some cases; however, the pattern of mortality may not result in a resilient forest structure that mimics the impacts of frequent fire (Stephens et al. 2018). For instance, beetle-related mortality is concentrated in large trees that would be more likely to survive wildfire, while small- and medium-sized trees that are more vulnerable to wildfire are less likely to be affected by beetle attack (Stephens et al. 2018).

Because conifer recovery is strongly associated with moisture availability, the combined impacts of warmer, drier conditions and larger or more frequent high-severity fires may slow regeneration, even where adequate seed sources still occur (Stevens-Rumann et al. 2017; Tepley et al. 2017). Forests at low elevations and on dry sites are particularly vulnerable, as trees are already close to their tolerance limits for water stress (Stevens-Rumann et al. 2017). Given adequate conditions, ponderosa pine and Douglas-fir can rapidly reseed following fire, and Douglas-fir is particularly competitive in mesic areas (Vuln. Assessment Workshop, pers. comm., 2017). However, continued increases in climatic water deficit may increase the likelihood of delays or failures in conifer regeneration (Stevens-Rumann et al. 2017; Tepley et al. 2017).

Management potential

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

Mixed conifer and ponderosa forests are valued by the public and society for recreation (e.g., ATVs, camping, hiking, hunting), aesthetic and intrinsic value, resource extraction (e.g., timber), and wildlife habitat (Vuln. Assessment Workshop, pers. comm., 2017). Mixed conifer and ponderosa forests are also highly valued by northern California tribes, who have managed black oak stands and many other cultural resources within mixed conifer and ponderosa forests for centuries (Lake 2007; Norgaard et al. 2016; Karuk Tribe 2019). Climate change is likely to impact the distribution and abundance of cultural resources found in mixed conifer and ponderosa forests (Voggesser et al. 2013), including many wildlife and plant species that are considered traditional foods (Lynn et al. 2014). In addition to directly impacting culturally-important species, changes in climate and climate-driven factors and altered disturbance regimes may affect access to forests and valued wildlife and plant species, as well as the ability to carry on traditional practices such as cultural burning (Norgaard 2014; Whyte 2014; Norgaard et al. 2016).

Despite their high perceived value, a lack of knowledge about what is required to manage this dynamic forest type can result in insufficient regulatory and societal support for proactive treatments to increase ecosystem resilience to climate change, especially at landscape scales (Hessburg et al. 2016; Schoennagel et al. 2017; Vuln. Assessment Workshop, pers. comm., 2017; Karuk Tribe 2019). For instance, restrictions due to endangered species protections and other regulations can limit the ability of land managers to reduce tree density in overstocked stands through mechanical treatments, prescribed burning, and/or managed wildfire (Evans et al. 2011; Long et al. 2017; Metlen et al. 2017). Carbon sequestration is widely recognized as an important goal in forest management (Ellison et al. 2017; Krofcheck et al. 2018), and activities such as thinning and prescribed burning may not be supported because they reduce carbon stores and increase atmospheric carbon emissions in the short term (Krofcheck et al. 2018).

In order to increase the resilience of mixed conifer and ponderosa forests and promote adaptive ecosystem responses to fire under changing climate conditions, wildfire policy and management should support a shift towards acknowledging the role of fire in ecosystem maintenance and allowing the reintroduction of fire onto the landscape (Jewell & Vilsack 2014; Hessburg et al. 2016; Metlen et al. 2017; Schoennagel et al. 2017; Karuk Tribe 2019). This might include supporting more fire-adapted human communities through policies that disincentivize development within the WUI and require any development that does occur to be designed to reduce fire hazards (Syphard et al. 2014; Schoennagel et al. 2017). These practices would also minimize potentially unsustainable levels of fire management that may be necessary under changing climate conditions (Schoennagel et al. 2017). Currently, challenges related to these management practices include limited funds for management, difficulty obtaining permits for prescribed fire, lack of a year-round workforce, and managing perceptions of risk related to escaping fire (Evans et al. 2011; Metlen et al. 2017; Vuln. Assessment Workshop, pers. comm., 2017). However, many communities, tribal and county governments, and utility companies support fuel treatments that could be climate-adaptive by increasing forest resilience to uncharacteristically severe wildfires (Evans et al. 2011; Norgaard et al. 2016; Schoennagel et al. 2017; Vuln. Assessment Reviewer, pers. comm., 2018; Karuk Tribe 2019). Carefully optimizing the placement of fuel treatment areas and forest restoration efforts on the landscape can minimize impacts to rare species and cultural resources while limiting short-term carbon losses, thus providing the greatest benefits to both human communities and the ecosystem over the long term (Long et al. 2017; Metlen et al. 2017; Krofcheck et al. 2018; Karuk Tribe 2019).

Management capacity and ability to alleviate impacts[11]
Regional experts evaluated the potential for reducing climate impacts on mixed conifer and ponderosa forests through management as low-moderate (high confidence in evaluation).

Regional experts identified use conflicts and/or competing interests for mixed conifer and ponderosa forests as timber harvesting, recreation, cannabis (*Cannabis sativa, C. indica*) cultivation, and vineyards (especially within Mendocino County; Vuln. Assessment Workshop, pers. comm., 2017). Conflicts can arise between parties with differing management goals; for instance, preservation ("hands off") approaches can conflict with resource extraction (e.g., timber harvesting), fuels reduction, and restoration of ecocultural resources and forest conditions desired by tribes (Long et al. 2018; Vuln. Assessment Workshop, pers. comm., 2017).

The scientific literature identifies several management approaches focused on promoting spatial heterogeneity that could reduce the impact of climate stressors and climate-mediated changes in disturbance regimes (Evans et al. 2011; Fettig et al. 2013; Odion et al. 2014; Metlen et al. 2017; Stephens et al. 2018). Active management practices within mixed conifer and ponderosa forests often include mechanical treatments (including thinning), prescribed fire, and planting with the goals of reducing tree density, managing potential fuels, decreasing the risk of large-scale insect outbreaks, and supporting forest regeneration (Latham & Tappeiner

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

2002; Evans et al. 2011; Halofsky et al. 2014; Jewell & Vilsack 2014; Hessburg et al. 2016; Metlen et al. 2017; Stephens et al. 2018; Vernon et al. 2018; Karuk Tribe 2019). These practices will remain relevant under changing climate conditions, although the ideal timing, location, and/or method may shift (Vuln. Assessment Workshop, pers. comm., 2017). Because mixed conifer and ponderosa forests are dynamic systems subject to frequent disturbances, management strategies will need to be even more responsive to site conditions, including vegetation composition, topography, and disturbance history (Halofsky et al. 2014).

Management practices geared toward reducing forest density and removing accumulated fuel loads may limit the magnitude and rate of change in forest structure and distribution, although they are unlikely to prevent changes from occurring altogether (Halofsky et al. 2014; Metlen et al. 2017). For instance, altered wildfire regimes may decrease the number of large and late-seral or old-growth stands even in well-managed areas, though the use of prescribed fire and thinning from below could increase tree vigor and encourage the development of mature forests with larger, more fire-resistant trees (Latham & Tappeiner 2002; Halofsky et al. 2014; Metlen et al. 2017; Spies et al. 2018; Karuk Tribe 2019). Reducing forest density and accumulated fuels through practices such as mechanical restoration, prescribed fire (including cultural burns), and/or managed wildfire may also increase tree growth (Latham & Tappeiner 2002) and enhance forest resistance to drought-related tree mortality (van Mantgem et al. 2016; Vernon et al. 2018), large-scale insect outbreaks (Fettig et al. 2007; Bradford & Bell 2017), and uncharacteristically severe wildfires (Evans et al. 2011; Halofsky et al. 2014; Hessburg et al. 2016; Metlen et al. 2017; Schoennagel et al. 2017; Krofcheck et al. 2018; Karuk Tribe 2019). Given the high cost of fuel treatments across large areas, allocating resources to concentrate thinning in areas that are most likely to experience severe fires is likely to provide the greatest benefits to fire-adapted forest types such as mixed conifer (Krofcheck et al. 2018).

Following severe wildfires, reforestation in key areas with seed sources genetically adapted to projected climate conditions may support a transition towards forests better suited to future conditions (Vuln. Assessment Reviewer, pers. comm., 2018). Maintaining a matrix of forest areas in multiple stages of succession, including early seral stages and stands that exhibit old-growth characteristics (i.e, a mixture of clumped trees and more open areas, heterogeneous multi-layered vertical structure, and the inclusion of shade-tolerant species) is also important to promote landscape heterogeneity and high biodiversity (Spies et al. 2018; Vuln. Assessment Reviewer, pers. comm., 2018). Mature forests with dense closed canopies, in particular, retain moisture and can provide cooler microclimates that may promote species survival under changing climate conditions (Frey et al. 2016; Ellison et al. 2017).

The U.S. Forest Service is currently involved in collaborative efforts to promote fire-resilient systems and to reintroduce fire onto the landscape through prescribed burning (Jewell & Vilsack 2014; Metlen et al. 2017; USDA Forest Service 2018). For instance, the Somes Bar Integrated Fire Management project includes partnerships with the Western Klamath Restoration Partnership (WKRP), and is designed 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 establishing strategic ridge-top fuelbreaks and defensible space in and around the WUI, with the goal of enabling the restoration of Karuk cultural burning practices on Offield Mountain (USDA Forest Service 2018).

Ecosystem services
Mixed conifer and ponderosa forests provide a variety of ecosystem services, including:
- Provisioning of food, fiber, fuel, genetic resources, natural medicines, ornamental resources, and fresh water;
- Regulation of air quality, climate/microenvironments (e.g., shade), flood/erosion control, water purification, pest/disease regulation, pollination, and natural hazard regulation;
- Support of primary production, oxygen production, soil formation/retention, nutrient cycling, water cycling, and carbon sequestration; and
- Cultural/tribal uses for spiritual/religious purposes, knowledge systems, educational values, aesthetic values, social relations, sense of place, cultural heritage, inspiration, and recreation (Vuln. Assessment Workshop, pers. comm., 2017).

Mature and late-seral closed-canopy forests, which primarily occur on productive sites within the mesic areas of the region (Reilly & Spies 2015; Spies et al. 2018), play a particularly large role in regulating water, energy, and carbon cycles (Ellison et al. 2017).

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Further information on the Northern California Climate Adaptation Project is available on the project website (https://tinyurl.com/NorCalAdaptation).

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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),[12] 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, individual species, and the ecosystem services that 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 the vulnerability of each resource. Participants were first asked to describe the habitat and/or to list the species to be considered in 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.[13]

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

[13] 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 were 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 because the uncertainty about the magnitude and rate of future change is greater. 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, from these rankings, estimates the overall level of confidence for each component of vulnerability and then for overall vulnerability.

Stakeholders and decision-makers can consider the rankings and scores presented as measures of relative vulnerability and confidence to compare the level of vulnerability among the focal resources evaluated in this project. Elements that received lower confidence rankings indicate knowledge gaps that applied scientific research could help address.

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 water flow regimes, shifts in vegetation types)
- **Stressors Not Related to Climate:** e.g., tectonic and volcanic events; 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; mining; etc.

Sensitivity & Exposure (Applies to Species Groups and Species)

- **Dependencies:** e.g., dependencies on sensitive habitats, specific prey or forage species, and the timing of the appearance of these prey and forage species (concern for mismatch)

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 distribution
- **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 the impact of stressors once they 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 that a resource produces for human well-being

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 its distribution 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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