Coastal Conifer-Hardwood Forests

Northern California Climate Change Vulnerability Assessment Synthesis

An Important Note About this Document: *This document represents an initial evaluation of vulnerability for coastal conifer-hardwood 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.*

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Habitat Description

For the purposes of this assessment, coastal conifer-hardwood forests are being defined as forests characterized by Sitka spruce (*Picea sitchensis*) and/or grand fir (*Abies grandis*) as dominant canopy species.[1] These occur in areas influenced by coastal fog and within 0.6–4.8 km (1–3 mi) of the coast (Sawyer 2007; Vuln. Assesssment Reviewers, pers. comm., 2018). They experience a mild climate characterized by high annual rainfall and generally cool summers as a result of ocean breezes and frequent summer fog (Johnstone & Dawson 2010; Barbour et al. 2014), with annual temperatures ranging from 5–25°C (41–77°F) and annual precipitation of up to 300 cm (118 in) in the northernmost parts of the state (Mooney & Dawson 2016). Coastal conifer-hardwood forests generally occur east (i.e., landward) of coastal dune forests where salt spray prevents dominance by coast redwood, and are often interspersed with coastal scrub (Sawyer 2007; USDA Forest Service 2009; CNPS 2019).

The relative dominance of Sitka spruce versus grand fir changes along a north-south gradient (USDA Forest Service 2009). Sitka spruce is typically dominant in stands below 366 m (1,200 ft) in the northern portions of the study region, including Humboldt and Del Norte Counties (USDA Forest Service 2009). Sitka spruce-dominated forests occupy seaward bluffs and ravines, bottomlands, and steep upland slopes, and tend to occur on seasonally flooded or permanently saturated soils (CNPS 2019). Grand fir typically occurs as the canopy dominant north of the Russian River in areas below 427 m (1,400 ft), which comprises the southern portion of the study region (e.g., Mendocino County USDA Forest Service 2009). Grand fir-dominated forests occur in more upland areas, including bluffs, coastline slopes, maritime terraces, and mesic slopes above creeks and river mouths (CNPS 2019). Within the transition zone between these two areas, Sitka spruce and grand fir often co-dominate the canopy (USDA Forest Service 2009). Moving inland, forests dominated by Sitka spruce and/or grand fir become less abundant, transitioning into chaparral, Oregon white oak woodlands, and coast redwood or Douglas-fir/tanoak forests (Sawyer 2007; Mooney & Dawson 2016). However, this transition occurs at variable distances from the coast, depending on elevation and many other factors (Vuln. Assessment Reviewer, pers. comm., 2019).

Coastal conifer-hardwood canopy dominants (i.e., Sitka spruce, grand fir) typically occur in combination with other conifer and hardwood species, including coast redwood (*Sequoia sempervirens*), Douglas-fir (*Pseudotsuga menziesii*), California bay (*Umbellularia californica*), Pacific madrone (*Arbutus menziesii*), coast live oak (*Quercus agrifolia*), tanoak (*Notholithocarpus densiflorus*), red alder (*Alnus rubra*), bigleaf maple (*Acer macrophyllum*), western hemlock (*Tsuga heterophylla*), Port-Orford-cedar (*Chamaecyparis lawsoniana*), bishop pine (*Pinus muricata*), and shore pine (*P. contorta contorta;* USDA Forest Service 2009; CNPS

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[1] Although a diverse range of forest alliances occur within the coastal band described, those dominated by Sitka spruce and/or grand fir were selected for special consideration by workshop participants because they are listed as vulnerable to critically imperiled vegetation types by the Bureau of Land Management. Shore pine on stabilized dune systems and coast redwood forests are both being considered in separate chapters. Pygmy conifer forests, bishop pine forests, and coast live oak woodlands are not being fully considered in this project because they are primarily distributed outside of the study area and/or are not found on USFS/BLM lands.

2019). In general, Sitka spruce-dominated stands have a more intermittent canopy than grand fir stands, resulting in a more continuous shrub layer and an abundant herbaceous layer often comprised of ferns (USDA Forest Service 2009; CNPS 2019). By comparison, grand fir-dominated stands tend to have a closed canopy, an intermittent or sparse shrub layer, and a more open herbaceous layer (USDA Forest Service 2009; CNPS 2019). Common understory shrubs and herbaceous species may include salmonberry (*Rubus spectabilis*), thimbleberry (*R. parviflorus*), huckleberry (*Vaccinium* spp.), California hazelnut (*Corylus cornuta var. californica*); salal (*Gaultheria shallon*), vine maple (*A. circinatum*), coyote brush (*Baccharis pilularis*), western swordfern (*Polystichum munitum*), and western trillium (*Trillium ovatum*; USDA Forest Service 2009).

Many species that occur within coastal conifer-hardwood habitats, as well as adjacent coast redwood forests, have significant cultural value to northern California Native American tribes (Huntsinger & McCaffrey 1995; Anderson 2005). Species used for food, fiber, and other materials include Sitka spruce, tanoak, coast redwood, big-leaf maple, and California hazelnut (Schenck & Gifford 1952; Huntsinger & McCaffrey 1995; Anderson 2005). For example, hazelnut stems and Sitka spruce roots are used for basketry (Schenck & Gifford 1952; Anderson 2005).

Executive Summary

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

Coastal Conifer-Hardwood Forests	Rank	Confidence
Sensitivity	Moderate-High	Moderate
Future Exposure	Moderate	Moderate
Adaptive Capacity	Moderate	Moderate
Vulnerability	**Moderate**	**Moderate**

Sensitivity & Exposure Summary	
	<u>*Climate and climate-driven factors*</u>: • Sea level rise, air temperature, precipitation timing, coastal fog <u>*Disturbance regimes*</u>: • Storms, flooding, disease, insects, wildfire, wind <u>*Non-climate stressors*</u>: • Invasive vegetation, roads/highways/trails, livestock grazing, agriculture, and residential/commercial development

Coastal conifer-hardwood forests are generally sensitive to climate factors and disturbance regimes that reduce remnant habitat extent and affect tree survival and recruitment. Sea level rise is likely to alter current forest distribution, potentially eliminating some low-lying forests via inundation and some bluff-top forests via bluff erosion, but also potentially creating new habitat areas by increasing salt-spray exposure in coast redwood forests. Coastal forests are also vulnerable to fragmentation and loss as human infrastructure vulnerable to flooding is relocated inland. Shifts in precipitation timing and coastal fog duration and extent can alter moisture availability, affecting tree growth, recruitment, and disease risk. Warmer air temperatures are also likely to contribute to enhanced water stress as evaporative demand is increased. Storms contribute to elevated canopy damage, windthrow, and blowdown, which alters forest structure and light availability. Flooding delivers mineral-rich sediment to alluvial stands, but too frequent flooding events and prolonged inundation can cause tree mortality and facilitate species composition changes. Similarly, wildfire may enhance tree mortality, and forest recovery may be limited within the interior of large burned patches. Coastal forests are also experiencing significant levels of damage from many native and non-native insects and pathogens. Climate change is likely to exacerbate these problems by altering the prevalence and severity of insect and pathogen outbreaks, contributing to more significant changes in forest species composition, structure, and ecosystem processes.

Coastal conifer-hardwood forests are also sensitive to a variety of non-climate stressors. Invasive vegetation alters forest structure and species composition and affects food availability for wildlife; spread of invasive plants and pathogens is often facilitated by road, highway, and trail use. Coastal conifer-hardwood forests also experience habitat fragmentation and loss due to the presence and construction of roads, highways, and trails, as well as land-use conversion for livestock grazing, agriculture, and residential/commercial development.

Adaptive Capacity Summary	*<u>Factors that enhance adaptive capacity</u>*: + Resilient to some stressors (e.g., salt spray, seasonal flooding) + Long-lived canopy species maintain recruitment opportunities + Management partnerships exist that are focused on restoring and protecting habitat *<u>Factors that undermine adaptive capacity</u>*: – Limited regional geographic extent – Past management has reduced heterogeneity, degrading habitat integrity and resistance to disturbances – Only a few characteristic/dominant canopy species, and these species reach their southern distribution limit in the study region – Major canopy species are not able to reproduce until 20 years of age and have short seed longevity

Coastal conifer-hardwood forests range from Alaska to northern California but have a very limited distribution in the study region, where they are restricted to a narrow band 1–3 miles from the ocean. Habitat integrity is generally degraded due to a history of timber harvest, with very few old-growth stands remaining. While coastal conifer-hardwood forests occupy topographically diverse sites, forests dominated by Sitka spruce and grand fir feature few dominant canopy species. Additionally, both species reach their southern distribution limit in the study region, indicating they may not be resistant to future warming and drying conditions. Coastal conifer-hardwood forests are resistant to some stressors (e.g., salt spray, seasonal flooding), but are less so to others (e.g., fire). Wind-dispersed seeds are critical for post-disturbance recovery, and many canopy species are long-lived, which maintains recruitment opportunities. However, dominant canopy species do not reach reproductive maturity until 20 years of age, and seed longevity and seedling recruitment are low, which may inhibit habitat recovery. Coastal conifer-hardwood forests are valued by the public for their unique recreation opportunities, and there are several management partnerships between federal and state agencies and local organizations focused on protecting and restoring remnant habitats.

Sensitivity and Exposure

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

Potential Changes in Habitat Distribution
• Northward shifts in climatically suitable area for coastal forests characterized by Sitka spruce and grand fir, with small to moderate range contractions likely by 2080 • Range contractions may occur more rapidly than projected due to large-scale disturbances
Source(s): Fernández et al. 2015; Thorne et al. 2016, 2017; DellaSala et al. 2018; Vuln. Assessment Reviewer, pers. comm., 2019

Climate change is projected to alter climatically suitable habitat area for northern California forests dominated by Sitka spruce and grand fir due to increases in temperature and water stress, as well as possible changes in the timing and frequency of coastal fog (Flint & Flint 2012; Fernández et al. 2015; Thorne et al. 2016, 2017; DellaSala et al. 2018). Most studies agree that the climatically suitable area for coastal forests, including those dominated by Sitka spruce and grand fir, will generally shift north (Fernández et al. 2015; Thorne et al. 2016, 2017; DellaSala et al. 2018).[2] The largest losses in climatically suitable area would likely occur at the southern end of their range (in northern California) and under hotter/drier future scenarios (Fernández et al. 2015; Thorne et al. 2016, 2017; DellaSala et al. 2018). However, it is difficult to accurately predict shifts in habitat distribution for coastal forests because the global climate models (GCMs) used for many studies are unable to accurately capture the complex interactions between ocean currents and atmospheric processes that drive patterns of temperature and moisture (including fog) in coastal climates (Wang et al. 2010; Johnstone & Dawson 2010; Fernández et al. 2015). Damage and mortality due to insects, pathogens, and intense wildfire may result in more rapid range contractions due to lack of regeneration in areas that are climatically unsuitable for seedling establishment but still suitable for persistence of mature trees (Vuln. Assessment Reviewer, 2019).

[2] Projections for Thorne et al. (2016, 2017) 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). Fernández et al. (2015) created scenarios based on historically anomalous conditions within the coast redwood zone, which were compared to an average of 19 CMIP5 GCMs using the RCP 4.5 emissions scenario for 2020–2030. Finally, DellaSala et al. (2018) used 3 GCMs (CSIRO, CCCMA, HADCM3) using the A2 high-emissions scenario (comparable to RCP 8.5). These spanned a range of future temperature and precipitation projections for the 2050s and 2080s compared to baseline conditions from 1950–2000.

A study specifically looking at shifts in species distribution for grand fir and Sitka spruce projected small to moderate reductions in species distribution across the species' range by the 2080s (DellaSala et al. 2018). Northern California largely represents the southern distribution limit for these species, so while populations within the study area may decline as a result of climate change, these species are likely to persist and/or expand outside of the state in the northern portion of their range (DellaSala et al. 2018). Sitka spruce is projected to experience a 2% range reduction by 2080, with most of that occurring in the southern parts of its range (i.e., southern Oregon and northern California; DellaSala et al. 2018). Comparatively, grand fir is projected to experience 36% overall range loss, with very little to no area in northern California projected to remain suitable by the 2080s (DellaSala et al. 2018). Other canopy co-dominant species in coastal conifer-hardwood forests are similarly projected to experience distribution losses or contractions (e.g., 7% for western hemlock, 23% for coast redwood), primarily concentrated at the southern edge of the study area (DellaSala et al. 2018).

Sensitivity and future exposure to climate and climate-driven factors

Regional experts evaluated coastal conifer-hardwood forests as having moderate-high sensitivity to climate and climate-driven factors (moderate confidence in evaluation), with an overall moderate future exposure to these factors within the study region (low confidence). Key climatic factors that affect coastal conifer-hardwood forests include sea level rise, air temperature, precipitation timing, and coastal fog.[3]

Sea level rise

Habitat exposure to sea level rise will vary according to location along the coast (National Research Council 2012; NHE 2015; Patton et al. 2017). North of Cape Mendocino, land uplift is likely to reduce the relative rate of local sea level rise (National Research Council 2012; NHE 2015; Patton et al. 2017); by contrast, land subsidence in the Humboldt Bay area may increase relative rates of sea level rise relative to global averages (National Research Council 2012; Chiniewicz 2015; NHE 2015; Patton et al. 2017; Laird 2018) and associated risks to coastal conifer-hardwood habitats.

Sea level rise may contribute to the loss of some coastal conifer-hardwood stands by increasing seawater inundation, including both gradual inundation and periodic inundation associated with king tides and/or storm surge (Laird 2018). In general, coastal conifer-hardwood forests occurring at the lowest coastal elevations (often dominated by Sitka spruce, red alder, and other riparian hardwoods) are most vulnerable to inundation-related mortality, while forests occurring on or behind coastal bluffs may be less vulnerable to inundation (Vuln. Assessment Reviewers, pers. comm., 2019). Significant seawater inundation can elevate tree mortality; for example, fossil records show that Sitka spruce and red alder occurring along the back of an estuary in Washington State suffered mortality following significant, rapid seawater inundation (likely due to a tsunami; Atwater & Yamaguchi 1991). Gradual inundation associated with sea

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

level rise may also contribute to habitat loss via conversion to more aquatic habitat types (e.g., forested tidal swamp, salt marsh; Glick et al. 2007; Heberger et al. 2009).

Sea level rise may also contribute to coastal conifer-hardwood habitat alteration and loss by exacerbating wave exposure, particularly during king tides, high tides, and storm events, and elevating coastal erosion (Russell & Griggs 2012; Laird 2018). For example, increased wave exposure may increase tree vulnerability to toppling (Laird 2018). Additionally, sea level rise may contribute to habitat loss by eroding existing landforms that support coastal conifer-hardwood forests (e.g., coastal terraces; Vuln. Assessment Workshop, pers. comm., 2018). Projections of sea level rise and associated coastal erosion indicate that bluffs along the western U.S. coast may retreat by 10 to 30 m (32.8 to 98.4 ft) by 2100 (Runyan & Griggs 2003; Hampton et al. 2004; National Research Council 2012).

Alternatively, sea level rise may contribute to habitat expansion opportunities by exposing new areas to salt spray. Sitka spruce are salt-spray tolerant (Harris 1990), while coast redwoods are sensitive to salt spray (e.g., can incur limb damage; Olson et al. 1990), which has likely contributed to current patterns of Sitka spruce versus coast redwood dominance in areas experiencing salt spray in northern California (Harris 1990). However, colonization of newly-created habitats could take a long time (Vuln. Assessment Reviewer, pers. comm., 2019).

In addition to these direct impacts, sea level rise may also have secondary impacts on coastal conifer-hardwood forests by forcing relocation and realignment of critical existing human infrastructure. For example, a recent analysis found that 240 miles of roads in Humboldt County, including 58 miles of highway, are at significant risk of inundation with 1.4 m (4.59 ft) of sea level rise (Heberger et al. 2009). Realignment and relocation activities will likely occur on or through currently undeveloped areas, such as these coastal forests (Vuln. Assessment Reviewers, pers. comm., 2018).

Regional Sea Level Rise Trends[4]	
Historical & current trends: • At the Crescent City station, sea levels decreased by an average of 0.08 cm (0.03 in) per year from 1933–2018 (equivalent to a decrease of 0.08 m [0.26 ft] in 100 years; NOAA/National Ocean Service 2019) • At the Humboldt Bay (North Spit) station, sea levels increased by an average of 0.49 cm (0.19	*Projected future trends:* • Sea level rise trends by 2100 (compared to 2000), based on likelihood of occurrence (range includes projections linked to low-, moderate-, and high-emissions scenarios; Kopp et al. 2014; Griggs et al. 2017; Sweet et al. 2017; Anderson 2018):

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

Regional Sea Level Rise Trends[4]	
in) per year from 1977–2018 (equivalent to an increase of 0.49 m [1.6 ft] in 100 years; NOAA/National Ocean Service 2019) • At the Arena Cove station, sea levels increased by an average of 0.08 cm (0.03 in) per year from 1978–2018 (equivalent to a change of 0.08 m [0.27 ft] in 100 years; NOAA/National Ocean Service 2019)	○ For the Crescent City station, likely (66% probability) range of 0.03–0.65 m (0.1–2.1 ft), 0.5% probability will meet or exceed 1.55 m (5.1 ft); extreme scenario (representing ice sheet collapse) of 2.79 m (9.1 ft) ○ For the Humboldt Bay (North Spit) station, likely (66% probability) range of 0.62–1.24 m (2.0–4.1 ft), 0.5% probability will meet or exceed 2.15 m (7.0 ft); extreme scenario (representing ice sheet collapse) of 3.37 m (11.0 ft) ○ For the Arena Cove station, likely (66% probability) range of 0.21–0.94 m (0.7–3.1 ft), 0.5% probability will meet or exceed 2.04 m (6.7 ft); extreme scenario (representing ice sheet collapse) of 3.02 m (9.9 ft)

Summary of Potential Impacts on Habitat *(see text for citations)*
• Potential habitat loss due to increasing inundation and erosion and/or sea level rise-driven infrastructure relocation • Small areas of potential habitat expansion if increased exposure to salt spray alters distribution of coast redwood forests

Air temperature

Coastal forests in northern California experience a relatively cool and mild climate year-round compared to other forest ecosystems within the state due to the proximity of the ocean (Harris 1990; Mooney & Dawson 2016). The limited temperature variability characteristic of maritime climates allows coastal forests to attain great height and biomass, largely by maintaining a steady balance between the energetic costs of productivity (i.e., growth) and maintenance (i.e., respiration and transpiration; Larjavaara 2013). However, warmer temperatures can enhance drought stress by increasing evaporative demand (Flint & Flint 2012), particularly if fog is also reduced (Johnstone & Dawson 2010). Studies in coastal redwood forests suggest that the impact of warmer temperatures largely depends on whether growth is already limited by water stress (Carroll et al. 2014, 2018). That is, warmer temperatures may enhance growth in forests that are currently water-limited (at least in the short term). In warmer, drier areas in the southern portion of the study area, increasing temperatures are likely to be associated with growth declines due to increasing evapotranspiration (Carroll et al. 2014, 2018).

Regional Air Temperature Trends	

Historical & current trends:
• 0.02°C (0.04°F) increase in the average annual temperature between 1900 and 2009 for the Northwestern California ecoregion (Rapacciuolo et al. 2014)
  ○ No seasonal temperature trends available
	Projected future trends:
	• 2.2–5.2°C (4.0–9.4°F) increase in the average annual temperature by 2100 (compared to 1951–1980) for the North Coast ecoregion (Flint et al. 2013; Flint & Flint 2014)
	  ○ 1.9–4.5°C (3.4–8.1°F) increase in average winter minimum temperatures
	  ○ 2.3–6.1°C (4.1–11.0°F) increase in average summer maximum temperatures

Summary of Potential Impacts on Habitat *(see text for citations)*	
• Increased water stress due to greater evaporative demand associated with warmer temperatures • Possible growth declines in forests already limited by water stress	

Precipitation timing (i.e., seasonality) and coastal fog

Coastal conifer-hardwood forests are adapted to a Mediterranean-type climate where a majority of precipitation falls in winter, and summers are characterized by drought. However, in coastal areas maritime air and frequent coastal fog ameliorate dry summer conditions, contributing to the cool, humid conditions required for tree growth (Harris 1990; Johnstone & Dawson 2010; Barbour et al. 2014). Fog patterns are dependent on wind-driven upwelling of the California Current and can be impacted by large-scale patterns of climate variability (e.g., the Pacific Decadal Oscillation) and land-sea temperature contrasts (Schwing et al. 2006; Wang et al. 2010; Johnstone & Dawson 2010; O'Brien 2011), as well as inversion heights, wind direction, and topographical features that impact the movement of fog inland (Torregrosa et al. 2016). During the summer months when the upwelling of the California Current is at its strongest (Wang et al. 2010), heavy fog and low clouds blanket the coast, occurring on up to 44% of days between June and September (although significant interannual variability is present; Johnstone & Dawson 2010). Over the past several decades, a study at 2 locations in northern and central California found that the frequency of days with fog and low clouds (i.e., all cloud cover under 400 m [1,312 ft]) has declined by up to 33% (Johnstone & Dawson 2010).

In coastal forests, fog can contribute moisture to the ecosystem via fog drip and foliar absorption (i.e., direct absorption of fog water into the plant while the leaf is wet), reducing summer drought stress (Dawson 1998; Fischer et al. 2009; Limm et al. 2009, 2012; Carbone et al. 2012). Shading by low coastal clouds and fog also reduces air temperatures and the amount of water lost to transpiration (Fischer et al. 2009; Limm et al. 2009; Carbone et al. 2012). This can increase plant growth where coastal forests are limited by a lack of sunlight rather than water stress (Johnstone et al. 2013).

In general, projected shifts towards increases in winter rain and drier summers (Flint & Flint 2014; Pierce et al. 2018) may interact with changes in the frequency of coastal fog and increasing evaporative demand to alter patterns of moisture availability in coastal forests (Johnstone & Dawson 2010; Thorne et al. 2015; Fernández et al. 2015; DellaSala et al. 2018).

These changes are likely to affect tree growth and recruitment in these moist-adapted forests (Johnstone & Dawson 2010; DellaSala et al. 2018). For example, Sitka spruce requires moist conditions for growth (Harris 1990), but Sitka spruce seed dispersal is also influenced by moisture; cones open and release seeds during dry conditions, but close during wet conditions (Harris 1990; CNPS 2019). Grand fir seedlings are also vulnerable to water stress, and drought mortality is more common on moist sites where seedlings fail to develop deeper rooting systems, leaving them vulnerable to any depletion in surface soil moisture (Foiles et al. 1990). Comparatively, seedlings in drier microsites develop deep roots first, and thus are more resilient to soil drying (Foiles et al. 1990). In general, seedlings and saplings are more sensitive to changes in shallow soil moisture inputs compared to adult trees, and likely benefit to a greater degree from fog water deposition that augments late-season water availability (Baguskas et al. 2017).

There is limited evidence that reduced fog and associated increases in light availability contributes to patterns of greater-than-expected growth in coast redwood ecosystems that are not water stressed, perhaps due to greater light availability for photosynthesis at the height of the growing season (Johnstone et al. 2013; Carroll et al. 2014; Sillett et al. 2015). Overall, however, studies suggest that fog is critical to conifer physiological performance during the dry summer months and continued reductions in coastal fog would further enhance drought stress (Carbone et al. 2012; Fischer et al. 2016; Baguskas et al. 2016), likely contribute to shifts towards systems that are increasingly limited by water rather than light (Vuln. Assessment Reviewer, pers. comm., 2019).

Lastly, changes in the distribution of precipitation volume, timing, and distribution over the year could alter the vulnerability of forests to insect and pathogen attack (Hubbart et al. 2016). For example, soil-borne *Phytophthora* species proliferate during episodes of intense precipitation that saturate the soil for extended periods (Sturrock et al. 2011). Heavy precipitation also increases spore production and transmission of the exotic airborne pathogen *Phytophthora ramorum*, increasing sudden oak death infection risk (Kliejunas 2011; DiLeo et al. 2014; Haas et al. 2016). When episodes of heavy rain are followed by dry periods, trees and shrubs are at increased risk of pathogen attack (Vuln. Assessment Reviewer, pers. comm., 2019). Bark beetle attacks are also often followed by longer-than-normal dry seasons, extended droughts, or periods of excessive wetness (Christiansen et al. 1987; Raffa et al. 2008).

Regional Precipitation & Coastal Fog Trends	
Historical & current trends: • 7.2 cm (2.8 in) increase in mean annual precipitation between 1900 and 2009 for the Northwestern California ecoregion (Rapacciuolo et al. 2014)	*Projected future trends:* • 20% decrease to 27% increase in mean annual precipitation by 2100 (compared to 1951–1980) for the North Coast ecoregion (Flint et al. 2013; Flint & Flint 2014)[5]

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

Regional Precipitation & Coastal Fog Trends	
• 3–4% decline per decade in the frequency of California coastal fog and low clouds under 400 m (1,312 ft) from 1920–1950, then 0.5–1% decline per decade from 1950–2008 (O'Brien 2011) • Observed 33% decrease in the frequency of days with coastal fog and low clouds at two locations on the northern and central California coast over the past century (Johnstone & Dawson 2010)	• 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) • Weak decline (0.1% per decade) in the frequency of California coastal fog and low clouds by 2100, driven primarily by warming sea surface temperatures (O'Brien 2011) • Potential changes in global circulation patterns (e.g., the jet stream), the timing and strength of upwellings, wind intensity, and other interacting oceanic and atmospheric factors that influence coastal fog are largely unknown (Grantham 2018)
Summary of Potential Impacts on Habitat *(see text for citations)*	
• Altered moisture availability, potentially affecting growth, seed dispersal, and seedling recruitment • Wetter winter and spring conditions could increase pathogen production and infection risk	

Sensitivity and future exposure to changes in natural disturbance regimes

Regional experts evaluated coastal conifer-hardwood forests as having moderate sensitivity to changes in natural disturbance regimes (moderate 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 coastal conifer-hardwood forests include storms, flooding, disease, insects, wildfire, and wind.[6] In general, natural disasters and weather events (e.g., floods, storms) can alter coastal landscapes in dramatic ways and under much shorter timeframes than climate change (Vuln. Assessment Workshop, pers. comm., 2018).

<u>Storms and flooding</u>

Winter storms are part of the disturbance regime in coastal forests, and occur from October to March as mid-latitude cyclones form over the Northern Pacific Ocean (Lorimer et al. 2009). Winter storms can cause flooding, erosion, landslides, and windthrow in coastal forests (Lorimer et al. 2009). Thus, changes in the frequency, intensity, and/or duration of storms

[6] Disturbance regimes presented are those ranked as having a moderate or higher impact on this habitat type. Additional changes in disturbance regimes that may influence the habitat to a lesser degree include dune movement in response to storms and sea level rise, which could bury coastal conifer-hardwood forest distribution and/or alter species composition by promoting more sand-burial tolerant species (Vuln. Assessment Reviewers, pers. comm., 2018).

(Dettinger 2011; Shields & Kiehl 2016; Prein et al. 2017) may increase the likelihood of tree injury and mortality and associated creation of canopy gaps, ultimately affecting stand structure and species composition (Lorimer et al. 2009; Hadley & Knapp 2016).

Sitka spruce-dominated coastal conifer-hardwood stands occurring in bottomlands typically experience seasonal flooding (CNPS 2019), which delivers fine sediment rich in minerals (Lorimer et al. 2009). However, tree roots of common canopy species require some degree of aeration; established Sitka spruce are vulnerable to mortality from permanent flooding (e.g., from beaver ponding; Harris 1990). Additionally, large floods can be stand-replacing (Lorimer et al. 2009), and too-frequent flooding can inhibit stand recovery by sweeping away or burying seeds (Harris 1990) and/or result in stand replacement by more flood-tolerant species (e.g., coast redwood; Lorimer et al. 2009).

Regional Storm & Flooding Trends	
Historical & current trends: • 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)	*Projected future trends:* • 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) • More frequent/severe winter flooding due to an increase in extreme precipitation events (Dettinger 2011; AghaKouchak et al. 2018; Swain et al. 2018; Grantham et al. 2018) • State-wide, 200-year floods are expected to increase in frequency by 300–400%, becoming 50-year floods (Swain et al. 2018)
Summary of Potential Impacts on Habitat *(see text for citations)*	
• Increased tree injury and mortality due to elevated flooding, erosion, landslides, and windthrow, ultimately altering tree growth and forest structure • Increased tree mortality with inundation • Increased flooding frequency and volume may cause more stands to be in early successional stages and/or transition to more flood-tolerant species composition	

<u>Diseases and insects</u>

A wide range of both native and non-native pathogens and insects affect coastal conifer forests, many of which can cause significant levels of damage to component species. Root disease fungi can occur in all tree species, causing decay that can slow growth or cause total mortality (Cannon et al. 2016; Lockman & Kearns 2016). At endemic levels, insects and disease act as natural disturbance agents in coastal forests, contributing to a heterogenous forest structure by creating canopy gaps, snags, and downed logs (Fettig 2016; Lockman & Kearns 2016; Spies et al. 2018). However, climate change has the potential to alter the dynamics of these biotic disturbances on forest structure, species composition, and ecosystem functioning (Kliejunas 2011; Sturrock et al. 2011; Spies et al. 2018).

In coastal conifer hardwood forests, common root disease fungi include native black stain root disease (*Leptographium wageneri*), which impacts Douglas-fir, and non-native soil-borne *Phytophthora* species, which can impact most trees in this forest type (Vuln. Assessment Reviewer, pers. comm., 2019). Pines can additionally be impacted by western gall rust (caused by the fungal pathogen *Endocronartium harknessii*) and pine pitch canker (caused by the non-native fungus *Fusarium circinatum*; Vuln. Assessment Reviewer, pers. comm., 2019). Exotic *Phytophthora ramorum* (sudden oak death; impacts tanoak and true oaks) and *Phytophthora lateralis* (Port-Orford-cedar root rot) have also contributed to changes in the composition and structure of stands that feature these species (Kliejunas 2011; Jules et al. 2015). Sudden oak death, in particular, has attracted significant publicity in coastal areas, despite the fact that several other insects and diseases impact these forests to a similar or greater degree (Vuln. Assessment Reviewer, pers. comm., 2019). For instance, grand fir in Humboldt and Mendocino Counties is currently suffering high levels of damage from balsam woolly adelgid (*Adelges piceae*), a non-native pest that may reduce the grand fir population even before climate-related stressors have had a significant impact (Vuln. Assessment Reviewer, pers. comm., 2019). Other insects that impact this forest type include bark beetles, flatheaded fir borers (*Phaenops drummondi*; affects Douglas-fir), and non-native green spruce aphids (*Elatobium abietinum*; affects Sitka spruce; Vuln. Assessment Reviewer, pers. comm., 2019).

Changing climate conditions may impact the prevalence and severity of insect outbreaks and disease by directly influencing disease production and transmission and/or insect development and survival (Kliejunas 2011; Sturrock et al. 2011; Weed et al. 2013; Kolb et al. 2016). The spread and establishment of exotic insects and pathogens, in particular, may increase with changes in climate such as warmer temperatures and altered patterns of precipitation (Kliejunas 2011; Sturrock et al. 2011; Weed et al. 2013; Kolb et al. 2016). Climate change is also likely to alter tree defenses, host susceptibility, and community interactions (Raffa et al. 2008; Bentz et al. 2010; Kliejunas 2011; Sturrock et al. 2011; Weed et al. 2013; Kolb et al. 2016). For instance, warming temperatures may contribute to earlier insect emergence, more completed life cycles within a season, and expanding distributions for both insects and disease, especially those typically restricted to more southern ranges (Bentz et al. 2010; Kliejunas 2011; Sturrock et al. 2011). Changes in seasonal precipitation patterns are also likely to impact pathogens that require moisture (e.g., fungal diseases such as sudden oak death; Chmura et al. 2011; Kliejunas 2011; Fettig et al. 2013), though drought could reduce the prevalence of those that depend on moist conditions for spore production (Davidson et al. 2005; Kolb et al. 2016). However, drought stress can increase the risk of large-scale insect outbreaks (Bentz et al. 2010; Weed et al. 2013; Kolb et al. 2016), as water stress reduces tree vigor and limits the ability of trees to expel attacking insects (Bentz et al. 2010; Weed et al. 2013; Kolb et al. 2016). Root disease pathogens are also more likely to colonize drought-stressed trees, suggesting that these could become more severe and/or widespread under drought conditions (Sturrock et al. 2011; Kolb et al. 2016). Conversely, injury from insects and disease can also increase tree vulnerability to drought stress and associated mortality (Kolb et al. 2016)

Regional Disease Trends	
Historical & current trends: • Recent observed declines in grand fir within Humboldt and Mendocino Counties as a result of balsam wooly adelgid outbreaks (Vuln. Assessment Reviewer, pers. comm., 2019)	*Projected future trends:* • 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)
Summary of Potential Impacts on Habitat *(see text for citations)*	
• Increased tree mortality and increased likelihood of large-scale dieback due to insects and disease, especially during periods of drought • Possible shifts in species composition and forest structure	

Wildfire

Little information has been documented about historical fire regimes in forests dominated by Sitka spruce and/or grand fir. Fires occur infrequently (fire return intervals likely vary between 150–350+ years), as most stands occur in moist or wet/frequently flooded areas (Stuart & Stephens 2006; CNPS 2019). Fires that do occur are generally severe and are associated with drought periods and/or are under the influence of warm, dry east winds (Stuart & Stephens 2006; CNPS 2019). Fire suppression has had little significant impact in coastal conifer forests dominated by these species, as the typical fire return interval is much longer than the 60–70 years that effective fire suppression practices have occurred (Stuart & Stephens 2006).

Sitka spruce and most of the other commonly associated trees in this forest type are relatively fire-sensitive due to thin bark and shallow roots, although mature grand fir trees develop thicker bark that allows them to withstand low- to moderate-intensity surface fires (Griffith 1992; Stuart & Stephens 2006). Burned areas are colonized by wind-dispersed seeds from seed sources within 800 m (2,600 ft; Griffith 1992). This suggests that forest area within the interior of large burned patches may be unable to recover following large, severe fires that are projected to become more common over the coming century (Westerling 2018).

Regional Wildfire Trends	
Historical & current trends: • 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 fire size, area burned, and fire frequency over the past several decades remain well below historical tribally-influenced	*Projected future trends:* • State-wide, up to 77% increase in mean annual area burned and 50% increase in the frequency of extremely large fires (>10,000 ha) by 2100 (Westerling 2018) ○ Less significant increases or possible decrease along the North Coast (Westerling et al. 2011)

Regional Wildfire Trends	

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 (Parks et al. 2015; Law & Waring 2015; Keyser & Westerling 2017)
  ○ The relatively short period of record for fire severity data may obscure long-term trends
  ○ To date, there are no peer-reviewed studies on trends in northern California fire severity that include data from the last ten years
	• Little projected change in fire severity in northwestern California by 2050 in models based solely on historical fire-climate relationships (Parks et al. 2016)
	  ○ However, human activity and fuel buildup from decades of fire suppression have altered historical fire-climate relationships (Taylor et al. 2016; Syphard et al. 2017; Wahl et al. 2019), and projections that incorporate these factors suggest that more significant increases in fire severity and size may occur (Mann et al. 2016; Wahl et al. 2019)
	• The majority of impacts to natural and human ecosystems come from extreme fire events (i.e., fires that have a low probability of occurring in any given place and time), which are likely to increase over the coming century (Westerling 2018)
	  ○ Generally, these patterns are not well-represented in studies that evaluate indices of mean fire size, intensity/severity, etc.

Summary of Potential Impacts on Habitat *(see text for citations)*
• Increases in fire-related tree mortality • Reduced recovery of Sitka spruce and grand fir following large, severe fires

<u>Wind</u>

Wind plays a significant role in shaping coastal conifer-hardwood forest distribution, species composition, and stand structure (Foiles et al. 1990; Harris 1990; Olson et al. 1990; USDA Forest Service 2009; Hadley & Knapp 2016; CNPS 2019). For example, wind is the primary seed dispersal agent for dominant coastal conifer-hardwood canopy species, including grand fir and Sitka spruce (Foiles et al. 1990; Harris 1990; CNPS 2019). Wind also influences patterns of salt-spray exposure, which affects forest distribution by promoting Sitka spruce dominance over coast redwood (Olson et al. 1990; USDA Forest Service 2009). Similarly, studies in coastal Oregon have found Sitka spruce to be more responsive following major wind disturbance than Douglas-fir, contributing to higher Sitka spruce dominance on wind-prone sites (Hadley & Knapp 2016). Wind also influences species composition and stand structure by causing tree injury and canopy gaps (Hadley & Knapp 2016). Canopy damage decreases the radial growth in damaged trees and provides an entry for decay agents and pathogens while increasing light availability and associated growth for non-damaged trees and understory species (Hadley & Knapp 2016). Additionally, downed coarse woody debris caused by wind events eventually creates nurse logs for future tree germination (Hadley & Knapp 2016). Wind-created wounds eventually provide important cavity habitat for nesting, roosting, and hiding animal species (Vuln. Assessment Reviewer, pers. comm., 2019).

Wind can also affect stand structure at larger scales. The shallow-rooting behavior and high susceptibility to root-rot fungi of several dominant canopy species (e.g., Sitka spruce, western hemlock) increases individual tree vulnerability to windthrow and stand vulnerability to blowdown during high wind events, particularly during stormy periods when soils are waterlogged (Harris 1990; Packee 1990; CNPS 2019). Windthrow and blowdown can reset stand succession and significantly alter forest structure (Harris 1990; Packee 1990; CNPS 2019).

<table>
<tr><th colspan="2">Regional Wind Trends</th></tr>
<tr><td>Historical & current trends:
Alongshore winds increased from 1940-1990 (Bakun 1990; Schwing & Mendelssohn 1997; Mendelssohn & Schwing 2002)
No trends available for storm-related wind events</td><td>Projected future trends:
Winds are expected to increase in all seasons, particularly in summer and fall, due to increasing differences in land-ocean pressures and temperatures (Bakun 1990; Snyder et al. 2003; Auad et al. 2006; Largier et al. 2010)
No projections available for storm-related wind events</td></tr>
<tr><th colspan="2">Summary of Potential Impacts on Habitat (see text for citations)</th></tr>
<tr><td colspan="2">Altered seed dispersal
Potential changes in habitat distribution on seaward and inland edges by altering sand movement and salt-spray exposure
Potential changes in species composition (e.g., if more wind-sensitive species are prevented from achieving canopy dominance)
Increased canopy damage and gap formation, increasing light availability and altering tree growth rates
Potential increases in tree windthrow and stand blowdown, especially when soils are waterlogged</td></tr>
</table>

Sensitivity and current exposure to non-climate stressors

Regional experts evaluated coastal conifer-hardwood forests as having moderate sensitivity to non-climate stressors (moderate confidence in evaluation), with an overall moderate current exposure to these stressors within the study region (moderate confidence). Key non-climate stressors that affect coastal conifer-hardwood forests include invasive vegetation, roads/highways/trails, livestock grazing, agriculture, and residential/commercial development.[7]

<u>Invasive vegetation</u>

The moist, cool conditions characterizing coastal conifer-hardwood forests facilitate the establishment and growth of non-native and/or invasive plant species, including English ivy

[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 fire suppression, pollution/poisons (e.g., brown sites), and recreation.

(*Hedera helix*), Cape ivy (*Delairea odorata*), gorse (*Ulex europeas*), Scotch broom (*Cytisus scoparius*), French broom (*Genista monspessulana*), and Himalayan blackberry (*Rubus armeniacus*; Vuln. Assessment Reviewer, pers. comm., 2019). These compete with or degrade native vegetation and alter soil properties, ultimately affecting forest structure, species composition, and ecosystem processes (Alvarez & Cushman 2002; Dukes & Mooney 2004; Caldwell 2006; Waggy 2010; Cortenbach & Madurapperuma 2018).

English ivy is an exotic evergreen woody climbing vine introduced as an ornamental plant. It grows prolifically on many canopy species, including Sitka spruce, western hemlock, red alder, and Douglas-fir (Cortenbach & Madurapperuma 2018). Trees infested by English ivy may be more susceptible to windfall during storms, especially if they are in an already weakened condition due to drought stress, disease, or injury (Waggy 2010). English ivy forms near monocultures in the forest understory, suppressing establishment and growth of native shrubs, herbaceous plants, and tree seedlings (Hyland & Roye 2004; Waggy 2010). Changes in forest community composition as a result of exotic species likely negatively affects foraging resources available to wildlife (Hyland & Roye 2004; Waggy 2010). For example, English ivy likely reduces foraging opportunities for ground-feeding mammals and birds such as the dark-eyed junco (*Junco hyemalis*) by obscuring the ground and reducing the abundance of annual plants used as food sources (Hyland & Roye 2004).

Invasive grasses such as pampasgrass (*Cortaderia selloana*) and jubatagrass (*Cortaderia jubata*) may vigorously out-compete native vegetation in disturbed areas with higher light availability (e.g., roadsides, clear-cuts, burned areas; Lambrinos 2000; Norman et al. 2009; DiTomaso et al. 2010). These species can reduce conifer seedling recruitment, and may become more permanently established if climate-driven changes in disturbance regimes result in a more open canopy within coastal conifer hardwood forests (Norman et al. 2009; DiTomaso et al. 2010).

<u>Roads, highways, and trails</u>
Similar to impacts in other ecosystems, coastal conifer-hardwood forests likely experience elevated fragmentation and altered ecosystem dynamics as a result of road, highway, and trail construction and use (Trombulak & Frissell 2000; Coffin 2007). For example, transportation corridors contribute to the introduction and spread of exotic species (Trombulak & Frissell 2000; Coffin 2007). English ivy was found to exhibit more growth along roads than along trails in Patrick's Point State Park, likely because changes in air speed due to vehicle passage enhanced lateral seed dispersal (Cortenbach & Madurapperuma 2018). Similarly, disturbed conditions associated with road construction and maintenance facilitate the establishment and persistence of jubatagrass and pampasgrass on road edges (DiTomaso et al. 2010; USDA Forest Service 2018). Roads can also act as migration barriers to wildlife and/or enhance predation by providing travel corridors for wild and domestic predators (Trombulak & Frissell 2000; Coffin 2007). Finally, roads and trails provide enhanced opportunities for pathogen establishment in forest vegetation, both because of greater air movement that increases spore dispersal and because spores can be transported in soil adhering to vehicles, footwear, tools, and equipment (Jules et al. 2002; Cushman & Meentemeyer 2006a, 2006b; Hansen 2008).

Livestock grazing, agriculture, and residential/commercial development

Many coastal forests in northern California have been lost, fragmented, and/or modified as a result of land-use conversion for livestock grazing, agriculture, and urban development (Vuln. Assessment Reviewers, pers. comm., 2018). For example, several grand fir-dominated coastal conifer-hardwood stands have been converted to residential housing development in Mendocino County (CNPS 2019).

Adaptive Capacity

Coastal conifer-hardwood forests were evaluated by regional experts as having moderate overall adaptive capacity (moderate confidence in evaluation).

Habitat extent, integrity, continuity, and permeability

Regional experts evaluated coastal conifer-hardwood forests as having a moderate geographic extent (e.g., occurs across multiple states; high confidence in evaluation), low-moderate structural and functional integrity (moderate confidence), and low-moderate continuity (moderate confidence). Landscape permeability for coastal conifer-hardwood forests was evaluated as moderate (moderate confidence). Land-use conversion, invasive vegetation, and geologic features were identified as the primary barriers to habitat continuity and dispersal across the study region.[8]

Both Sitka spruce and grand fir reach their southern distribution limit in northern California (Foiles et al. 1990; Harris 1990; Sawyer 2007; CNPS 2019). Within the study region, Sitka spruce dominated stands are most continuous from Del Norte to Humboldt Counties, although three disjunct stands also occur between Big River and Fort Bragg in Mendocino County (CNPS 2019). Grand fir-dominated stands occur from Del Norte to Sonoma counties (Foiles et al. 1990; CNPS 2019), but are generally very small in size (<20 ha; <49.4 acres; CNPS 2019).

The ecological integrity of coastal conifer-hardwood forests varies across the region. Most stands were logged extensively over the last 100 years, although Redwood National Park features some old-growth Sitka spruce stands (CNPS 2019). The majority of remnant stands are second-growth, and many are in early successional stages (CNPS 2019). These are generally dense, even-aged, and relatively homogenous in structure, which increases their vulnerability to insects and disease (Vuln. Assessment Reviewers, pers. comm., 2018). The loss of large trees also reduces fog capture within the ecosystem, causing more xeric conditions (Dawson 1998). In general, habitat integrity is likely greatest for stands located within protected areas such as Redwood National Park, various state parks, and ecological reserves (CNPS 2019).

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

Habitat diversity

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

Coastal conifer-hardwood forests occupy topographically diverse areas in close proximity to the coast (CNPS 2019). The high landscape heterogeneity of the region contributes to varying species composition and stand structure, depending on substrate, topography, water balance, and disturbance history, among other factors (CNPS 2019).

Resistance and recovery

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

Coastal conifer-hardwood forests exhibit variable resistance to climatic stressors. For example, grand fir seedlings are somewhat resilient to drought (CNPS 2019). Additionally, grand fir has an adaptable root system that promotes survival on a variety of microsites (i.e., can develop a deep taproot in dry sites and shallow roots on moist sites; Foiles et al. 1990). However, coastal conifer-hardwood forests are generally not resistant to fire (CNPS 2019), and the legacy of past management practices (e.g., fire suppression) has further reduced resistance to disturbances such as fire and large-scale outbreaks of insects and disease (Vuln. Assessment Reviewer, pers. comm., 2019). Sitka spruce is sensitive to fire mortality due to thin bark and shallow roots, so fires that do occur are often categorized as severe (i.e., result in high levels of mortality; CNPS 2019). Mature grand fir are moderately fire-resistant due to thick bark, but smaller trees are vulnerable to fire mortality from thin bark (CNPS 2019).

Coastal conifer-hardwood forest recovery is similarly variable. Seeds of dominant canopy species are dispersed by wind, but seed dispersal distances are rather short (e.g., 750 m [2460 ft] for Sitka spruce, 120 m [400 ft] for grand fir; Foiles et al. 1990; CNPS 2019). Seeds are not long-lived, which can limit recruitment opportunities if favorable germination conditions do not occur in the near-term (Foiles et al. 1990; CNPS 2019). Seedling survival and recruitment is also low, as seedlings are more prone to mortality from heat stress, drought, and disease compared to established trees (Foiles et al. 1990). Both Sitka spruce and grand fir take at least 20 years to reach reproductive maturity, and seed crop abundance and quality vary annually, with good crops occurring only every few years (Foiles et al. 1990; Harris 1990). However, once established, these trees are fairly long-lived, with Sitka spruce living up to 800 years (Harris 1990; CNPS 2019), and grand fir living for 250–300 years (Foiles et al. 1990), maintaining opportunities for continued recruitment events. Both species are also relatively shade-tolerant (CNPS 2019), allowing some generation to occur within the forest understory.

Management potential

Public and societal value

Regional experts evaluated coastal conifer-hardwood forests as having moderate-high public and societal value (moderate confidence in evaluation).

Coastal forests offer a unique recreational experience that cannot be found anywhere else in the region (Vuln. Assessment Workshop, pers. comm., 2018). Extreme events, particularly tsunamis, droughts, and fires, may influence societal support for managing coastal conifer-hardwood forests in the future (Vuln. Assessment Workshop, pers. comm., 2018).

Management capacity and ability to alleviate impacts[9]

Regional experts evaluated the potential for reducing climate impacts on coastal conifer-hardwood forests through management as low-moderate (low confidence in evaluation). Regional experts identified residential and commercial development and recreation as potential use conflicts and/or competing interests for coastal conifer-hardwood forests (Vuln. Assessment Workshop, pers. comm., 2017).

Local non-profit collaborative groups are partnering with federal and state agencies to manage and conserve this habitat for the public and future use (Vuln. Assessment Workshop, pers. comm., 2017). Management options may include relocating trails and other recreation areas to reduce disturbance and invasive species introductions (Vuln. Assessment Workshop, pers. comm., 2017). In general, managers are knowledgeable on local coastal impacts, which they can use to educate the public and generate policies to direct future use and management efforts (Vuln. Assessment Workshop, pers. comm., 2017).

Ecosystem services

Coastal conifer-hardwood forests provide a variety of ecosystem services typically associated with forested habitats, including:

- Provisioning of food, fiber, fuel, and ornamental resources;
- Regulation of flood and erosion control, water purification, and natural hazard regulation;
- Support of nutrient cycling and water cycling; and
- Cultural/tribal uses for spiritual/religious purposes, knowledge systems, educational values, aesthetic values, social relations, sense of place, cultural heritage, inspiration, and recreation (Vuln. Assessment Workshop, pers. comm., 2017).

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

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

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Northern California Climate Adaptation Project:

Vulnerability Assessment Methods and Application

Defining Terms

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

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

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

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

Vulnerability Assessment Model

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

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[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

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