



True Fir Forests

Northern California Climate Change Vulnerability Assessment Synthesis

An Important Note About this Document: This document represents an initial evaluation of vulnerability for true fir 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 Jeff Jones (U.S. Forest Service), Malcolm North (U.S. Forest Service and University of California, Davis), Hugh Safford (U.S. Forest Service), and Carl Skinner (retired U.S. Forest Service). Vulnerability scores were provided by Carl Skinner.

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Habitat Description

True fir forests, dominated by species in the *Abies* genus, generally occur in the mid- to upper montane zone in northern California, with elevation generally increasing with distance from the coast (Laacke 1990; Sawyer 2007). Across the region, true fir forests are most commonly dominated by white fir (*Abies concolor*) and Shasta red fir (*A. magnifica* var. *shastensis*), but patterns of dominance are variable across the region (Barbour & Woodward 1985; Sawyer 2007; Vuln. Assessment Reviewer, pers. comm., 2018). For instance, grand fir (*A. grandis*) is common in the lower elevations of the montane forest belt on western slopes of the North

Coast Range, but is replaced inland by extensive white fir forests across the Klamath Mountains and southern Cascades at elevations ranging from 1,070–2,100 m (3,500–6,900 ft; Stuart & Salazar 2000; Sawyer 2007; Briles et al. 2011). White fir forests start to mix with red fir over 1,400–1,500 m (4,600–4,900 ft), before transitioning to red fir-dominated forests and then subalpine forest types above about up to 2,200 m (7,200 ft) in elevation (Laacke 1990; Sawyer 2007; Briles et al. 2011). Red fir forests are most extensive at higher elevations in the montane belt and on the driest slopes of the subalpine zone (Taylor 1990; Sawyer 2007). North and west of the Klamath River in the Siskiyou Mountains, noble fir (*A. procera*) dominates high-elevation forests rather than Shasta red fir, though the two are ecologically very similar and are often lumped together as *A. magnifica* (Laacke 1990; Sawyer 2007). Noble fir also occurs at the upper elevations of the North Trinity Mountains, just east of the town of Hoopa in an area burned by the Megram fire (Cheng 2004; Vuln. Assessment Reviewer, pers. comm., 2018). Additionally, small, isolated stands of subalpine fir (*A. lasiocarpa*) or Pacific silver fir (*A. amabilis*) can be found throughout the Klamath Mountains (Sawyer 2007; Vuln. Assessment Reviewer, pers. comm., 2018).

Because firs are shade-tolerant and can regenerate in the absence of disturbance (Laacke 1990; Taylor & Halpern 1991), true fir forests are often multi-layered and have high canopy cover (Taylor & Halpern 1991; North et al. 2016; CNPS 2019). White fir becomes dominant on mesic sites in northern California, generally corresponding to a shift towards cooler, wetter conditions at higher elevations with less frequent fire compared to mixed conifer forests (Van de Water & Safford 2011; North et al. 2016). White fir comprises at least 60% of the canopy cover; other species that may occur include incense cedar (*Calocedrus decurrens*), Pacific dogwood (*Cornus nuttallii*), Jeffrey pine (*Pinus jeffreyi*), sugar pine (*P. lambertiana*), ponderosa pine (*P. ponderosa*), Douglas-fir (*Pseudotsuga menziesii*), canyon live oak (*Quercus chrysolepis*), and black oak (*Q. kelloggii*; CNPS 2019). At higher elevations, red fir forests are closely associated with the elevations where most precipitation occurs as snow (Barbour et al. 1991). The canopy is strongly dominated by red fir, though white fir, incense cedar, Brewer spruce (*Picea breweriana*), Sierra lodgepole pine (*P. contorta* ssp. *murrayana*), western white pine (*P. monticola*), Jeffrey pine, and mountain hemlock (*Tsuga mertensiana*) may also occur (Laacke 1990; Taylor 1990; CNPS 2019). Within all true fir forests, the presence and proportion of co-occurring species is dependent on moisture balance, soil characteristics, snowpack, disturbance and management history, and other factors (Laacke 1990; Taylor 1990; Barbour et al. 1991).

Shrub and herbaceous understory cover vary depending on site conditions, but can be sparse in closed-canopy stands and on xeric or less productive sites (Barbour & Woodward 1985; Sawyer 2007; North et al. 2016). Common understory species can include snowbrush (*Ceanothus velutinus*), mountain whitethorn (*C. cordulatus*), greenleaf manzanita (*Arctostaphylos patula*), bush chinquapin (*Castanopsis sempervirens*), mahala mat (*C. prostratus*), pinemat manzanita (*A. nevadensis*), and *Ribes* spp. (Taylor & Halpern 1991; Shatford et al. 2007; Skinner & Taylor 2018). Stands of montane chaparral occur frequently within landscapes dominated by true fir forests, often in severely-burned areas and on sites with shallow soils or high exposure to wind and ice damage (Airey Lauvaux et al. 2016; Skinner & Taylor 2018). Because heartrot is common

and fires are less frequent than in lower-elevation forest types, large cavities, snags, and downed wood are typically abundant (North et al. 2016).

True fir forests provide important foraging and nesting/denning habitat for many vertebrate wildlife species, including rare and/or sensitive species (Barrett 1988; Shimamoto 1988; Laacke 1990). For instance, fir cones provide food resources for squirrels and other small mammals, while deer browse new growth in the spring and insect-gleaning birds forage in the branches of mature trees (Laacke 1990; North et al. 2016). Several species, such as the Pacific fisher (*Pekania pennanti pacifica*), depend on old-growth habitat characteristics, which typically develop after about 200 years in true fir forests (Laacke 1990; North et al. 2016). Fir bark and boughs are also valued by northern California tribes for fuel, shelter, bedding, medicine, and other purposes (Turner et al. 2011).

Executive Summary

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

True Fir Forests	Rank	Confidence
Sensitivity	Moderate-High	High
Future Exposure	Moderate-High	Moderate
Adaptive Capacity	Low-Moderate	High
Vulnerability	Moderate-High	High

Sensitivity & Exposure Summary	<p><u>Climate and climate-driven factors:</u></p> <ul style="list-style-type: none"> • Air temperature, precipitation amount and timing, snowpack amount, timing of snowmelt and runoff, drought <p><u>Disturbance regimes:</u></p> <ul style="list-style-type: none"> • Disease, insects, wildfire, storms, wind <p><u>Non-climate stressors:</u></p> <ul style="list-style-type: none"> • Fire suppression, timber harvest
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True fir forests are sensitive to climate factors that alter the timing/length of the growing season and impact moisture availability, including warmer temperatures, changes in precipitation amount and timing, reduced snowpack, earlier timing of snowmelt and runoff, and increased drought. While changes in these factors could increase water stress, particularly at lower elevations, warming temperatures and changes in snowpack and snowmelt may also enhance forest productivity by increasing rates of tree growth and seedling establishment within the higher-elevation red fir zone. Changes in the frequency and/or severity of storms may increase the incidence of stem/top damage and windthrow in true fir forests, particularly in trees impacted by fungal diseases that cause decay. Increases in the severity of fire are likely to cause type conversion of true fir forests to shrubfields, particularly in a warmer, drier climate where competition for soil moisture is enhanced. Finally, fire suppression and timber harvest have significantly impacted true fir forests by altering forest structure and the availability and

type of potential fuels. Changes in forest structure as a result of logging and fire suppression can also increase vulnerability to disturbance such as intense wildfires, disease, insect attack, and windthrow.

Adaptive Capacity Summary	<p><i>Factors that enhance adaptive capacity:</i></p> <ul style="list-style-type: none"> + Somewhat protected from human disturbance in remote locations (for red fir forests) + High capacity for recruitment, especially in the absence of disturbance + Often able to outcompete other conifers following low- to moderate-severity fires + Valued for timber production, recreational opportunities, and wildlife habitat <p><i>Factors that undermine adaptive capacity:</i></p> <ul style="list-style-type: none"> – Upslope migration limited by lack of higher-elevation areas in most of the region – Low species diversity – Slow recovery following severe disturbances (e.g., wildfire, logging) – Management limited by lack of financial support
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True fir forests are less extensive than lower-elevation forest types, and are discontinuous across northern California because of their association with isolated mountaintops. Thus, upslope shifts in response to climate change will be limited by the lack of higher elevation areas in the Klamath Mountains and most of the southern Cascades. Because they are strongly dominated by fir and often have closed canopies, true fir forests support relatively low species diversity. However, spatial and temporal variation in disturbance regimes in combination with the ability to regenerate in the absence of disturbance results in high structural diversity in true fir forests across the landscape. High shade tolerance and abundant seed production also allow firs to outcompete other conifers following disturbances, though they are less drought- and fire-tolerant than pines. Recovery following severe disturbances (e.g., intense fire, heavy logging) can be very slow, leaving them vulnerable to type conversion to shrubfields, especially in the event of repeat high severity fires. Management of true fir forests is primarily limited by a lack of financial resources. However, this habitat type is valued by the public and research interest in true fir forests has been gradually increasing over the past several decades. Within the scientific literature, suggested management strategies include addressing changes associated with fire suppression (e.g., increased forest density, high fuel accumulations) to reduce forest vulnerability to large-scale disturbances and enhance natural regeneration.

Sensitivity and Exposure

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

There are no available projections that model potential range shifts in northern California true fir forests over the coming century. However, range shifts in response to warmer, drier conditions occurred during the rapid climate changes of the Holocene period (starting 11,700 years before present; Briles et al. 2011). Specifically, true firs shifted their range upslope by

800–1,000 m (2,620-3,280 ft) over 500 years in the early Holocene, which corresponded to a 5°C (9°F) increase in temperature (Briles et al. 2011). This suggests that past climate changes significantly impacted forest composition and structure across the region, contributing to range contractions and shifts in forest distribution that may occur again in response to future climate changes (Briles et al. 2011). In several stands dominated by mature red fir in the Trinity Alps Wilderness, the understory appears to be almost entirely comprised of white fir, suggesting that white fir may be currently moving upslope in response to a warming climate and may eventually displace red fir (C. Skinner, pers. obs., 2018).

Potential Changes in Habitat Distribution
<ul style="list-style-type: none"> • Likely shifts in distribution due to warmer, drier conditions • Possible refugia include sheltered north-facing slopes and/or ultramafic soils, as well as the few high peaks that exist in northern California (e.g., Mt. Shasta)
Source(s): Briles et al. 2011; Vuln. Assessment Reviewer, pers. comm., 2018

In the southern Cascades, potential refugia may occur on the few high peaks such as Mt. Shasta and Mt. Lassen (Vuln. Assessment Reviewer, pers. comm., 2018), although Mt. Lassen lies outside of the project study area. Refugia in the Klamath Mountains will likely be limited to sheltered north-facing slopes or ultramafic (serpentine) soils that might inhibit competition (Briles et al. 2011; Vuln. Assessment Reviewer, pers. comm., 2018).

Sensitivity and future exposure to climate and climate-driven factors

Regional experts evaluated true fir forests as having moderate-high sensitivity to climate and climate-driven factors (high confidence in evaluation), with an overall moderate-high future exposure to these factors within the study region (moderate confidence). Key climatic factors that affect true fir forests include air temperature, precipitation amount and timing, snowpack amount, timing of snowmelt and runoff, and drought.¹

Air temperature

Air temperature is strongly tied to patterns of tree growth and recruitment in high-elevation true fir forests, largely due to the impact of temperature on the timing (e.g., onset of spring/fall) and length of the growing season (Royce & Barbour 2001; Dolanc et al. 2013; Vuln. Assessment Reviewer, pers. comm., 2018). For instance, warmer temperatures may allow an earlier start to spring growth, in part by driving reductions in snowpack and earlier snowmelt (Royce & Barbour 2001; Dolanc et al. 2013). However, increases in evaporative demand may also exacerbate water stress as soil moisture becomes depleted later in the growing season, particularly at lower elevations where growth is limited by water availability to a greater degree (Royce & Barbour 2001; Chmura et al. 2011; Das et al. 2013).

Spring and dry season temperatures are positively correlated with area burned in mid- to high-elevation forests in northern California (Keeley & Slyphard 2015; Schwartz et al. 2015), due to

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

both reductions in fuel moisture (Abatzoglou & Williams 2016) and increases in the length of the fire season associated with increasing temperatures (Westerling 2016).

Regional Air Temperature Trends²	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> ● 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) <ul style="list-style-type: none"> ○ No seasonal temperature trends available 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> ● 2.2–5.5°C (4.0–9.9°F) increase in the average annual temperature by 2100 (compared to 1951–1980) for the North Coast, Northern Coast Range, Klamath Mountain, and Southern Cascade ecoregions (Flint et al. 2013; Flint & Flint 2014) <ul style="list-style-type: none"> ○ 1.9–5.8°C (3.4–10.4°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 <i>(see text for citations)</i>	
<ul style="list-style-type: none"> ● Possible increases in tree growth and recruitment at high elevations due to longer growing seasons ● Increased water stress due to enhanced evaporative demand at lower elevations where forests are more limited by water availability ● Increase in annual area burned due to reductions in fuel moisture and increasing length of the fire season 	

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

True fir forests occupy higher elevations where a significant proportion of annual precipitation occurs as snow (Barbour et al. 1991; North et al. 2016). The dominance of red fir, in particular, is closely associated with the transition to deep snowpack (Barbour et al. 1991). True fir growth and seedling establishment is strongly tied to patterns of precipitation, snowpack, and snowmelt, in large part because the length of the growing season is limited by snowpack that may persist late into the spring or early summer (Royce & Barbour 2001; Chmura et al. 2011; Dolanc et al. 2013). In general, snowpack persists later in the year where drifts are captured by vegetation and shaded by the forest canopy (Royce & Barbour 2001), and soil moisture at high elevations is strongly mediated by snowpack depth and timing of snowmelt (Dolanc et al. 2013).

For red fir, in particular, occasional years with low precipitation and early snowmelt appears to increase seedling establishment and the growth of young trees (Laacke 1990; Taylor 1990). However, over longer time periods, transitions to a greater proportion of precipitation

² 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.

occurring as rain rather than snow (Knowles et al. 2006), reduced snowpack (Hamlet et al. 2005; Mote et al. 2005; Gergel et al. 2017), and earlier snowmelt (Stewart et al. 2004; Thorne et al. 2015) could result in greater plant water stress by contributing to an earlier onset of the summer drought period and a shorter window for annual growth (Royce & Barbour 2001). However, trees at high elevations and on protected sites that maintain snowpack later into the season may be less affected (Dolanc et al. 2013).

Finally, drier conditions are strongly associated with increased wildfire activity (Miller et al. 2009, 2012; Yang et al. 2015; Westerling 2016; Gergel et al. 2017). While fires at high elevations have historically been limited by cool, wet conditions and a short fire season, warmer, drier conditions 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 (Schwartz et al. 2015; Yang et al. 2015; Restaino & Safford 2018).

Regional Precipitation, Snowpack, & Snowmelt Trends	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> • 7.2–9.4 cm (2.8–3.7 in) increase in mean annual precipitation and 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 (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) 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> • 20% decrease to 28% increase in mean annual precipitation by 2100 (compared to 1951–1980) for the North Coast, Northern Coast Range, Klamath Mountain, and Southern Cascade ecoregions (Flint et al. 2013; Flint & Flint 2014)³ • Seasonal changes are projected to be more significant as the wet season becomes wetter and shorter (i.e., later onset of fall rains and earlier onset of summer drought) and the dry season becomes drier and longer (Pierce et al. 2018; Swain et al. 2018) • Overall, interannual variability is expected to increase (Pierce et al. 2018; Swain et al. 2018) • Decreases in April 1 SWE by 2100 (compared to 1951–1980; Flint et al. 2013; Flint & Flint 2014): <ul style="list-style-type: none"> ○ 86–99% decrease on the North Coast ○ 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

³ 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	
	U.S.; Stewart et al. 2004; Rauscher et al. 2008)
Summary of Potential Impacts on Habitat <i>(see text for citations)</i>	
<ul style="list-style-type: none"> Increases in seedling establishment and tree growth during occasional dry periods and years of early snowmelt Possible decrease in the window for annual growth over the long term due to longer periods of summer drought Longer fire season and increased wildfire risk 	

Drought

Although red fir can benefit from occasional periods of low precipitation and early snowmelt (Laacke 1990; Taylor 1990), periods of severe and/or prolonged drought are likely to limit true fir growth (Dolanc et al. 2013) and recruitment (Chappell & Agee 1996; Young & Latimer 2017). For instance, a study in the southern Cascades and northern Sierra Nevada found that, across all sites, post-fire establishment of white fir decreased by up to 50% during periods of significant drought compared to weak drought (Young & Latimer 2017). Increased drought stress also increases the risk of mortality in dense stands and on dry sites (Guarín & Taylor 2005; Lutz et al. 2010) and enhances forest vulnerability to large-scale disturbances (Millar & Stephenson 2015), including wildfires (Miller et al. 2009; Littell et al. 2016) and insect outbreaks (Raffa et al. 2008; Kolb et al. 2016; Scheller et al. 2018).

Regional Drought Trends	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> 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) 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> Drought years are twice as likely to occur over the next several decades due to increased co-occurrence of dry years with very warm years (Cook et al. 2015) 80% chance of multi-decadal drought by 2100 under a high-emissions scenario (Cook et al. 2015) Severe droughts that now occur once every 20 years will occur once every 10 years by 2100 and once-in-a-century drought will occur once every 20 years (Pierce et al. 2018)
Summary of Potential Impacts on Habitat <i>(see text for citations)</i>	
<ul style="list-style-type: none"> Reduced growth and recruitment during periods of prolonged and/or severe drought Increased vulnerability to mortality, including from large-scale disturbances 	

Sensitivity and future exposure to changes in natural disturbance regimes

Regional experts evaluated true fir forests as having moderate-high sensitivity to changes in natural disturbance regimes (high confidence in evaluation), with an overall moderate-high future exposure to these stressors within the study region (moderate confidence). Key natural

disturbance regimes that affect true fir forests include disease, insects, wildfire, storms, and wind.⁴

Disease and insects

True firs are highly vulnerable to rotting diseases such as annosus root rot (*Heterobasidium annosum*), which can reduce growth and increase tree mortality (Schmitt et al. 2000; Zhang et al. 2007; Kliejunas 2011). Intensively managed stands are particularly vulnerable to this disease, as aerially-dispersed spores can infect fresh stumps or wounds caused by equipment, then spread from tree to tree through root contact (Laacke 1990). Trees with root damage experience increased moisture stress and reduced vigor, predisposing them to beetle attack (e.g., from fir engraver beetles [*Scolytus ventralis*]; Laacke 1990; DeSiervo et al. 2018). Other rotting diseases that can impact true firs in northern California include yellow cap fungus (*Pholiota limonella*) and Indian paint fungus (*Echinodontium tinctorium*; Laacke 1990). Losses to heart rot fungi can be particularly significant among old-growth trees (Laacke 1990).

Dwarf mistletoe is common throughout the range of both white fir and red fir, infecting about 40% of stands in California (Scharpf 1978; Laacke 1990). Dwarf mistletoe are parasitic flowering plants that use water and photosynthate produced by the host tree, reducing available resources (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). Because they have reduced vigor, trees impacted by dwarf mistletoe are more vulnerable to *Cytospora*, which is a canker-forming fungus that kill branches and reduces tree growth (Scharpf 1978; Laacke 1990; DeSiervo et al. 2018), as well as to the entry of heart rots, which weaken limbs and can make the tree more vulnerable to stem breakage (Laacke 1990).

Changing climate conditions may impact the prevalence and severity of insects and disease by directly influencing pathogen production/transmission and insect survival and reproduction; indirectly, the impacts of climate change may alter tree defenses, host susceptibility, and community interactions (Bentz et al. 2010; Kliejunas 2011; Sturrock et al. 2011; Weed et al. 2013; Kolb et al. 2016). For instance, warmer temperatures over the coming century are likely to increase dwarf mistletoe survival and host infection rates, particularly at higher elevations where short growing seasons may currently limit reproduction (Kliejunas 2011). *Armillaria*, dwarf mistletoe, bark beetles, and many other insects and pathogens are also more likely to colonize drought-stressed trees, suggesting that these could become more severe and/or widespread under drought conditions (Kliejunas 2011; Sturrock et al. 2011; Kolb et al. 2016; DeSiervo et al. 2018). Conversely, infected trees experience higher mortality rates due to drought stress and beetle attack (Laacke 1990; Kliejunas 2011; Kolb et al. 2016).

Climate change is particularly likely to exacerbate interactions between new and/or existing stressors. In the Klamath Mountains, widespread die-off of Shasta red fir has recently been observed due to a combination of increased incidence of fir engraver beetle attack and Wien's

⁴ All disturbance regimes presented were ranked as having a moderate or higher impact on this habitat type.

dwarf mistletoe (*Arceuthobium abietinum wiensii*), followed by subsequent *Cytospora* infections (DeSiervo et al. 2018). Declines in Shasta red fir appear to be correlated with increasing winter minimum temperatures and high stand densities, suggesting that both climate change and the legacy of fire exclusion are playing a role (DeSiervo et al. 2018).

Regional Disease & Insect Trends	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> Recent observed declines in Shasta red fir within the Klamath Mountains due to a combination of insect attack, dwarf mistletoe, and fungal disease, particularly where stand density is high (DeSiervo et al. 2018) 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> Changes in patterns of insect and disease prevalence and/or severity, depending on site conditions and limiting factors (Bentz et al. 2010; Kliejunas 2011; Sturrock et al. 2011) <ul style="list-style-type: none"> Increased dwarf mistletoe survival and host infection rates due to warmer temperatures (Kliejunas 2011) Increase in <i>Armillaria</i> and other root rot pathogens due to increased drought (Kliejunas 2011) Reduced cold-related mortality and accelerated insect development, increasing the likelihood of large-scale attack (Bentz et al. 2010)
Summary of Potential Impacts on Habitat <i>(see text for citations)</i>	
<ul style="list-style-type: none"> Increased tree mortality and increased likelihood of large-scale dieback due to insects and disease, especially during periods of drought Reduced seed and cone production in trees infected with dwarf mistletoe 	

Wildfire

Fire regimes in true fir forests are generally characterized by moderately-frequent mixed-severity burns (Skinner & Taylor 2018; Spies et al. 2018), with most fires occurring in the late summer or fall after the end of annual growth (Taylor 2000; Bekker & Taylor 2001). Historically, mean fire return intervals reported for true fir forests state-wide ranged from 24–110 years (Taylor & Halpern 1991; Taylor 1993, 2000; Stuart & Salazar 2000; Bekker & Taylor 2001; Van de Water & Safford 2011). At higher elevations, fire frequency and intensity decrease due to cool, wet conditions that limit productivity and the length of the fire season (Taylor 2000; Bekker & Taylor 2001; Skinner et al. 2006; Spies et al. 2018). However, estimates of fire frequency may be underestimated because few healed fire scars are available as evidence due to the prevalence of heartrot in trees with open wounds (Skinner & Taylor 2018; Vuln. Assessment Reviewer, pers. comm., 2018).

Fire behavior is strongly associated with fuel structure and availability, such as understory vegetation continuity and the presence of natural fuels breaks in the terrain (e.g., rocky areas or pinemat manzanita; Skinner 2003; Whitlock et al. 2004; Skinner & Taylor 2018). As a result, isolated red fir stands surrounded by extensive exposed rock likely have longer fire return intervals compared to more continuous forests, particularly those that may experience fire

carried up from connected lower-elevation forest types (North 2014). Following the implementation of fire exclusion and in the early 1900s, fire return intervals in some areas have increased (especially at lower elevations), resulting in significant changes in forest structure and fuel availability (Taylor & Halpern 1991; Taylor 1993, 2000; Stuart & Salazar 2000; Bekker & Taylor 2001; Skinner et al. 2018; Skinner & Taylor 2018).⁵

True firs are more sensitive to fire than pines, particularly as seedlings and saplings when they have very thin bark and drooping lower branches that make them highly susceptible to fire (Zouhar 2001; Skinner et al. 2018). Unlike the yellow and white pines, fir species also do not self-prune lower branches as they mature (Safford & Stevens 2017). Dominance by white fir typically develops where longer fire intervals allow trees to mature and develop characteristics such as thicker bark that will allow them to survive low- to moderate-intensity fire (Van de Water & Safford 2011; North et al. 2016). Red fir is more vulnerable to injury and mortality than white fir at all ages (Skinner et al. 2018); however, it is still less sensitive than either mountain hemlock or lodgepole pine (Agee 1996; Skinner & Taylor 2018). Old-growth red fir stands are less vulnerable to wildfire because they have a compact litter bed that contributes to slow, smoldering fires (Vuln. Assessment Reviewer, pers. comm., 2018). However, the risk of high-intensity crown fires increases in the presence of ladder fuels (Skinner et al. 2018) or a buildup of dead fuels such as occurs following periods of high mortality (Vuln. Assessment Reviewer, pers. comm., 2018).

Low-intensity fires in true fir forests create multi-aged stands by initiating recruitment in both white fir and red fir while also allowing many larger trees to survive (Taylor & Halpern 1991; Spies et al. 2018). By contrast, higher-intensity fire kills most mature trees but can allow abundant post-fire regeneration on mineral seedbeds in burned areas (Taylor 1990; Taylor & Halpern 1991; Shatford et al. 2007; Crotteau et al. 2013). However, severely burned areas are often colonized by shrubs (Shatford et al. 2007; Crotteau et al. 2013; Airey Lauvaux et al. 2016; Miller et al. 2016), which may dominate a site for many years or decades until they are gradually shaded out by overtopping firs (Airey Lauvaux et al. 2016). Across the region, conifer regeneration in true fir forests generally decreases as shrub cover increases (Shatford et al. 2007; Crotteau et al. 2013; Welch et al. 2016), likely due to greater competition for soil moisture (Crotteau et al. 2013). Because shrubfields tend to burn at higher severity compared to the surrounding forest (Airey Lauvaux et al. 2016), a shift towards more frequent fires as a result of warmer, drier conditions may prevent fir trees from maturing enough to withstand fire before reburning, resulting in permanent conversion of true fir forests to shrubfields (Airey Lauvaux et al. 2016; Tepley et al. 2017). Conversion to shrubfields following repeated fires likely represents one of the most significant threats to red fir forests as the climate warms and fire season lengthens (Vuln. Assessment Reviewer, pers. comm., 2018).

⁵ Refer to the section on non-climate stressors for a more complete discussion of the impacts of fire suppression on true fir forests.

Regional Wildfire Trends	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> • 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): <ul style="list-style-type: none"> ○ 184–274% increase in frequency ○ 270–492% increase in total area burned ○ 215% increase in length of the fire season • Changes in fire size, area burned, and fire frequency over the past several decades remain well below historical tribally-influenced frequency and extent of burning in California (Stephens et al. 2007) • No significant trends in the average areal proportion of high-severity fire were documented in northwestern CA from 1984–2008 (Miller et al. 2012; Parks et al. 2015; Law & Waring 2015; Keyser & Westerling 2017) <ul style="list-style-type: none"> ○ The relatively short period of record for fire severity data may obscure long-term trends ○ To date, there are no peer-reviewed studies on trends in northern California fire severity that include data from the last ten years 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> • 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) <ul style="list-style-type: none"> ○ 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) <ul style="list-style-type: none"> ○ 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) <ul style="list-style-type: none"> ○ 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 <i>(see text for citations)</i>	
<ul style="list-style-type: none"> • <i>Immediate:</i> <ul style="list-style-type: none"> ○ Enhanced regeneration following low-intensity fires ○ Increased tree mortality during moderate- and high-intensity fires, especially where high stand densities promote crown fires • <i>Short-term (~2-year):</i> <ul style="list-style-type: none"> ○ Increased competition from shrubs on severely burned sites • <i>Long-term:</i> <ul style="list-style-type: none"> ○ Possible type conversion to shrubfields following repeated high-severity fires 	

Storms and wind

Northern California can experience intense winter storms between November and March, which track eastwards from the Pacific and often include heavy rain, snow, and strong winds (Grantham 2018). Changes in storm patterns as a result of climate change would likely increase the frequency, duration, and/or intensity of strong wind events and heavy snow that result in stem breakage and windthrow in true fir forests (Gordon 1973; Jimerson & Jones 2003).

Diseased trees are more vulnerable to windthrow (Laacke 1990; Schmitt et al. 2000), particularly at the edges of stands that have been opened up by logging (Gordon 1973) or fire (Vuln. Assessment Reviewer, pers. comm., 2018). For instance, in the winter of 1996 a large proportion of the Swain Mountain Experimental Forest in northeastern California was flattened by a strong wind event in areas that were being managed by shelter-wood and seed tree cutting to achieve natural regeneration (C. Skinner, pers. obs., 2018). Strong winds over ridges also snap the tops of trees, leading to surface fuels that can promote severe fires (Jimerson & Jones 2003). Large trees are particularly vulnerable to blowdowns, though the resulting canopy gaps can initiate recruitment (Taylor & Halpern 1991). Top damage also results in reduced cone production, as cones are located primarily in the uppermost crown of the tree (Laacke 1990).

Regional Storm & Wind Trends	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> • Decline in the frequency of extreme two-day precipitation events between 1950 and 2009, with a slight decrease in the amount of precipitation received during extreme two-day events (Mass et al. 2010) • No trends available for storm-related wind events 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> • Increase in storm intensity and duration, resulting in greater maximum precipitation rates and volume (Dettinger 2011; Shields & Kiehl 2016; Prein et al. 2017) • Slight to moderate increase in storm frequency (up to 30% increase in atmospheric river days, or ~2.5 days per year; Dettinger 2011) • Projected statewide increases in daily extreme precipitation values of 5–20% by 2100 (Pierce et al. 2018) • No projections available for storm-related wind events
Summary of Potential Impacts on Habitat <i>(see text for citations)</i>	
<ul style="list-style-type: none"> • Increased tree injury and mortality (e.g., stem breakage, windthrow) due to strong winds and heavy snow, particularly in diseased trees and at the edges of stands opened up by logging • Possible increases in surface fuels following large wind events, potentially enhancing the risk of intense fire • Reduced cone production following damage in tree crowns 	

Sensitivity and current exposure to non-climate stressors

Regional experts evaluated true fir forests as having moderate-high sensitivity to non-climate stressors (high confidence in evaluation), with an overall moderate-high current exposure to

these stressors within the study region (high confidence). Key non-climate stressors that affect true fir forests include fire suppression and timber harvest.⁶

Fire suppression

Fire suppression has reduced fire frequency in northern California true fir forests since the early 1900s (Taylor & Halpern 1991; Taylor 1993; Stuart & Salazar 2000; Skinner 2003a; Steel et al. 2015; Safford & Stevens 2017; Skinner et al. 2018; Skinner & Taylor 2018), and in some areas no fires have occurred since the early- to mid-1900s (Taylor & Halpern 1991; Stuart & Salazar 2000). However, overall the effect has been less acute than in lower montane forests with a major yellow pine component (Vuln. Assessment Reviewer, pers. comm., 2019).

Fire suppression has contributed to increased tree density and canopy cover in some true fir forests across the region, resulting in significant changes to forest structure (Stuart & Salazar 2000; Taylor 2000; Zhang et al. 2007). In the absence of fire, firs have a strong growth advantage over many other conifers because they better resource competitors (Laacke 1990; Fites-Kaufmann et al. 2007). They are also shade-tolerant and can regenerate in the absence of disturbance (Laacke 1990; Fites-Kaufmann et al. 2007). White fir, in particular, has expanded into many areas formerly dominated by fire-tolerant species such as pines and hardwoods (Stuart & Salazar 2000; Taylor & Skinner 2003; Crotteau et al. 2013; Skinner et al. 2018). High stand densities have been associated with reduced growth and increased tree mortality due to insects and disease in some areas, as well as leading to an accumulation of fuel that can increase the risk of high-severity wildfire (Guarín & Taylor 2005; Zhang et al. 2007; DeSiervo et al. 2018).

Anecdotal evidence also suggests that lack of fire on the landscape may have resulted in decreased seed production and/or viability due to over-mature/senescent trees in some white fir stands. These stands exhibited little to no post-fire recruitment, even though there was no significant vegetative competition. However, it is unknown whether the lack of recruitment was due to low seed production in senescent trees or some other factor (e.g., timing of cone crop production, seedling drought stress), as low regeneration rates were also noted in stands with an overstory that was presumably capable of producing viable seeds (J. Jones, pers. comm., 2019).

Timber harvest

Although firs were historically considered nuisance species because of their low timber value, demand for lumber has increased and, with the exception of wilderness areas, many true fir forests have been logged and/or are now being managed for commercial timber production (Zhang et al. 2007; Vuln. Assessment Reviewer, pers. comm., 2018). Because firs have thin bark, the movement of logging equipment often results in damage to residual trees in harvested areas, and most or all wounded trees are infected by fungal diseases that result in decay (Aho et al. 1989; Schmitt et al. 2000). However, in some cases sustainable timber harvest can make

⁶ All non-climate stressors presented were ranked as having a moderate or higher impact on this habitat type.

fuel reduction more affordable and thus counteract some of the negative influences of fire suppression (Vuln. Assessment Reviewer, pers. comm., 2019).

Logging and various silvicultural prescriptions (e.g., shelter-wood and seed tree cutting) also increase susceptibility of true firs at the stand edge to windthrow (Gordon 1973; Vuln. Assessment Reviewer, pers. comm., 2018), particularly where the spread of rotting diseases has occurred (Laacke 1990). Once stands have been opened up, shrub establishment can take over, making the area more prone to high-severity fire and potential conversion to shrubfields (Vuln. Assessment Reviewer, pers. comm., 2018).

Adaptive Capacity

True fir forests were evaluated by regional experts as having low-moderate overall adaptive capacity (high confidence in evaluation).

Habitat extent, integrity, continuity, and permeability

Regional experts evaluated true fir forests as having a low-moderate geographic extent (high confidence in evaluation), moderate structural and functional integrity (moderate confidence), and moderate continuity (moderate confidence).

Landscape permeability for true fir forests was evaluated as low-moderate (moderate confidence). Geologic features (e.g., lack of higher-elevation sites) and timber harvest/clear-cuts were identified as the primary barriers to habitat continuity across the study region.⁷

True fir forests are much less extensive than the mixed conifer forests typical of lower elevations (North et al. 2016). Because they tend to be associated with isolated mountaintops, true fir forests are discontinuous across the region (North et al. 2016). Red fir forests grow on the highest ridges in the Klamath Mountains (Sawyer 2007); thus, with the exception of Mt. Shasta and Lassen Volcano National Park (outside of the study area), upslope shifts in habitat distribution are strongly limited by elevational availability within the region (Vuln. Assessment Reviewer, pers. comm., 2018). Although their remote location protects them from human disturbance to some degree (Long et al. 2014), habitat continuity in true fir forests have been impacted by timber harvest and clear-cuts as fir is increasingly harvested to meet growing demands (Zhang et al. 2007; Vuln. Assessment Reviewer, pers. comm., 2018).

Habitat diversity

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

⁷ All barriers presented were ranked as having a moderate or higher impact on this habitat type.

True fir forests are dominated by species within the *Abies* genus, which act as keystone species in this habitat type. In northern California, true fir forests are primarily comprised of white fir and red fir, but noble fir, Pacific silver fir, and subalpine fir also occur in a few stands within the region (Sawyer 2007). Within the true fir forest zone in the western Klamath Mountains, some enriched stands occur with up to 16 conifer species in a small area (Cheng 2004; Sawyer 2007).

Mature stands are often monotypic (i.e., dominated by a single species) and have closed canopies on productive sites, with few understory species due to the presence of shade and a thick layer of duff (Barbour & Woodward 1985; Laacke 1990). Red fir, in particular, prefers deep, nutrient rich soils (Laacke 1990; Taylor 1990), while white fir can grow on a wide range of soils from almost every type of parent material, including ultramafic soils (Zouhar 2001; Briles et al. 2011). Plant species diversity in true fir forests is typically low, especially in red fir forests (Barbour & Woodward 1985; Barrett 1988; Shimamoto 1988), though they can be structurally complex (Taylor & Halpern 1991). High spatial and temporal variation in disturbances contribute to forest heterogeneity, as regeneration is often abundant in canopy openings created by wind and wildfire (Taylor 1990; Taylor & Halpern 1991). However, shade-tolerant firs can also regenerate in the absence of disturbance or in small canopy openings, creating multi-aged stands (Taylor & Halpern 1991; North et al. 2016). This combination of even-aged and multi-aged stands further enhances structural complexity across the landscape (Taylor & Halpern 1991; Taylor 1993; North et al. 2016).

Resistance and recovery

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

Firs produce large amounts of seed and can regenerate under shaded conditions (including under chaparral stands), as well as at high densities following low- and moderate-severity fire (Laacke 1990; Taylor & Halpern 1991; Taylor 1993; Chappell & Agee 1996; Fites-Kaufmann et al. 2007; North et al. 2016). Thus, they are often able to outcompete other conifers following wildfire, as long as high-severity patches are small enough that living, nearby seed sources are available and recruits have sufficient time to mature before the next fire comes (Stuart & Salazar 2000; Taylor & Skinner 2003; Fites-Kaufmann et al. 2007; Crotteau et al. 2013). However, true firs are less tolerant of drought and fire compared to pines (Skinner et al. 2018), and recovery following severe disturbances (e.g., intense wildfire, heavy logging) can be very slow, leaving them vulnerable to type conversion if disturbances reoccur before trees have had a chance to mature (Airey Lauvaux et al. 2016; Vuln. Assessment Reviewer, pers. comm., 2018).

Management potential

Public and societal value

Regional experts evaluated true fir forests as having moderate-high public and societal value (high confidence in evaluation).

True fir forests are valued by the public for recreation (Laacke 1990; Vuln. Assessment Reviewer, pers. comm., 2018), as well as for timber production, wildlife habitat, and water storage and delivery (Zhang et al. 2007). As a result, societal support and interest in the management of true fir forests has been increasing in recent years (Zhang et al. 2007).

Management capacity and ability to alleviate impacts⁸

Regional experts evaluated the potential for reducing climate impacts on true fir forests through management as low-moderate (high confidence in evaluation). Regional experts identified use conflicts and/or competing interests for true fir forests as recreation, wildlife habitat, timber production, and water (Vuln. Assessment Reviewer, pers. comm., 2018).

Management actions to reduce the impacts of climate on true fir forests are largely limited by lack of financial support (Vuln. Assessment Reviewer, pers. comm., 2018). Additionally, true fir forests are typically managed passively (that is, they receive few thinning, planting, or prescribed fire treatments), so many management efforts occur in response to disturbances rather than due to proactive measures to support resilient forest structure and composition (Long et al. 2014; North 2014; Vuln. Assessment Reviewer, pers. comm., 2018).

Addressing changes associated with fire suppression, such as increased forest density and high fuel accumulations, may reduce forest vulnerability to disturbances such as uncharacteristically severe fire, insects, and disease (Long et al. 2014; DeSiervo et al. 2018). For instance, restoring fire to red fir forests through the use of prescribed fire or managed wildfire, where feasible, is an effective way to ensure natural red fir regeneration (Taylor & Halpern 1991; Taylor 1993; Skinner 2003b; Long et al. 2014; North 2014). Restoring regular fire may also reduce the risk of repeated high-severity fires that could result in type conversion to shrublands (Skinner et al. 2018; Skinner & Taylor 2018). To date, only a few places have attempted to allow fire to play a more natural role with mixed results (Skinner et al. 2018; Skinner & Taylor 2018). In drier, lower-elevation forests, fuel reduction (e.g., thinning from below) prior to the reintroduction of fire may be necessary, particularly in productive stands with high fuel accumulations (Zhang & Oliver 2006; Long et al. 2014; North 2014). In addition to reducing potential fuels that have built up over the past century of fire exclusion, thinning from below to remove shrubs and small trees can increase individual tree growth and size without impacting overall stand productivity (Zhang et al. 2007).

Ecosystem services

Healthy true fir forests provide a variety of ecosystem services, including:

- Provisioning of fiber and fresh water;
- Regulation of natural hazards (e.g., landslides), erosion control/sediment retention, and water quality;
- Supporting water cycling and carbon sequestration; and

⁸ Further information on climate adaptation strategies and actions for northern California can be found on the project page (<https://bit.ly/31AUGs5>).

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

True fir forests play an important role in the storage and delivery of California's water supply, as well as providing wildlife habitat and opportunities for human recreation (Zhang et al. 2007).

Recommended Citation

Hilberg LE, Reynier WA, Kershner JM. 2019. True Fir 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>).

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Northern California Climate Adaptation Project: Vulnerability Assessment Methods and Application

Defining Terms

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

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

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

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

Vulnerability Assessment Model

The vulnerability assessment model applied in this process was developed by EcoAdapt (EcoAdapt 2014a; EcoAdapt 2014b; Kershner 2014; Hutto et al. 2015; Gregg 2018),⁹ 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.¹⁰

⁹ Sensitivity and adaptive capacity elements were informed by Lawler 2010, Glick et al. 2011, and Manomet Center for Conservation Sciences 2012.

¹⁰ 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 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:

$$\text{Vulnerability} = [(\text{Climate Exposure} * 0.5) \times \text{Sensitivity}] - \text{Adaptive Capacity}$$

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

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

Vulnerability Assessment Model Elements

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

- **Climate and Climate-Driven Factors:** e.g., air temperature, precipitation, freshwater temperature, soil moisture, snowpack, extreme events: drought, altered streamflows, etc.
- **Disturbance Regimes:** e.g., wildfire, flooding, drought, insect and disease outbreaks, wind
- **Future Climate Exposure:** e.g., consideration of projected future climate changes (e.g., temperature and precipitation) as well as climate-driven changes (e.g., altered fire regimes, altered water flow regimes, shifts in vegetation types)
- **Stressors Not Related to Climate:** e.g., tectonic and volcanic events; residential or commercial development; agriculture and/or aquaculture; roads, highways, trails; dams and water diversions; invasive and other problematic species; livestock grazing; fire suppression; timber harvest; mining; etc.

Sensitivity & Exposure (Applies to Species Groups and Species)

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

Sensitivity & Exposure (Applies to Species ONLY)

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

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

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

Adaptive Capacity (Applies to Habitats ONLY)

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

Adaptive Capacity (Applies to Species Groups, Species)

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

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