EcoAdapt™
Meeting the challenges of climate change

Subalpine Forests
Northern California Climate Change Vulnerability Assessment Synthesis

An Important Note About this Document: This document represents an initial evaluation of vulnerability for subalpine 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 Phil Rundel (University of California, Los Angeles). Vulnerability scores were provided by Eureka and Redding workshop participants.

Table of Contents

Habitat Description

Subalpine forests are relatively rare in northern California, and occur primarily at the highest elevations in the Klamath Mountains region (2,200–2,750 m [7,200–9,000 ft]) and on Mount Shasta and adjacent peaks in the southern Cascades (2,300–2,700 m [7,500–8,860 ft]; Millar & Rundel 2016a). Subalpine forests in northern California are primarily limited by abiotic factors, and are characterized by shallow soils, short growing seasons, and extended periods of snowpack (Skinner et al. 2006; Fites-Kaufmann et al. 2007; Millar & Rundel 2016a). Stand structure typically consists of relatively open woodlands with a discontinuous shrub layer (Skinner et al. 2006). Dominant species may include whitebark pine (*Pinus albicaulis*), foxtail

pine (*P. balfouriana*), mountain hemlock (*Tsuga mertensiana*), Shasta red fir (*Abies magnifica* var. *shastensis*), western white pine (*P. monticola*), Jeffrey pine (*P. jeffreyi*), lodgepole pine (*P. contorta* spp. *murrayana*), and curl-leaf mountain-mahogany (*Cercocarpus ledifolius*; Skinner et al. 2006; Skinner & Taylor 2006; Sawyer 2007; CNPS 2019). Serpentine soils support more open woodlands, which are generally dominated by foxtail pine, western white pine, and whitebark pine (Sawyer 2007; CNPS 2019). Subalpine stands are interspersed with rock outcrops, talus slopes, wet meadows, and morainal lakes (Skinner et al. 2006; Fites-Kaufmann et al. 2007).

Historically, many northern California tribes travelled to high-elevation sites in the summer for hunting and gathering plants, as well as to attend social and ceremonial gatherings (Turner et al. 2011). Many species found within subalpine forests are culturally valued for food, medicine, basketry, and other purposes, including red fir, whitebark pine, lodgepole pine, and mountain hemlock, among others (Turner et al. 2011). More widely distributed species are often sought out at high-elevation sites, as these plants are considered more potent than those from lower-elevation sites (Turner et al. 2011).

Executive Summary

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

Subalpine Forests	Rank	Confidence
Sensitivity	Moderate	High
Future Exposure	High	Moderate
Adaptive Capacity	Low-Moderate	High
Vulnerability	**Moderate-High**	**Moderate**

Sensitivity & Exposure Summary	*Climate and climate-driven factors*: • Air temperature, precipitation amount, snowpack amount, timing of snowmelt and runoff, climatic water deficit, drought *Disturbance regimes*: • Insects, disease, wildfire *Non-climate stressors*: • Livestock grazing

Subalpine forests are primarily sensitive to climate factors that alter moisture availability and the timing/length of the growing season, including changes in precipitation, reduced snowpack, earlier timing of snowmelt, and increased drought. While changes in these factors may increase water stress, particularly at lower elevations within the subalpine zone, warming temperatures and changes in snowpack and snowmelt may also enhance forest productivity by increasing rates of tree growth and seedling establishment. As climate conditions become less harsh at high elevations, tree density could increase in some stands and lower-elevation trees (e.g., fir) and invasive plants may encroach into subalpine zones, potentially outcompeting subalpine

species. Encroachment by fir and other lower-elevation species may also increase the risk of high-severity wildfire due to higher fuel loads within these historically fuel-limited systems. Large-scale disturbances such as mountain pine beetle attacks and outbreaks of white pine blister rust can cause high mortality rates among affected species, particularly where trees are already stressed by drought. Overall, subalpine habitats are much less affected by non-climate stressors compared to other habitats in the assessment area. The legacy impacts of livestock grazing historically degraded forests, but is currently prohibited in most areas.

Adaptive Capacity Summary	*Factors that enhance adaptive capacity*:
	+ High levels of endemism within northern California subalpine zones
	+ Limited impacts of anthropogenic stressors due to remote location
	Factors that undermine adaptive capacity:
	− Small, isolated populations are particularly vulnerable to extirpation
	− Slow tree growth and low habitat continuity limit ability of subalpine species to track climatic changes across the landscape
	− Upslope shifts limited by lack of higher elevation areas in most of the region

Subalpine forests occur on disconnected sky islands in northern California, which limits genetic exchange and opportunities for migration and dispersal in response to climate changes. Additionally, upslope shifts are limited by the lack of higher elevation areas in the northwestern California study area except Mount Shasta. Although subalpine species are adapted to harsh conditions, the slow growth and low reproductive rates characteristic of many species may not allow adaptive genetic traits (e.g., resistance to white pine blister rust) to be passed on before stand extirpation occurs. Subalpine forests have high levels of endemism and harbor many rare and/or relict species, due in large part to the high topographic diversity within the region. Although this habitat is valued for its rarity and is largely protected from human activity, management options are limited in most locations by inaccessibility. Within the scientific literature, suggested management strategies that may reduce subalpine habitat vulnerability to climate change include expanding the development of disease-resistant trees, reducing stand density in some areas to increase resilience to drought and other stressors, and establishing early detection and monitoring programs for invasive plants that may expand into high-elevation areas.

Sensitivity and Exposure

Subalpine forests were evaluated by regional experts as having moderate overall sensitivity (high 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 changes are projected to alter suitable habitat area for subalpine forests across the state, with range contractions likely to occur due to upslope advancement of lower-elevation forest types (e.g., red fir) as temperatures increase (Lenihan et al. 2003, 2008; Hayhoe et al. 2004; Thorne et al. 2016). Several studies project that subalpine forest distribution state-wide may decline by anywhere from 50-90%, with the greatest declines occurring under high-

emissions scenarios (Lenihan et al. 2003, 2008; Hayhoe et al. 2004). Because most subalpine forests extend to the highest elevations of northern California peaks, few opportunities will be available for the upslope shifts that may occur in other regions of the state (Vuln. Assessment Workshop, pers. comm., 2017).

Recent habitat distribution modeling by Thorne et al. (2016, 2017) projects areas of future climatic exposure (i.e., areas likely to experience significant environmental stress and/or type conversion) and climatic

Potential Changes in Habitat Distribution
• Up to 90% decline in subalpine forest distribution state-wide under a high-emissions scenario • *Klamath Mountains:* Up to 93% of current subalpine distribution may no longer be in climatically suitable areas • *Southern Cascades:* Most subalpine areas will no longer be climatically suitable, but some expansion could occur at the highest elevations on Mount Shasta
Source(s): Lenihan et al. 2003, 2008; Hayhoe et al. 2004; Thorne et al. 2016

suitability for subalpine forests by the end of the century.[1] For subalpine forests within the Klamath region, as little as 7% (total of 69 km^2) of the current mapped habitat distribution may remain in climatically suitable areas by 2100 under high-emissions scenarios that represent the hottest/driest conditions (Thorne et al. 2016, 2017). In general, forests closer to the coast will be exposed to greater changes, while interior forests are more likely to remain climatically suitable (Thorne et al. 2016). Some areas may become newly suitable for this habitat type (up to 1,259 km^2) under warmer/wetter climate scenarios, particularly in the Yolla Bolly Mountains on the North Coast; by contrast, only losses of climatically suitable areas occur under the hottest/driest scenarios (Thorne et al. 2016, 2017). Within subalpine forests of the southern Cascades, most areas will likely become climatically unsuitable under the hottest/driest scenarios; however, the highest elevations of Mount Shasta are projected to remain suitable, representing an area of 58 km^2 (Thorne et al. 2016, 2017).

Sensitivity and future exposure to climate and climate-driven factors

Regional experts evaluated subalpine forests as having moderate-high sensitivity to climate and climate-driven factors (moderate confidence in evaluation), with an overall moderate-high future exposure to these factors within the study region (moderate confidence). Key climatic factors that affect subalpine forests include air temperature, precipitation amount, snowpack amount, timing of snowmelt and runoff, climatic water deficit, and drought.[2]

It is possible that warming temperatures and increased water availability could benefit the habitat through greater decomposition and nutrient cycling, lengthening the growing season and decreasing the harshness of subalpine sites (Vuln. Assessment Workshop, pers. comm.,

---

[1] Projections in this study are based on two different future climate models, MIROC ESM (warmer and drier) and CNRM CM5 (warmer and wetter), and two future greenhouse gas emissions scenarios, RCP 8.5 (business as usual emissions) and RCP 4.5 (Paris Accord target emissions). These scenarios encompass minimum temperature increases of 1.9–4.5°C (3.42-8.1°F) and a -24.8 to +22.9% change in precipitation by 2100 relative to 1980–2010 (Thorne et al. 2016, 2017).

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

2017). However, these same factors also increase opportunities for lower-elevation vegetation types (e.g., true firs) and invasive species to shift upslope and outcompete subalpine species (Taylor 1995; Rundel & Keeley 2016; Lembrechts et al. 2018).

<u>Air temperature</u>
Air temperature is strongly tied to patterns of tree growth, recruitment, and mortality in subalpine forests (Taylor 1995; Royce & Barbour 2001; Bunn et al. 2005a; Das et al. 2013; Dolanc et al. 2013b). Warming temperatures, particularly combined with steady or increased precipitation amounts, may reduce the harshness of subalpine sites, allowing increased forest productivity (Taylor 1995; Millar et al. 2004; Bunn et al. 2005a). This pattern has been observed in the past; for instance, tree recruitment in lodgepole pine and mountain hemlock increased during warm periods over the last two centuries (Taylor 1995; Millar et al. 2004). Subalpine forests have increased in density in some areas within the state, such as the central Sierra Nevada, largely due to a shift in stand structure towards a larger proportion of small trees and increased mortality in large trees, particularly Jeffrey pine and western white pine (Dolanc et al. 2013a). Subalpine tree growth responds most readily to warm spring and early summer conditions following a year with high winter precipitation that increases growing season soil moisture availability (Fites-Kaufmann et al. 2007; Dolanc et al. 2013b). However, warmer temperatures may also increase the suitability of subalpine zones for invasive species expanding upslope (Rundel & Keeley 2016; Lembrechts et al. 2018), as many of these are annual species that are currently unable to complete their life cycle within the short growing season characteristic of subalpine areas (Körner 2003).

While warmer temperatures can increase forest productivity at high elevations (Taylor 1995; Millar et al. 2004; Bunn et al. 2005a), they also contribute to reduced snowpack (Hamlet et al. 2005; Mote et al. 2005; Gergel et al. 2017), earlier snowmelt (Stewart et al. 2004; Thorne et al. 2015), and increased evaporative demand (Stephenson 1998; Cook et al. 2014). These factors exacerbate water stress by causing an earlier start and/or longer duration of the summer drought period (Royce & Barbour 2001). Additionally, increased temperatures may reduce cold-related mortality for mountain pine beetles (*Dendroctonus ponderosae*) and allow one or more beetle life cycles to be completed within a single year (Logan & Powell 2001; Hicke et al. 2006; Bentz et al. 2010; Logan et al. 2010). Because of the direct link between temperature and the timing and synchrony of bark beetle development (Logan & Powell 2001; Hicke et al. 2006), warmer conditions are likely to contribute to large-scale outbreaks that may cause extensive mortality in northern California subalpine forests, particularly within whitebark pine populations (Hicke et al. 2006; Millar et al. 2012).

Regional Air Temperature Trends[3]	

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, 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 productivity at high elevations, including tree growth and recruitment   • Shifts in stand structure towards smaller trees and denser forests   • Reduced snowpack and earlier snowmelt, resulting in an earlier start of the summer drought period and increased water stress for subalpine vegetation   • Upslope expansion of invasive species from lower elevations   • Changes in the timing and synchrony of bark beetle development, potentially allowing more frequent large-scale outbreaks

Precipitation amount, snowpack amount, and timing of snowmelt and runoff

Tree growth and seedling establishment is strongly tied to patterns of precipitation and snowpack in northern California subalpine forests (Taylor 1995; Bunn et al. 2005a; Millar et al. 2012; Dolanc et al. 2013b; Millar et al. 2015). Precipitation amounts in northern California occur on a gradient from west to east, with areas closer to the coast (e.g., Yolla Bolly Mountains) receiving more precipitation than those farther inland (Millar & Rundel 2016a). In northern California subalpine zones, most of this precipitation occurs as snow (Millar & Rundel 2016a), and both the growing season and the fire season are limited by snowpack that may persist into late June or July (Skinner et al. 2006). Snowpack persists later in the year where drifts are captured by vegetation and shaded by the forest canopy (Royce & Barbour 2001; Millar & Rundel 2016a), and soil moisture in subalpine zones is strongly mediated by snowpack depth and timing of snowmelt (Dolanc et al. 2013b).

The timing of snowmelt is tied to the beginning of annual growth in conifers (Royce & Barbour 2001; Chmura et al. 2011). Reduced snowpack and earlier snowmelt increase the length of the growing season, allowing higher rates of seedling establishment and recruitment in species

---

[3] 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.

typically limited by deep snowpack (e.g., mountain hemlock; Taylor 1995). However, longer snow-free periods may also increase the potential for invasion by lower-elevation species, including non-native plants (Rundel & Keeley 2016; Lembrechts et al. 2018). In the long term, loss of snowpack and earlier onset of the summer drought period can result in a shorter window for annual growth (Royce & Barbour 2001).

Tree growth in a given year is also dependent, in part, on conditions in the previous growing season (Dolanc et al. 2013b). For instance, wet conditions in the previous year and dry conditions in the current growing season are correlated with increased tree growth, especially for species typical of mesic sites (e.g., mountain hemlock; Dolanc et al. 2013b). This is likely because early-season growth is supported by excess photosynthate that was produced after annual growth had ceased the previous year (Dolanc et al. 2013b). In general, low interannual variability in water balance, as well as temperature, over the course of multiple years facilitates recruitment and growth in high-elevation trees (Millar et al. 2015).

Reduced precipitation, transitions to a greater proportion of precipitation occurring as rain rather than snow, and earlier snowmelt would all result in greater plant water stress (Hamlet et al. 2005; Mote et al. 2005; Knowles et al. 2006; Mote 2006; Gergel et al. 2017), limiting tree growth and recruitment (Bunn et al. 2005a; Dolanc et al. 2013b). However, trees on protected sites that maintain snowpack later into the season may be less affected (Dolanc et al. 2013b). The impacts of increased moisture stress may also be exacerbated by atmospheric pollutants such as ozone and nitrogen, which can cause injuries that decrease water use efficiency in conifers (Grulke 2003), resulting in decreased tree growth (Bytnerowicz et al. 2013) and increased vulnerability to drought (Grulke 2003). By contrast, increased water availability could increase productivity in subalpine zones (Bunn et al. 2005b). However, this would also lead to more favorable conditions for lower-elevation vegetation (e.g., fir), which could expand upslope and outcompete subalpine species (Taylor 1995; Rundel & Keeley 2016; Vuln. Assessment Workshop, pers. comm., 2017).

Regional Precipitation, Snowpack, & Snowmelt 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) • 15–39% decrease in April 1 snow water equivalent (SWE) between 1951 and 2010 for the Northwestern California and Southern Cascade ecoregions (Flint et al. 2013) • 15–40-day shift towards earlier date of 90% snowmelt across the western U.S. since 1915	• 20% decrease to 28% increase in mean annual precipitation by 2100 (compared to 1951–1980) for the Northern Coast Range, Klamath Mountain, and Southern Cascade ecoregions (Flint et al. 2013; Flint & Flint 2014)[4] • 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.

---

[4] 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, Snowpack, & Snowmelt Trends	

(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)
	2018; Swain et al. 2018)
	• Overall, interannual variability is expected to increase (Pierce et al. 2018; Swain et al. 2018)
	• Decreases in April 1 SWE by 2100 (compared to 1951–1980; Flint et al. 2013; Flint & Flint 2014):
	    ○ 82–99% decrease in the Northern Coast Range
	    ○ 72–94% decrease in the Klamath Mountains
	    ○ 61–89% decrease in the Southern Cascades
	• 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 tree growth and seedling establishment where water stress increases, which could be exacerbated by interactions between changing climate conditions and atmospheric pollutants • Earlier start to the growing season as snowpack melts earlier, though earlier onset of the summer drought period can result in a shorter window for annual growth • Longer fire season due to earlier snowmelt

Climatic water deficit and drought

Subalpine tree growth and survival is regulated, in part, by the difference between water supply and evaporative demand (Stephenson 1998; Millar et al. 2012; Das et al. 2013), and moisture stress strongly influences subalpine species distribution (Bunn et al. 2005b; Millar & Rundel 2016a). 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).[5] The balance between water supply and demand shifts over the course of the year, with CWD increasing as soil moisture from the melting snowpack is depleted and evapotranspiration increases in warmer months (Stephenson 1998). In general, protected areas that retain snowpack later into the season have lower CWD (Millar & Rundel 2016a), mediating drought stress on vegetation in those areas (Dolanc et al. 2013b).

Topography and substrate determine the water-holding capacity of a site, and interact with precipitation (rain and snow) to influence plant growth and distribution based on the drivers of water balance on a particular site (Stephenson 1998; Bunn et al. 2005b; Millar & Rundel 2016a).

---

[5] 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). 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; Micheli et al. 2018)..

For instance, subalpine species can grow at lower elevations on steep north-facing slopes that experience lower solar radiation and associated evaporative demand; the same species may be found on a south-facing slope only at much higher elevations (Fites-Kaufmann et al. 2007). Similarly, sites with high CWD due to high evaporative demand (e.g., south-facing slopes) support different plant communities than sites that have high CWD due to thin, rocky soils with low moisture-holding capacity (Stephenson 1998).

While increasing CWD over the past several decades has been correlated with higher background mortality rates in lower-elevation forest types, primarily due to insects and disease, subalpine forests have not exhibited the same trend (van Mantgem & Stephenson 2007). However, CWD does play a role in forest mortality, along with temperature, and CWD may become increasingly important as snowpack declines and the balance shifts towards water as the limiting factor in subalpine forests, rather than temperature (Das et al. 2013). Increased drought stress limits tree growth and increases mortality, especially in large trees (Lutz et al. 2010; Millar et al. 2012; Dolanc et al. 2013a). In general, mortality rates are higher for the *Pinus* genus compared to other conifers, with hemlocks (*Tsuga*) experiencing lower mortality (van Mantgem et al. 2009; Lutz et al. 2010; Dolanc et al. 2013a).

Interannual variability in CWD, together with warm summer temperatures, can increase the potential for insect outbreaks and associated tree mortality (Millar et al. 2012). Drought stress can also increase forest vulnerability to large-scale disturbances, triggering insect and disease outbreaks where stressed trees are unable to adequately defend against pathogens (Raffa et al. 2008; Millar et al. 2012; Millar & Stephenson 2015; Kolb et al. 2016). For instance, during a recent large-scale outbreak, insect-related mortality in whitebark pine was greater on sites with higher CWD relative to the species' range (Millar et al. 2012). Even when annual precipitation is not low, warm winters and early snowmelt can increase CWD; for instance, prolonged exposure to water stress can affect resource accumulation and tree vigor, increasing vulnerability to drought stress in subsequent years (Millar et al. 2012).

Regional Climatic Water Deficit (CWD) & Drought Trends	
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)
• 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)
	• 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
	    ○ 10–32% increase in the Klamath Mountains
	    ○ 16–43% increase in the Southern Cascades
	• Increased CWD 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)
	• Drought years are twice as likely to occur over the next several decades due to increased co-

Regional Climatic Water Deficit (CWD) & Drought Trends	
	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 in large trees
- Increased vulnerability to large-scale disturbances (e.g., insect and disease outbreaks), with higher rates of tree mortality occurring on dry sites

Sensitivity and future exposure to changes in natural disturbance regimes

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

Insects
Native insects such as bark beetles are considered part of the natural disturbance regime within subalpine forests (Logan & Powell 2001). Insects have not historically been a major cause of tree mortality at high elevations, as outbreaks tend to be less frequent and occur at a smaller scale than they do in lower-elevation conifer forests (Logan et al. 2010; Millar & Rundel 2016a; Das et al. 2016). However, high levels of tree mortality can occur in affected stands, particularly those impacted by outbreaks of mountain pine beetles (Logan & Powell 2001; Millar et al. 2012; Krist et al. 2014). The primary hosts of this species are lodgepole pine and whitebark pine, though it can also attack most other high-elevation pines, including foxtail pine and western white pine (Logan & Powell 2001; Krist et al. 2014).

Low temperatures and the persistence of snowpack in late spring and early summer limits mountain pine beetle populations at high elevations by preventing the completion of one or more life cycles during the growing season (Logan & Powell 2001; Bentz et al. 2010). In order for a successful outbreak to occur, a combination of timing (i.e., emergence at the appropriate time of the season) and synchrony (i.e., life cycle completed in one year) must occur (Logan & Powell 2001). These conditions appear to be most sensitive to summer temperatures (Hicke et al. 2006), but the influence of temperature is non-linear and beetles may become more or less synchronous as temperatures increase, resulting in sudden large outbreaks where previously unfavorable conditions become benign (Logan & Powell 2001; Bentz et al. 2010). Generally, the

---

[6] All disturbance regimes presented were ranked as having a moderate or higher impact on this habitat type.

frequency of suitable outbreak conditions in high-elevation forests will increase with warmer temperatures, likely exposing subalpine forests to novel levels of insect disturbance (Logan & Powell 2001; Hicke et al. 2006; Bentz et al. 2010). However, further warming may ultimately cause a decline in beetle outbreaks as the optimal temperature range for both developmental timing and synchrony is surpassed (Hicke et al. 2006). Depending on temperature-dependent life history strategies (e.g., cold-induced mortality and developmental timing), other bark beetle species may be affected differently by warming temperatures (Bentz et al. 2010).

In addition to directly influencing insect development and survival, changing climate conditions can impact the prevalence and severity of insect outbreaks indirectly by altering tree defenses, host susceptibility, and community interactions (Kliejunas 2011; Sturrock et al. 2011; Weed et al. 2013; Kolb et al. 2016). For instance, bark beetle outbreaks may be more likely to occur when the abundance of host species increases and/or forests become denser and more homogenous (Bentz et al. 2010; Millar et al. 2012). Additionally, drought stress, disease, air pollution, and other interacting stressors may reduce tree vigor and/or impair defenses, increasing vulnerability to attack (Eatough Jones et al. 2004; Bentz et al. 2010; Weed et al. 2013; Kolb et al. 2016; Millar & Rundel 2016a). Conversely, insect attack can increase tree vulnerability to drought stress and associated mortality (Kolb et al. 2016).

Both increased temperatures and greater drought stress have been identified as contributing factors in the large-scale mountain pine beetle outbreaks that have caused widespread mortality in lodgepole and whitebark pines in subalpine forests of the northern Rockies (Logan et al. 2010; Tomback & Achuff 2010). Although bark beetle mortality in California has historically been low compared to other areas of the western U.S., an outbreak between 2007 and 2010 resulted in whitebark pine mortality rates of 18.2% on Mt. Shasta, and rates up to 70% in the central Sierra Nevada (Millar et al. 2012). The highest mortality rates occurred in young, dense, and/or even-aged stands on warmer, drier sites relative to the species' range. Mortality rates were also higher on low-elevation sites and on north-facing slopes, possibly due to the fact that whitebark pine hosts are more abundant at lower elevations on northerly aspects (Millar et al. 2012). Within California, patterns of mortality suggest that increased beetle outbreaks are likely occurring in a north-to-south wave (Millar et al. 2012). This may be because of greater vulnerability to insect-related mortality in trees affected by the expansion of white pine blister rust, which is spread more readily in the cool, wet climate of northern California (Millar et al. 2012). Over the coming century, upslope shifts in the elevation of large-scale mountain pine beetle outbreaks are expected to occur (Logan & Powell 2001), with vulnerability increasing first at the lower edge of subalpine forests (Millar et al. 2012).

Regional Insect 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	• More severe insect outbreaks are likely as temperatures increase and periods of drought become more frequent (Raffa et al. 2008; Bentz et al. 2010; Kolb et al. 2016)

Regional Insect Trends	
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)	• 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) • By 2027, up to 58% of current whitebark pine basal area could be lost across the entire species' range due to the combined impacts of mountain pine beetles and white pine blister rust; areas of significant loss (more than 25% of basal area within a single watershed) could occur in the southern Cascades, but the greatest losses are expected to occur outside of northern California (Krist et al. 2014)

Summary of Potential Impacts on Habitat *(see text for citations)*
• High tree mortality rates during large-scale insect outbreaks, particularly in stands affected by mountain pine beetles • Increased risk of outbreaks and higher rates of tree mortality where stands are young, dense, and/or even-aged, as well as in areas where interacting stressors (e.g., drought, disease, air pollution) reduce tree vigor and impair defenses

Disease

White pine blister rust (caused by the invasive fungus *Cronartium ribicola*) is the most significant threat to high-elevation forests (Millar & Rundel 2016a), primarily impacting whitebark pine and foxtail pine in northwestern California subalpine forests (Kauffmann et al. 2014a, 2014b). The disease has been spreading across the western U.S. since its introduction on nursery stock in the early 1900s (McDonald & Hoff 2001). White pine blister rust affects all five-needled pine species, which includes sugar pine (*P. lambertina*), western white pine, foxtail pine, and whitebark pine (Tomback & Achuff 2010; Maloney 2011; Dunlap 2012; Krist et al. 2014).

However, high-elevation pines within the western U.S. have been increasingly affected by white pine blister rust over the past several decades, with extensive landscape-scale mortality occurring in whitebark pine stands across large areas of the northern Rocky Mountains (Tomback & Achuff 2010). Whitebark pine mortality in northern California has been more localized (Maloney 2011; Dunlap 2012; Millar et al. 2012), but infection rates of white pine blister rust are expected to increase significantly in cooler, wetter parts of the state such as northern California (Millar et al. 2012). Areas with abundant moisture are more suitable for *C. ribicola* spore dispersal and life cycle completion, as well as for currents and gooseberries (*Ribes* spp.) that act as alternate host species (Geils et al. 2010; Millar et al. 2012).

Whitebark pine is particularly vulnerable to population declines as a result of interactions between white pine blister rust infection and other stressors, such as drought, insect outbreaks, and severe fire (Logan & Powell 2001; Six & Adams 2007; Tomback & Achuff 2010; Schwartz et al. 2015). For instance, trees infected by white pine blister rust are more likely to be attacked by bark beetles, particularly when they are also stressed by drought (Six & Adams 2007). Although all species affected by the disease have shown some evidence of resistance to white pine blister rust (Schoettle & Sniezko 2007), low reproductive rates in whitebark pine reduce the likelihood that inherited resistance will be adequate to prevent stand extirpation (Kauffmann et al. 2014b). Where high rates of mortality occur, particularly in keystone species, disease-related shifts in species composition can significantly alter the structure and functioning of subalpine ecosystems (Millar & Rundel 2016a).

Another concern for northern California subalpine forests is dwarf mistletoe (*Arceuthobium* spp.), a parasitic flowering plant that reduces available resources by using water and photosynthate produced by the host tree (Hawksworth & Wiens 1996; Kliejunas 2011). Dwarf mistletoe can result in reduced seed and cone production, branch death, and mortality in high-elevation pines (Hawksworth & Wiens 1996; Mathiasen & Daugherty 2010). Additionally, infected trees experience higher mortality rates due to drought stress and beetle attack (Kliejunas 2011). In general, rates of disease increase as host density increases, so enhanced forest productivity in subalpine zones under future climate conditions may support the expansion of diseases such as dwarf mistletoe and white pine blister rust (Vuln. Assessment Workshop, pers. comm., 2017).

Regional Disease Trends	

Historical & current trends:
• Expansion of white pine blister rust across the western U.S. since the early 1900s (McDonald & Hoff 2001)
	Projected future trends:
	• Changes in patterns of disease prevalence and/or severity, depending on site conditions and limiting factors of the disease (Kliejunas 2011)
	  o Range expansion in white pine blister rust as temperatures increase, including dwarf mistletoe and white pine blister rust
	  o However, drier conditions may limit the spread of fungal disease such as white pine blister rust

Summary of Potential Impacts on Habitat *(see text for citations)*	
• Increased tree mortality, especially during periods of drought and/or in response to introduced diseases such as white pine blister rust • Reduced seed and cone production in trees infected with dwarf mistletoe • Population declines in whitebark pine due to white pine blister rust in combination with other stressors (e.g., drought, insect outbreaks) and life history traits that increase vulnerability (e.g., low reproductive rates) • Shifts in species composition and forest structure following the loss of keystone species such as whitebark pine	

Wildfire

Fire occurs relatively infrequently in subalpine forests, due in large part to low fuel availability and a short fire season limited by snowpack (Bekker & Taylor 2001; Skinner 2003; Skinner et al. 2006; Fites-Kaufmann et al. 2007). Because forest productivity is relatively low at high elevations, fuel accumulates slowly and is primarily limited to compact surface fuels that promote low-intensity surface fires that spread slowly (Skinner et al. 2006). This is the case despite frequent lightning ignitions in the southern Cascades (Skinner & Taylor 2006; van Wagtendonk & Cayan 2008), which may ignite individual trees but rarely spreads into more extensive fires (Skinner 2003; Millar & Rundel 2016a). More intense fires can occur in localized areas of higher fuel availability, generally during periods of extreme fire weather (Agee 1996; Skinner et al. 2006).

Fire return intervals are significantly longer in subalpine forests compared to those in lower-elevation forest types (Agee 1996; Bekker & Taylor 2001; Skinner et al. 2006; Miller et al. 2009, 2012; Safford & Van de Water 2014). Across the state, average fire return intervals in subalpine forests before Euro-American settlement (mid-1800s) were between 100 and 420 years (mean interval was 133 years), although fire return intervals were shorter in curl-leaf mountain mahogany (mean of 52 years), western white pine (mean of 50 years), and lodgepole pine (mean of 37 years) forests (Bekker & Taylor 2001; Van de Water & Safford 2011; Safford & Van de Water 2014). Periodic cultural burning of subalpine forests likely occurred at the upper treeline in order to increase the productivity and predictability of valued plant and animal resources (Anderson 2005; Turner et al. 2011). Given the long fire return intervals typical of subalpine forests, fire suppression over the past century has not had a significant impact at high elevations, and fire frequency remains within the historical range of variability (Safford & Van de Water 2014; Steel et al. 2015).

Most tree species in subalpine forests have thin bark and are easily injured or killed by moderate- to high-intensity wildfire (Agee 1996; Skinner et al. 2006). Lodgepole pine is a notable exception; this species has both serotinous and semi-serotinous cones, and regenerates rapidly in canopy gaps created by fire (Logan et al. 2010). Because most subalpine species are so sensitive to fire, even fires burning at low to moderate intensity can result in high levels of tree mortality (Taylor & Skinner 1998, 2003), particularly where stand densities are high (Millar & Rundel 2016a). In the southern Cascades, severe fires appear to be highly correlated with spring and summer precipitation, which is likely related to increased fuel availability (Miller et al. 2009).

Post-fire recovery in subalpine forests is slow due to the low growth rates typical of high-elevation sites with harsh environmental conditions (Fites-Kaufmann et al. 2007). At the lower limit of the subalpine zone, the dense canopies and frequent surface fires characteristic of forests dominated by true firs limit the establishment of subalpine species (Millar & Rundel 2016a). Following high-intensity fire at lower-elevation sites, subalpine species may be able to advance downslope into canopy openings at the edge of the fir forest; however, they are eventually pushed back again through forest succession back to fir (Millar & Rundel 2016a). Over the coming century, more favorable high-elevation growing conditions may increase

encroachment of species from lower elevations (Schwartz et al. 2015), which may allow fires from true fir forests to be vectored into subalpine forests (Vuln. Assessment Workshop, pers. comm., 2017). Enhanced forest productivity may also increase fuel availability (Schwartz et al. 2015; Steel et al. 2015), while warmer temperatures and reduced snowpack are likely to increase the length of the fire season by causing fuels to dry out sooner (Schwartz et al. 2015; Westerling 2016). As a result, wildfires at high elevations may increase in size, frequency, and intensity over the coming century, likely resulting in greater mortality for existing subalpine trees and making space for further expansion of lower-elevation species (Schwartz et al. 2015).

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):
  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)
  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
	• 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)
	• 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 (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 tree mortality during moderate- and high-intensity fires, especially where stand	

Regional Wildfire Trends

densities are high
• *Short-term (~2-year)*:
    ○ Slow recovery due to the harsh conditions typical of subalpine sites
    ○ Possible shifts in habitat distribution at the lower ecotone due to recruitment of subalpine trees in canopy openings and/or upslope expansion of fir following severe fire
• *Long-term*:
    ○ Encroachment of firs into subalpine zones due to increasing mortality of high-elevation species following fire

Sensitivity and current exposure to non-climate stressors

Regional experts evaluated subalpine forests as having low-moderate sensitivity to non-climate stressors (high confidence in evaluation), with an overall low current exposure to these stressors within the study region (high confidence). Non-climate stressors are relatively rare and have little impact compared to other habitat types in the assessment area. The key non-climate stressor that affects subalpine forests is livestock grazing.[7]

Livestock grazing

Historically, unregulated grazing had a significant effect on high-elevation vegetation, and by the mid-1800s, few ungrazed areas remained (Millar & Rundel 2016a). Although the majority of grazing took place within high-elevation meadows, livestock movement degraded subalpine forests, contributing to soil compaction and erosion, the introduction of invasive species, and loss of biodiversity in these fragile, slow-growing systems (Millar & Rundel 2016a). Grazing was eliminated in National Parks in the early- to mid-1900s, and U.S. Forest Service grazing allotments continue to be reduced across the state (Millar & Rundel 2016a).

Within northern California, grazing in subalpine zones still occurs in the Marble Mountains and southern Cascades (Vuln. Assessment Workshop, pers. comm., 2017). Livestock grazing and other disturbances at high elevations are likely to increase the risk of invasive species establishment as the climate becomes more favorable for these species (Rundel & Keeley 2016), which were historically limited by harsh environmental conditions rather than lack of opportunities for plant dispersal (Körner 2003; Pauchard et al. 2009).

Adaptive Capacity

Subalpine forest habitats were evaluated by regional experts as having low-moderate overall adaptive capacity (high confidence in evaluation).

---

[7] 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 air pollution, invasive plants, historic timber harvest, trails, and fire suppression. Invasive plants and air pollution, in particular, may become a greater threat as climate changes exacerbate their impact on subalpine vegetation (Vuln. Assessment Reviewer, pers. comm., 2018).

Habitat extent, integrity, continuity, and permeability

Regional experts evaluated subalpine forests as having a high geographic extent (high confidence in evaluation), high structural and functional integrity (high confidence), and moderate-high continuity (high confidence). Landscape permeability for subalpine forests was evaluated as low (high confidence). Geologic features and genetic isolation were identified as the primary barriers to habitat continuity and dispersal across the study region.[8]

Subalpine forests in northern California are restricted to isolated mountaintops within the southern Cascades (including Mount Shasta), the Klamath Mountains, and small areas in the South Yolla Bolly Mountains on the North Coast (Sawyer 2007; Kauffmann et al. 2014a, 2014b). The region contains just over 20% of the state's total extent of subalpine forest, with most of that area found within the Klamath Mountains (Millar & Rundel 2016a). At the lower edge of their distribution, subalpine forests are limited by evaporative stress and severe fires associated with true fir forests (Millar & Rundel 2016a). Only Mount Shasta is high enough to contain significant area of alpine habitat, and subalpine forests reach the highest elevations on other peaks in northern California (Sawyer 2007; Kauffmann et al. 2014a, 2014b). As a result, upslope shifts in habitat distribution are strongly limited by elevational availability within the region (Vuln. Assessment Workshop, pers. comm., 2017).

Subalpine forest patches are generally disconnected, occurring as "sky islands" that limit genetic exchange between small, isolated populations (Eckert et al. 2010; Kauffmann et al. 2014a, 2014b), which are at increased risk of extirpation following disturbance events that cause high rates of mortality (Schierenbeck 2017). Life history characteristics in subalpine trees, including very slow growth rates and long-lived species, also limit their ability to track climatic changes across the landscape (Fites-Kaufmann et al. 2007). In general, encroachment of lower-elevation forest species (e.g., white fir, Shasta red fir) into subalpine zones is likely to result in significant range contractions for this habitat type (Kauffmann et al. 2014a, 2014b). However, there is some evidence of expansion in high-elevation pines, including increased whitebark pine recruitment where lingering snowpack is decreasing within high-elevation swales on Mount Shasta (Kauffmann et al. 2014a), and downslope expansion of foxtail pine in the Klamath Mountains (Eckert & Eckert 2007). Changes in species distribution at the upper ecotone (treeline) have occurred during past changes in climate, suggesting that subalpine habitats are likely to shift in response to warming and increased water stress (Taylor 1995; Millar et al. 2004). However, no significant distributional shifts have yet been observed within subalpine habitats in other areas of the state (Dolanc et al. 2013a; Millar & Rundel 2016a), and lack of habitat continuity prevents northward shifts into areas that may remain climatically suitable (Ledig et al. 2012).

---

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

Habitat diversity

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

Forest structure in subalpine forests can range from closed canopy stands dominated by mountain hemlock to more open woodlands comprised primarily of high-elevation pines (Sawyer 2007). As elevation increases, harsh conditions (e.g., shallow soils, low temperatures, deep snowpack) limit tree growth and regeneration to a greater degree (Dolanc et al. 2013b; Millar & Rundel 2016a). Northern California subalpine stands are typically dominated by 1–3 conifers, but localized areas of enriched subalpine stands (up to 10 conifer species) have been documented in the Trinity Mountains (Cheng 2004; Sawyer 2007). Across the region, levels of endemism are particularly high on sky islands; however, the limited size of functional groups within subalpine forests increases climate change vulnerability for these species (Vuln. Assessment Workshop, pers. comm., 2017).

Most closed canopy stands are dominated by mountain hemlock and have a sparse understory of little prince's pine (*Chimaphila menziesii*), pipsissewa (*C. umbellata*), or white-veined wintergreen (*Pyrola picta*; Sawyer 2007). They occur primarily on protected sites and north-facing slopes; on drier slopes, red fir mixes in and can become dominant (Sawyer 2007). Mountain hemlock can also co-occur with lodgepole pine, western white pine and, in the extreme northwestern corner of the state, with noble fir (*Abies procera*; CNPS 2018). Whitebark pine dominates stands in exposed areas and on warmer, drier sites in the southern Cascades and, to a lesser extent, in the Klamath Mountains (Griffin & Critchfield 1972; Kauffmann et al. 2014a, 2014b). Isolated stands of Pacific silver fir (*Abies amabilis*), subalpine fir (*Abies lasiocarpa*), Engelmann spruce (*Picea engelmannii*), Brewer spruce (*P. breweriana*), and yellow cedar (*Callitropsis mootkatensis*) also occur in the Klamath region (Skinner et al. 2006; Sawyer 2007; Millar & Rundel 2016a; Thorne et al. 2016). Frequently, species typical of north-facing slopes at lower elevations occur on southerly aspects at higher elevations (Fites-Kaufmann et al. 2007; Vuln. Assessment Workshop, pers. comm., 2017).

On serpentine soils, which are particularly common in the Trinity Mountains (Sawyer 2007), subalpine forests are more open and comprised of mixed high-elevation pines, including foxtail pine, western white pine, and Jeffrey pine, with an understory of Sasketoon serviceberry (*Amelanchier alnifolia*), oceanspray (*Holodiscus discolor*), and huckleberry oak (*Quercus vaccinifolia*; Skinner et al. 2006; Eckert et al. 2010; Millar & Rundel 2016a). Foxtail pine is an endemic species with disjunct populations in northwestern California and the southern Sierra Nevada (Griffin & Critchfield 1972; CNPS 2019). High habitat heterogeneity allows the persistence of foxtail pine populations in subalpine zones (Eckert 2006; Eckert et al. 2010), although the small, isolated populations in the Klamath Mountains may be particularly sensitive to climate change (Eckert 2006; Eckert & Eckert 2007; Eckert et al. 2010).

Subalpine forests provide important wildlife habitat for many species, including sensitive or rare mammals such as American marten (*Martes caurina*) and snowshoe hare (*Lepus*

americanus; Millar & Rundel 2016a). Many birds use subalpine forests on a seasonal basis, including mountain bluebird (*Sialia currucoides*), red crossbill (*Loxia curvirostra*), Cassin's finch (*Carpodacus cassinii*), sooty grouse (*Dendragapus fuliginosus*), Williamson's sapsucker (*Sphyrapicus thyroideus*), and black-backed woodpecker (*Picoides arcticus*; Millar & Rundel 2016a). Several birds and small mammals play an important role in seed dispersal for subalpine tree species (Logan & Powell 2001; Fites-Kaufmann et al. 2007; Keane et al. 2012). Whitebark pine and Clark's nutcracker (*Nucifraga columbiana*), in particular, are keystone mutualists, and the loss of whitebark pine within the ecosystem would have cascading relationships on other wildlife species (e.g., bears, squirrels) that depend on its fleshy, nutritious seed (Logan & Powell 2001; Logan et al. 2010).

Resistance and recovery

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

The ability of subalpine species to resist and/or recover from climate-related stressors and changes in disturbance regimes varies widely depending on site requirements, dispersal ability, and establishment (Schierenbeck 2017; Smithers et al. 2017), as well as both landscape- and local-scale variation in topography (Macias-Fauria & Johnson 2013). Although most subalpine species evolved under harsh conditions, slow growth and low reproductive rates reduce the ability of these species to pass on adaptive traits rapidly enough to prevent population extirpation in the face of changing climate conditions (Ledig et al. 2012) and white pine blister rust expansion (Schoettle & Sniezko 2007; Kauffmann et al. 2014b). However, the presence of relict species in the region suggests that some high-elevation conifers may be able to survive in microclimates created by high topographic diversity (Schierenbeck 2017). Differing life history characteristics (e.g., reproductive potential, dispersal ability) are likely to lead to changes in species dominance at high elevations, as some species experience successful recruitment and/or range shifts (upslope or downslope) while others do not (Millar et al. 2015; Smithers et al. 2017).

Several life history characteristics of whitebark pine increase the vulnerability of this species to extirpation following large-scale disturbances (Logan & Powell 2001; Logan et al. 2010). Whitebark pine places high investment in few seeds, and is dependent on the Clark's nutcracker to disperse seeds in canopy openings caused by fire and other disturbances (Logan et al. 2010). Disruption of this mutualistic relationship, either because of reduced cone density or loss of the nutcracker, would result in reduced tree recruitment and population declines (Logan et al. 2010). Whitebark pine also has relatively weak chemical defenses compared to lodgepole pine, making it more vulnerable to mountain pine beetle attack (Logan & Powell 2001). Where it occurs, white pine blister rust is a significant stressor, and heightens vulnerability to climatic stress and beetle-related mortality (Six & Adams 2007; Tomback & Achuff 2010). Differences in growth between trees that survive and those that die during large-scale insect outbreaks suggest that there are adaptive genetic differences among stands, with surviving trees

exhibiting increased growth rates in warm, dry conditions and periods of high interannual variability compared to trees that died (Millar et al. 2012).

Warmer temperatures, longer snow-free periods, and more frequent disturbances such as wildfire are also likely to reduce the abiotic resistance of subalpine forests to invasion, potentially allowing upslope expansion of non-native species that were historically limited by harsh environmental conditions (Pauchard et al. 2009; Stevens & Latimer 2015; Rundel & Keeley 2016; Lembrechts et al. 2018). The establishment of invasive species could represent a significant threat to subalpine biodiversity and ecosystem processes (Rundel & Keeley 2016; Lembrechts et al. 2018).

Management potential

Public and societal value
Regional experts evaluated subalpine forests as having low-moderate public and societal value (high confidence in evaluation).

Subalpine forests are valued for their beauty and the unique recreational experiences they offer (e.g., mountaineering, skiing, hiking), and the rarity of this habitat type further increases their value (Vuln. Assessment Workshop, pers. comm., 2017). Subalpine forests are primarily located within protected areas (e.g., federal Wilderness Areas, National Forests), which minimizes potential human impacts (Millar & Rundel 2016a). Except for Mt. Shasta, most subalpine forests in northern California are inaccessible by car (Vuln. Assessment Workshop, pers. comm., 2017).

Management capacity and ability to alleviate impacts[9]
Regional experts evaluated the potential for reducing climate impacts on subalpine forests through management as low (high confidence in evaluation). Regional experts noted that ski resort development used to represent a conflict for land use, but is no longer and there is little direct economic value for resource extraction (Vuln. Assessment Workshop, pers. comm., 2017).

Management actions to reduce the impacts of climate on subalpine forests are somewhat limited, partly because of the inaccessibility and relative rarity of this forest type in northern California (Vuln. Assessment Workshop, pers. comm., 2017). Because most subalpine forests are already at the highest elevations in the region, there is little ability to promote upslope species migration into alpine areas (Vuln. Assessment Workshop, pers. comm., 2017). Unlike most lower-elevation forest types, reintroducing fire onto the landscape is less of a concern (Millar & Rundel 2016a). However, thinning stands that become dense due to increased forest productivity could increase resistance to insects and disease (Millar & Rundel 2016b).

The primary focus of management in subalpine forests is likely to be breeding and planting trees resistant to white pine blister rust (Schoettle & Sniezko 2007; Keane et al. 2012; Vuln.

---

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

Assessment Workshop, pers. comm., 2017). However, more extensive insect outbreaks and large-scale tree mortality, particularly in whitebark pine stands, could undermine these efforts (Logan & Powell 2001; Hicke et al. 2006).

While invasive species are not currently considered a significant stressor for subalpine forests in California (Millar & Rundel 2016; Rundel & Keeley 2016; Vuln. Assessment Reviewer, pers. comm., 2018), changing climate conditions and increased human disturbance at high elevations are likely to increase the risk of establishment by non-native plant species from lower elevations (Pauchard et al. 2009; Stevens & Latimer 2015). Creating programs designed to identify potential invasions through early detection and monitoring are likely to reduce this risk and allow managers to respond to threats immediately (Stevens & Latimer 2015; Millar & Rundel 2016b; Rundel & Keeley 2016).

Ecosystem services
Subalpine forests play a critical role in water storage by capturing snow and shading snowpack, which allows it to persist later into the summer, when it recharges aquifers and is delivered to lower-elevation habitats (Logan et al. 2010; Millar & Rundel 2016a). This habitat type also provides a wide variety of other ecosystem services typically associated with forested habitats, including:

- Provisioning of genetic resources and the potential for natural medicines;
- Regulation of water purification;
- Support of soil formation/retention and carbon sequestration;
- Wildlife habitat; and
- Cultural/tribal uses for spiritual/religious purposes, knowledge systems, educational values, aesthetic values, social relations, sense of place, cultural heritage, inspiration, and recreation (Turner et al. 2011; Keane et al. 2012; Vuln. Assessment Workshop, pers. comm., 2017).

Recommended Citation

Hilberg LE, Reynier WA, Kershner JM. 2019. Subalpine Forests: Northern California Climate Change Vulnerability Assessment Synthesis. Version 1.0. EcoAdapt, Bainbridge Island, WA.

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

Literature Cited

Agee JK. 1996. Fire ecology of Pacific Northwest forests. Island Press, Washington, D.C.

Anderson MK. 2005. Tending the wild: Native American knowledge and the management of California's natural resources. University of California Press, Berkeley, CA.

Bekker MF, Taylor AH. 2001. Gradient analysis of fire regimes in montane forests of the southern Cascade Range, Thousand Lakes Wilderness, California, USA. Plant Ecology **155**:15–28.

Bentz BJ, Regniere J, Fettig CJ, Hansen EM, Hayes JL, Hicke JA, Kelsey RG, Negron JF, Seybold SJ. 2010. Climate change and bark beetles of the western United States and Canada: direct and indirect effects. BioScience **60**:602–613.

Bunn AG, Graumlich LJ, Urban DL. 2005a. Trends in twentieth-century tree growth at high elevations in the Sierra Nevada and White Mountains, USA. The Holocene **15**:481–488.

Bunn AG, Waggoner LA, Graumlich LJ. 2005b. Topographic mediation of growth in high elevation foxtail pine (*Pinus balfouriana* Grev. et Balf.) forests in the Sierra Nevada, USA. Global Ecology and Biogeography **14**:103–114.

Bytnerowicz A, Fenn M, McNulty S, Yuan F, Pourmokhtarian A, Driscoll C, Meixner T. 2013. Interactive effects of air pollution and climate change on forest ecosystems in the United States: current understanding and future scenarios. Developments in Environmental Science **13**:333–369.

Cheng S. 2004. Forest Service research natural areas in California. Gen. Tech. Rep. PSW-GTR-188. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA.

Chmura DJ, Anderson PD, Howe GT, Harrington CA, Halofsky JE, Peterson DL, Shaw DC, Brad St. Clair J. 2011. Forest responses to climate change in the northwestern United States: ecophysiological foundations for adaptive management. Forest Ecology and Management **261**:1121–1142.

CNPS. 2019. A manual of California vegetation, online edition. California Native Plant Society Press. Available from http://www.cnps.org/cnps/vegetation/ (accessed January 2, 2019).

Cook BI, Ault TR, Smerdon JE. 2015. Unprecedented 21st century drought risk in the American Southwest and Central Plains. Science Advances **1**:e1400082.

Cook BI, Smerdon JE, Seager R, Coats S. 2014. Global warming and 21st century drying. Climate Dynamics **43**:2607–2627.

Das AJ, Stephenson NL, Davis KP. 2016. Why do trees die? Characterizing the drivers of background tree mortality. Ecology **97**:2616–2627.

Das AJ, Stephenson NL, Flint A, Das T, Mantgem PJ van. 2013. Climatic correlates of tree mortality in water- and energy-limited forests. PLoS ONE **8**:e69917.

Diffenbaugh NS, Swain DL, Touma D. 2015. Anthropogenic warming has increased drought risk in California. Proceedings of the National Academy of Sciences **112**:3931–3936.

Dolanc CR, Thorne JH, Safford HD. 2013a. Widespread shifts in the demographic structure of subalpine forests in the Sierra Nevada, California, 1934 to 2007. Global Ecology and Biogeography **22**:264–276.

Dolanc CR, Westfall RD, Safford HD, Thorne JH, Schwartz MW. 2013b. Growth–climate relationships for six subalpine tree species in a Mediterranean climate. Canadian Journal of Forest Research **43**:1114–1126.

Dunlap JM. 2012. Variability in and environmental correlates to white pine blister rust incidence in five California white pine species. Northwest Science **86**:248–263.

Eatough Jones M, Paine TD, Fenn ME, Poth MA. 2004. Influence of ozone and nitrogen deposition on bark beetle activity under drought conditions. Forest Ecology and Management **200**:67–76.

Eckert AJ. 2006. Influence of substrate type and microsite availability on the persistence of foxtail pine (*Pinus balfouriana*, Pinaceae) in the Klamath Mountains, California. American Journal of Botany **93**:1615–1624.

Eckert AJ, Eckert ML. 2007. Environmental and ecological effects on size class distributions of foxtail pine (*Pinus balfouriana*, Pinaceae) in the Klamath Mountains, California. Madroño **54**:117–125.

Eckert AJ, Eckert ML, Hall BD. 2010. Effects of historical demography and ecological context on spatial patterns of genetic diversity within foxtail pine (*Pinus balfouriana*; Pinaceae) stands located in the Klamath Mountains, California. American Journal of Botany **97**:650–659.

Fites-Kaufmann JA, Rundel P, Stephenson N. 2007. Montane and subalpine vegetation of the Sierra Nevada and Cascade ranges. Pages 456–501 in M. Barbour, T. Keeler-Wolf, and A. A.

Schoenherr, editors. Terrestrial vegetation of California, 3rd edition. University of California Press, Berkeley, CA.

Flint LE, Flint AL. 2014. California Basin Characterization Model: a dataset of historical and future hydrologic response to climate change (Ver. 1.1, May 2017). U.S. Geological Survey Data Release. Available from https://doi.org/10.5066/F76T0JPB.

Flint LE, Flint AL, Thorne JH, Boynton R. 2013. Fine-scale hydrologic modeling for regional landscape applications: the California Basin Characterization Model development and performance. Ecological Processes **2**:25.

Geils BW, Hummer KE, Hunt RS. 2010. White pines, *Ribes*, and blister rust: a review and synthesis. Forest Pathology **40**:147–185.

Gergel DR, Nijssen B, Abatzoglou JT, Lettenmaier DP, Stumbaugh MR. 2017. Effects of climate change on snowpack and fire potential in the western USA. Climatic Change **141**:287–299.

Griffin D, Anchukaitis KJ. 2014. How unusual is the 2012–2014 California drought? Geophysical Research Letters **41**:9017–9023.

Griffin JR, Critchfield WB. 1972. The distribution of forest trees in California. Res. Pap. PSW-RP-82. U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, CA.

Grulke NE. 2003. The physiological basis of ozone injury assessment attributes in Sierran conifers. Pages 55–81 in A. Bytnerowicz, M. J. Arbaugh, and R. Alonso, editors. Assessment of ozone distribution and its effects on Sierra Nevada ecosystems. Elsevier Publishers, The Hague, Netherlands.

Hamlet AF, Mote PW, Clark MP, Lettenmaier DP. 2005. Effects of temperature and precipitation variability on snowpack trends in the western United States. Journal of Climate **18**:4545–4561.

Hawksworth FG, Wiens D. 1996. Dwarf mistletoes: biology, pathology, and systematics. Agriculture Handbook 709. U.S. Department of Agriculture, Forest Service, Washington, D.C.

Hayhoe K et al. 2004. Emissions pathways, climate change, and impacts on California. Proceedings of the National Academy of Sciences **101**:12422–12427.

Hicke JA, Logan JA, Powell J, Ojima DS. 2006. Changing temperatures influence suitability for modeled mountain pine beetle (*Dendroctonus ponderosae*) outbreaks in the western United States. Journal of Geophysical Research: Biogeosciences **111**:G02019.

Hicke JA, Meddens AJH, Kolden CA. 2016. Recent tree mortality in the western United States from bark beetles and forest fires. Forest Science **62**:141–153.

Kauffmann M, Taylor S, Sikes K, Evans J. 2014a. Shasta-Trinity National Forest: whitebark pine pilot fieldwork report. Unpublished report. California Native Plant Society Vegetation Program, Sacramento, CA. Available from http://pacslope-conifers.com/conifers/pine/wbp/CNPS-Reports/index.html (accessed May 2, 2018).

Kauffmann M, Taylor S, Sikes K, Evans J. 2014b. Klamath National Forest: whitebark pine pilot fieldwork report. Unpublished report. California Native Plant Society Vegetation Program, Sacramento, CA. Available from http://pacslope-conifers.com/conifers/pine/wbp/CNPS-Reports/index.html (accessed May 2, 2018).

Keane RE et al. 2012. A range-wide restoration strategy for whitebark pine (*Pinus albicaulis*). U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Keyser A, Westerling AL. 2017. Climate drives inter-annual variability in probability of high severity fire occurrence in the western United States. Environmental Research Letters **12**:065003.

Kliejunas JT. 2011. A risk assessment of climate change and the impact of forest diseases on forest ecosystems in the western United States and Canada. Gen. Tech. Rep. PSW-GTR-236. U.S. Department of Agriculture, Forest Service, Pacific Southwest Station, Albany, CA.

Knowles N, Dettinger MD, Cayan DR. 2006. Trends in snowfall versus rainfall in the western United States. Journal of Climate **19**:4545–4559.

Kolb TE, Fettig CJ, Ayres MP, Bentz BJ, Hicke JA, Mathiasen R, Stewart JE, Weed AS. 2016. Observed and anticipated impacts of drought on forest insects and diseases in the United States. Forest Ecology and Management **380**:321–334.

Körner C. 2003. Alpine plant life: functional plant ecology of high mountain ecosystems. Springer Verlag, Berlin.

Krist FJ, Ellenwood JR, Woods ME, McMahan AJ, Cowardin JP, Ryerson DE, Sapio FJ, Sweifler MO, Romero SA. 2014. 2013–2027 National Insect and Disease Forest Risk Assessment. FHTET 14-01. U.S. Department of Agriculture, Forest Service, Forest Health Technology Enterprise Team, Fort Collins, CO.

Law BE, Waring RH. 2015. Carbon implications of current and future effects of drought, fire and management on Pacific Northwest forests. Forest Ecology and Management **355**:4–14.

Ledig FT, Rehfeldt GE, Jaquish B. 2012. Projections of suitable habitat under climate change scenarios: implications for trans-boundary assisted colonization. American Journal of Botany **99**:1217–1230.

Lembrechts JJ, Lenoir J, Nuñez MA, Pauchard A, Geron C, Bussé G, Milbau A, Nijs I. 2018. Microclimate variability in alpine ecosystems as stepping stones for non-native plant establishment above their current elevational limit. Ecography **41**:900–909.

Lenihan JM, Bachelet D, Neilson RP, Drapek R. 2008. Response of vegetation distribution, ecosystem productivity, and fire to climate change scenarios for California. Climatic Change **87**:215–230.

Lenihan JM, Drapek R, Bachelet D, Neilson RP. 2003. Climate change effects on vegetation distribution, carbon, and fire in California. Ecological Applications **13**:1667–1681.

Logan J, Powell J. 2001. Ghost forests, global warming, and the mountain pine beetle (Coleoptera : Scolytidae). American Entomologist **47**:160–173.

Logan JA, Macfarlane WW, Willcox L. 2010. Whitebark pine vulnerability to climate-driven mountain pine beetle disturbance in the Greater Yellowstone Ecosystem. Ecological Applications **20**:895–902.

Lutz JA, van Wagtendonk JW, Franklin JF. 2010. Climatic water deficit, tree species ranges, and climate change in Yosemite National Park. Journal of Biogeography **37**:936–950.

Macias-Fauria M, Johnson EA. 2013. Warming-induced upslope advance of subalpine forest is severely limited by geomorphic processes. Proceedings of the National Academy of Sciences **110**:8117–8122.

Maloney PE. 2011. Incidence and distribution of white pine blister rust in the high-elevation forests of California. Forest Pathology **41**:308–316.

Mann ML, Batllori E, Moritz MA, Waller EK, Berck P, Flint AL, Flint LE, Dolfi E. 2016. Incorporating anthropogenic influences into fire probability models: effects of human activity and climate change on fire activity in California. PLoS ONE **11**:e0153589.

Mathiasen RL, Daugherty CM. 2010. Susceptibility of Brewer spruce (*Picea breweriana*) to dwarf mistletoes (*Arceuthobium* spp., Viscaceae). Northwest Science **84**:295–301.

McDonald GI, Hoff RJ. 2001. Blister rust: an introduced plague. Pages 193–220 in D. F. Tomback, S. F. Arno, and R. E. Keane, editors. Whitebark pine communities: ecology and restoration. Island Press, Washington, D.C.

Micheli E, Dodge C, Flint L, Comendant T. 2018. Climate and natural resource analyses and planning for the North Coast Resource Partnership: a technical memorandum summarizing data products. A final report prepared by the Dwight Center for Conservation Science at Pepperwood, Santa Rosa, CA, for West Coast Watershed and the North Coast Resource Partnership.

Millar CI, Rundel PW. 2016a. Subalpine forests. Pages 579–611 in H. Mooney and E. Zavaleta, editors. Ecosystems of California. University of California Press, Oakland, CA.

Millar CI, Rundel PW. 2016b. Subalpine forests. Pages 43–44 in H. Mooney and E. Zavaleta, editors. Ecosystems of California: threats and responses. Supplement for decision-making. Woods Institute for the Environment, Stanford University, Stanford, CA.

Millar CI, Stephenson NL. 2015. Temperate forest health in an era of emerging megadisturbance. Science **349**:823–826.

Millar CI, Westfall RD, Delany DL, Bokach MJ, Flint AL, Flint LE. 2012. Forest mortality in high-elevation whitebark pine (*Pinus albicaulis*) forests of eastern California, USA: influence of environmental context, bark beetles, climatic water deficit, and warming. Canadian Journal of Forest Research **42**:749–765.

Millar CI, Westfall RD, Delany DL, Flint AL, Flint LE. 2015. Recruitment patterns and growth of high-elevation pines in response to climatic variability (1883–2013), in the western Great Basin, USA. Canadian Journal of Forest Research **45**:1299–1312.

Millar CI, Westfall RD, Delany DL, King JC, Graumlich LJ. 2004. Response of subalpine conifers in the Sierra Nevada, California, U.S.A., to 20th-century warming and decadal climate variability. Arctic, Antarctic, and Alpine Research **36**:181–200.

Miller JD, Safford HD, Crimmins M, Thode AE. 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada, USA. Ecosystems **12**:16–32.

Miller JD, Skinner CN, Safford HD, Knapp EE, Ramirez CM. 2012. Trends and causes of severity, size, and number of fires in northwestern California, USA. Ecological Applications **22**:184–203.

Mote PW. 2006. Climate-driven variability and trends in mountain snowpack in western North America. Journal of Climate **19**:6209–6220.

Mote PW, Hamlet AF, Clark MP, Lettenmaier DP. 2005. Declining mountain snowpack in western North America. Bulletin of the American Meteorological Society **86**:39–49.

Parks SA, Miller C, Abatzoglou JT, Holsinger LM, Parisien M-A, Dobrowski SZ. 2016. How will climate change affect wildland fire severity in the western US? Environmental Research Letters **11**:035002.

Parks SA, Miller C, Parisien M-A, Holsinger LM, Dobrowski SZ, Abatzoglou J. 2015. Wildland fire deficit and surplus in the western United States, 1984–2012. Ecosphere **6**:1–13.

Pauchard A et al. 2009. Ain't no mountain high enough: plant invasions reaching new elevations. Frontiers in Ecology and the Environment **7**:479–486.

Pierce DW, Kalansky JF, Cayan DR. 2018. Climate, drought, and sea level rise scenarios for the Fourth California Climate Assessment. California's Fourth Climate Change Assessment. Publication Number: CNRA-CEC-2018-006. California Energy Commission, Sacramento, CA.

Raffa KF, Aukema BH, Bentz BJ, Carroll AL, Hicke JA, Turner MG, Romme WH. 2008. Cross-scale drivers of natural disturbances prone to anthropogenic amplification: the dynamics of bark beetle eruptions. BioScience **58**:501–517.

Rapacciuolo G et al. 2014. Beyond a warming fingerprint: individualistic biogeographic responses to heterogeneous climate change in California. Global Change Biology **20**:2841–2855.

Rauscher SA, Pal JS, Diffenbaugh NS, Benedetti MM. 2008. Future changes in snowmelt-driven runoff timing over the western US. Geophysical Research Letters **35**:L16703.

Royce EB, Barbour MG. 2001. Mediterranean climate effects. II. Conifer growth phenology across a Sierra Nevada ecotone. American Journal of Botany **88**:919–932.

Rundel PW, Keeley JE. 2016. Dispersal limitation does not control high elevational distribution of alien plant species in the southern Sierra Nevada, California. Natural Areas Journal **36**:277–287.

Safford HD, Van de Water KM. 2014. Using fire return interval departure (FRID) analysis to map spatial and temporal changes in fire frequency on national forest lands in California. Res. Pap. PSW-RP-

266. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA.

Sawyer JO. 2007. Forests of northwestern California. Pages 253–295 in M. Barbour, T. Keeler-Wolf, and A. A. Schoenherr, editors. Terrestrial vegetation of California, 3rd edition. University of California Press, Berkeley, CA.

Schierenbeck KA. 2017. Population-level genetic variation and climate change in a biodiversity hotspot. Annals of Botany **119**:215–228.

Schoettle AW, Sniezko RA. 2007. Proactive intervention to sustain high-elevation pine ecosystems threatened by white pine blister rust. Journal of Forest Research **12**:327–336.

Schwartz MW, Butt N, Dolanc CR, Holguin A, Moritz MA, North MP, Safford HD, Stephenson NL, Thorne JH, van Mantgem PJ. 2015. Increasing elevation of fire in the Sierra Nevada and implications for forest change. Ecosphere **6**:1–10.

Six DL, Adams J. 2007. White pine blister rust severity and selection of individual whitebark pine by the mountain pine beetle (Coleoptera: Curculionidae, Scolytinae). Journal of Entomological Science **42**:345–353.

Skinner C. 2003. Fire history of upper montane and subalpine glacial basins in the Klamath Mountains of northern California. Pages 145–151 in R. C. Galley, R. C. Klinger, and N. G. Sugihara, editors. Proceedings of Fire Conference 2000: the first national congress on fire ecology, prevention, and management. Miscellaneous Publication No. 13. Tall Timbers Research Station, Tallahassee, FL.

Skinner CN, Taylor AH. 2006. Southern Cascades bioregion. Pages 195–224 in N. G. Sugihara, J. W. van Wagtendonk, J. Fites-Kaufmann, K. E. Shaffer, and A. E. Thode, editors. Fire in California's ecosystems. University of California Press, Berkeley, CA.

Skinner CN, Taylor AH, Agee JK. 2006. Klamath Mountains bioregion. Pages 170–194 in N. G. Sugihara, J. W. van Wagtendonk, J. Fites-Kaufmann, K. E. Shaffer, and A. E. Thode, editors. Fire in California's ecosystems. University of California Press, Berkeley, CA.

Smithers BV, North MP, Millar CI, Latimer AM. 2017. Leap frog in slow motion: divergent responses of tree species and life stages to climatic warming in Great Basin subalpine forests. Global Change Biology **24**:442–457.

Steel ZL, Safford HD, Viers JH. 2015. The fire frequency-severity relationship and the legacy of fire suppression in California forests. Ecosphere **6**:1–23.

Stephens SL, Martin RE, Clinton NE. 2007. Prehistoric fire area and emissions from California's forests, woodlands, shrublands, and grasslands. Forest Ecology and Management **251**:205–216.

Stephenson N. 1998. Actual evapotranspiration and deficit: biologically meaningful correlates of vegetation distribution across spatial scales. Journal of Biogeography **25**:855–870.

Stevens JT, Latimer AM. 2015. Snowpack, fire, and forest disturbance: interactions affect montane invasions by non-native shrubs. Global Change Biology **21**:2379–2393.

Stewart IT, Cayan DR, Dettinger MD. 2004. Changes in snowmelt runoff timing in western North America under a `business as usual' climate change scenario. Climatic Change **62**:217–232.

Stewart IT, Cayan DR, Dettinger MD. 2005. Changes toward earlier streamflow timing across western North America. Journal of Climate **18**:1136–1155.

Sturrock RN, Frankel SJ, Brown AV, Hennon PE, Kliejunas JT, Lewis KJ, Worrall JJ, Woods AJ. 2011. Climate change and forest diseases. Plant Pathology **60**:133–149.

Swain DL, Langenbrunner B, Neelin JD, Hall A. 2018. Increasing precipitation volatility in twenty-first-century California. Nature Climate Change **8**:427.

Syphard AD, Keeley JE, Pfaff AH, Ferschweiler K. 2017. Human presence diminishes the importance of climate in driving fire activity across the United States. Proceedings of the National Academy of Sciences **114**:13750–13755.

Taylor AH. 1995. Forest expansion and climate change in the mountain hemlock (*Tsuga mertensiana*) zone, Lassen Volcanic National Park, California, U.S.A. Arctic and Alpine Research **27**:207–216.

Taylor AH, Skinner CN. 1998. Fire history and landscape dynamics in a late-successional reserve, Klamath Mountains, California, USA. Forest Ecology and Management **111**:285–301.

Taylor AH, Skinner CN. 2003. Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. Ecological Applications **13**:704–719.

Taylor AH, Trouet V, Skinner CN, Stephens S. 2016. Socioecological transitions trigger fire regime shifts and modulate fire–climate interactions in the Sierra Nevada, USA, 1600–2015 CE. Proceedings of the National Academy of Sciences **113**:13684–13689.

Thorne JH, Boynton RM, Flint LE, Flint AL. 2015. The magnitude and spatial patterns of historical and future hydrologic change in California's watersheds. Ecosphere **6**:1–30.

Thorne JH, Boynton RM, Holguin AJ, Stewart JAE, Bjorkman J. 2016. A climate change vulnerability assessment of California's terrestrial vegetation. California Department of Fish and Wildlife, Sacramento, CA.

Thorne JH, Choe H, Boynton RM, Bjorkman J, Albright W, Nydick K, Flint AL, Flint LE, Schwartz MW. 2017. The impact of climate change uncertainty on California's vegetation and adaptation management. Ecosphere **8**:e02021.

Tomback DF, Achuff P. 2010. Blister rust and western forest biodiversity: ecology, values and outlook for white pines. Forest Pathology **40**:186–225.

Turner NJ, Deur D, Mellott CR. 2011. "Up on the mountain": ethnobotanical importance of montane sites in Pacific Coastal North America. Journal of Ethnobiology **31**:4–43.

Van de Water KM, Safford HD. 2011. A summary of fire frequency estimates for California vegetation before Euro-American settlement. Fire Ecology **7**:26–58.

van Mantgem PJ, Stephenson NL. 2007. Apparent climatically induced increase of tree mortality rates in a temperate forest. Ecology Letters **10**:909–916.

van Mantgem PJ van et al. 2009. Widespread increase of tree mortality rates in the western United States. Science **323**:521–524.

van Wagtendonk JW, Cayan DR. 2008. Temporal and spatial distribution of lightning strikes in California in relation to large-scale weather patterns. Fire Ecology **4**:34–56.

Wahl ER, Zorita E, Trouet V, Taylor AH. 2019. Jet stream dynamics, hydroclimate, and fire in California from 1600 CE to present. Proceedings of the National Academy of Sciences **116**:5393–5398.

Weed AS, Ayres MP, Hicke JA. 2013. Consequences of climate change for biotic disturbances in North American forests. Ecological Monographs **83**:441–470.

Westerling AL. 2016. Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring. Philosophical Transactions of the Royal Society B: Biological Sciences **371**:20150178.

Westerling AL. 2018. Wildfire simulations for California's Fourth Climate Change Assessment: projecting changes in extreme wildfire events with a warming climate. California's Fourth Climate Change Assessment. Publication Number: CCCA4-CEC-2018-014. California Energy Commission, Sacramento, CA.

Westerling AL, Bryant BP, Preisler HK, Holmes TP, Hidalgo HG, Das T, Shrestha SR. 2011. Climate change and growth scenarios for California wildfire. Climatic Change **109**:445–463.

Williams DW, Liebhold AM. 2002. Climate change and the outbreak ranges of two North American bark beetles. Agricultural and Forest Entomology **4**:87–99.

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),[10] 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.[11]

---

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

[11] 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

Literature Cited

EcoAdapt. 2014a. A climate change vulnerability assessment for aquatic resources in the Tongass National Forest. EcoAdapt, Bainbridge Island, WA. 124 pp.

EcoAdapt. 2014b. A climate change vulnerability assessment for resources of Nez Perce-Clearwater National Forests. Version 3.0. EcoAdapt, Bainbridge Island, WA. 398 pp.

Glick P, Stein BA, Edelson NA. 2011. Scanning the conservation horizon: A guide to climate change vulnerability assessment. National Wildlife Federation, Washington, D.C.

Gregg RM, editor. 2018. Hawaiian Islands climate vulnerability and adaptation synthesis. EcoAdapt, Bainbridge Island, WA. 284 pp.

Hutto SV, Higgason KD, Kershner JM, Reynier WA, Gregg DS. 2015. Climate change vulnerability assessment for the north-central California coast and ocean. Marine Sanctuaries Conservation Series ONMS-15-02. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Office of National Marine Sanctuaries, Silver Spring, MD. 473 pp.

Intergovernmental Panel on Climate Change (IPCC). 2007. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Pages 617–652 in M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden, and C. E. Hanson, editors. Cambridge University Press, Cambridge, UK.

Kershner JM, editor. 2014. A climate change vulnerability assessment for focal resources of the Sierra Nevada. Version 1.0. EcoAdapt, Bainbridge Island, WA. 418 pp.

Lawler J. 2010. Pacific Northwest Climate Change Vulnerability Assessment. From http://climatechangesensitivity.org

Manomet Center for Conservation Sciences and National Wildlife Federation. 2012. The vulnerabilities of fish and wildlife habitats in the Northeast to climate change. A report to the Northeastern Association of Fish and Wildlife Agencies and the North Atlantic Landscape Conservation Cooperative. Manomet Center for Conservation Sciences, Plymouth, MA.

Moss R, Schneider S. 2000. Towards consistent assessment and reporting of uncertainties in the IPCC TAR. In R. Pachauri and T. Taniguchi, editors. *Cross-Cutting Issues in the IPCC Third Assessment Report*. Global Industrial and Social Progress Research Institute (for IPCC), Tokyo.