Chaparral Shrublands

Northern California Climate Change Vulnerability Assessment Synthesis

An Important Note About this Document: *This document represents an initial evaluation of vulnerability for chaparral shrublands 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 Anonymous (Bureau of Land Management), Anonymous (University of California, Davis), Tom Parker (San Francisco State University), Chad Roberts (Tuleyome), and Jim Weigand (Bureau of Land Management). Vulnerability scores were provided by Eureka and Redding workshop participants. Upper Lake workshop participants provided additional comments on the climate change vulnerability of this habitat.

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Habitat Description

Chaparral shrublands are characterized by dominant sclerophyllous ("hard-leaved") evergreen shrubs and small trees (Keeley & Davis 2007; Halsey & Keeley 2016; Parker et al. 2016). In northern California, chaparral shrublands generally occupy drier (xeric) sites (e.g., west and south slopes) with shallow soils, grading into oak woodlands and evergreen sclerophyllous woodlands (Parker et al. 2016). In moister (more mesic) areas, chaparral often grades into coniferous forests (Parker et al. 2016). Chaparral shrublands occur on both serpentine and non-serpentine soils, and substrate type influences species composition and habitat distribution (Keeley & Davis 2007; Parker et al. 2016; Hidalgo-Triana et al. 2018). Other factors that control species composition and habitat distribution include local climate, elevation, topography, and fire regime (Keeley & Davis 2007; Parker et al. 2016).

Note that the research base for chaparral shrublands in northern California is sparse compared to other areas of the state. A majority of the research cited in this assessment originates from southern California, and to a lesser extent, the Sierra Nevada and the central California coast. Results of these studies should be interpreted and applied with caution until more is known about how the ecology of northern California chaparral functions in comparison to chaparral shrublands elsewhere in the state (e.g., see discussions in Keeley 2002; Duren & Muir 2010).

Northern California features two generalized chaparral communities, montane chaparral and mixed chaparral, both composed of diverse vegetation alliances (England 1988; Risser & Fry 1988; USDA Forest Service 2009a, 2009b; CNPS 2019). Montane chaparral is associated with mid- to higher elevations (914–2,743 m; 3,000–9,000 ft) in the Northern California Coast Range, Northern California Interior Coast Range, Klamath Mountains, and southern Cascades (Risser & Fry 1988). It forms a climax community on shallow soils, or an early successional community following disturbance in coniferous forest areas (Risser & Fry 1988). Along with evergreen species, montane chaparral may also feature deciduous or partially deciduous species, and often intergrades with mixed chaparral at lower elevations (Risser & Fry 1988). Numerous closely related plant species occur in montane chaparral throughout its range, and often different species in the same genus replace one another regionally and/or elevationally (Vuln. Assessment Reviewers, pers. comm., 2018).

Mixed chaparral is associated with mid- to lower elevations (generally below 1,520 m; 5,000 ft) (England 1988). It is most abundant on the east-facing slopes of the Northern California Coast Range and Interior Coast Range, and occurs in more fragmented patches in the southern Cascades, Klamath Mountains, and northernmost Sierra Nevada foothills (England 1988). At its upper-elevation limit, mixed chaparral intergrades with montane chaparral and coniferous forests (England 1988). At lower elevations, mixed chaparral intergrades with annual grassland and blue oak-foothill pine woodlands (England 1988).

Serpentine chaparral occurs within mixed chaparral and montane chaparral areas on ultramafic soils, which are characterized by low macronutrients and calcium levels, very high magnesium and iron, and generally rocky and dry conditions with shallow soils (Safford & Harrison 2004). Serpentine vegetation is adapted to these unique conditions, although species that appear on serpentine soils can also appear on non-ultramafic substrates (Brady et al. 2005). Low-productivity soils generally render serpentine chaparral shrublands less productive than non-serpentine chaparral (Harrison 1997; Keeley & Davis 2007). However, serpentine chaparral areas contribute to overall high chaparral plant diversity, and often host many rare and endemic species (Harrison 1997; Safford & Harrison 2004; USDA Forest Service 2009a, 2009b; Hidalgo-Triana et al. 2018).

Across all sub-types, chaparral vegetation is largely drought- and fire-adapted (Keeley & Davis 2007; Parker et al. 2016). Dominant chaparral species can be categorized by mode of regeneration following fire: seedling recruitment (i.e., obligate seeders), resprouting (i.e., obligate resprouter), or a combination of both strategies, termed facultative seeding (Keeley 1991; Keeley et al. 2005b; Cornwell et al. 2012). These regeneration strategies and other plant characteristics influence species sensitivity to moisture stress, thus controlling landscape distribution of different species (see table below; (Burk 1978; Meentemeyer & Moody 2002; Jacobsen et al. 2007, 2014; Anacker et al. 2011; Cornwell et al. 2012).

Post-Fire Recruitment Method	Moisture Stress Adaptations & Generalized Landscape Location	Sample Species
Obligate seeder *Description*: Accumulate seed stores that require fire for germination	Most common in drier areas, including serpentine soils: higher cavitation resistance (i.e., xylem resistance to collapse) and low specific leaf area enables survival during high levels of water stress	• Non-sprouting *Ceanothus* spp. • Non-sprouting manzanita (*Arctostaphylos* spp.)
Obligate resprouter *Description*: After fire these plants resprout from underground roots or burls; they also recruit from seed during fire-free intervals, but seeds are killed by fire	Most common in more mesic areas: typically have deeper root systems, allowing enhanced access to water during dry periods	• Shrub oaks (*Quercus* spp.) • Coffeeberry (*Frangula californica*) • Toyon (*Heteromeles arbutifolia*)
Facultative seeder *Description*: Utilize both vegetative resprouting and seed germination following fire-induced topkill; although their seeds largely germinate after fire, they have been documented to germinate in the absence of fire as well	More common in drier areas	• Chamise (*Adenostoma fasciculatum*) • Facultative seeding *Ceanothus* spp. • Facultative seeding *Arctostaphylos* spp.
Source(s): Burk 1978; Keeley 1991; Meentemeyer & Moody 2002; Safford & Harrison 2004; Keeley et al. 2005a; Jacobsen et al. 2007, 2014; Anacker et al. 2011; Cornwell et al. 2012		

Chaparral shrublands are critically important to many Native American tribes in northern California, supporting species utilized for food, medicines, and materials for ceremonies, basketry, cordage, clothing, utensils, tools, and dwellings (Anderson 2005; Anderson & Rosenthal 2015; Anderson & Keeley 2018). For example, hairy manzanita (*Arctostaphylus columbiana*) is a culturally significant shrub to the Wiyot Tribe, who use it as firewood in ceremonial fires, and use its berries to make manzanita cider (Anderson 2005; Vuln. Assessment Reviewers, pers. comm., 2018). Examples of other regionally important chaparral species include western redbud (*Cercis occidentalis*; utilized for basketry), yerba santa (*Eriodictyon californicum*; used for medicinal purposes), serviceberry (*Amelanchier* spp.; yields edible fruit), multiple other manzanita species (used for medicines, tools, and materials), and several herbaceous plants that yield edible seeds, including farewell-to-spring (*Clarkia purpurea*), red maids (*Calandrinia menziesii*), and chia (*Salvia columbariae*; Anderson 2005; Anderson & Keeley 2018). Across California, more than 400 plant species found in chaparral communities are known to have been utilized by Native American tribes (Anderson & Keeley 2018). However, most species are not as abundant today as they were prior to EuroAmerican settlement (Anderson & Keeley 2018). Chaparral shrublands also support many important wildlife species that are hunted for food or that provide materials (e.g., tools, ornamental use), such as mule deer (*Odocoileus hemionus*) and numerous bird, reptile, and insect species (Anderson 2005; Anderson & Keeley 2018; Keeley 2018).

Executive Summary

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

Chaparral Shrublands	**Rank**	**Confidence**
Sensitivity	Moderate	Moderate
Future Exposure	Moderate	Low
Adaptive Capacity	Moderate	High
Vulnerability	**Moderate**	**Moderate**

Sensitivity & Exposure Summary	
	<u>*Climate and climate-driven factors*</u>: • Precipitation timing, climatic water deficit, soil moisture, air temperature, heat waves <u>*Disturbance regimes*</u>: • Wildfire, disease <u>*Non-climate stressors*</u>: • Fire suppression activities (e.g., fuel breaks, prescribed fire, mastication), roads/highways/trails <u>*Other sensitivities*</u>: • Decoupling of plant and pollinator phenologies

Chaparral shrublands are sensitive to climate and climate-driven changes that alter moisture availability, including seasonal precipitation timing, soil moisture, and climatic water deficit. Changes in moisture availability will likely alter chaparral landscape distribution, species composition, survival, growth, and recruitment. Chaparral species are also sensitive to air temperature increases and heat waves; increasing air temperatures will alter chaparral landscape distribution, species composition, and stand growth, while heat waves may exacerbate the impacts of air temperature increases and increasing climatic water deficit and generate extreme fire weather. These climate changes may also shift plant phenologies, leading to a mismatch with pollinator life cycles and resultant impacts on chaparral reproduction, recruitment, and diversity.

Wildfire is one of the key disturbance regimes in chaparral shrublands, influencing successional dynamics and contributing to biodiversity across the landscape. While chaparral is generally fire-regime-adapted, increasing fire frequencies are likely to alter species composition and increase invasion of non-native annual species, potentially reducing biodiversity and impacting the provision of ecosystem services. Increasing fire frequencies may also facilitate vegetative type conversions from chaparral to non-native annual grasslands. In addition to fire, chaparral shrublands are affected by *Phytophthora* diseases, which cause elevated chaparral species mortality. Mortality rates may increase if climate change and/or human activity expands current disease ranges.

Chaparral shrublands are also affected by non-climate stressors, including fire suppression activities and roads, highways, and trails. Fire suppression and fuel reduction activities in northern California (e.g., fuel break creation, mastication, prescribed fire) physically alter stand structure and species composition, and contribute to non-native species invasion. Fire suppression also alters landscape vegetation and fuel characteristics (e.g., allows conifer encroachment into shrublands, permits shrub growth in forest understory), affecting chaparral distribution and landscape-scale fire regimes. Roads, highways, and trails fragment chaparral shrublands, limit seed dispersal opportunities by wildlife by negatively impacting small mammals, and promote invasion by non-native annual species. Additionally, vehicles can act as ignition sources, exacerbating changing fire regimes.

Adaptive Capacity Summary	*Factors that enhance adaptive capacity*: + Extensive habitat; non-serpentine chaparral fairly continuous, especially in inland areas + Disturbance-adapted vegetation community + Relatively high physical, topographical, functional group, and species diversity + Some climate-informed management opportunities (e.g., prescribed fire, restoration) *Factors that undermine adaptive capacity*: – Habitat integrity has been degraded by fire suppression and land use changes – Vulnerable to invasion and type conversion with increasing disturbance frequency – Serpentine chaparral not continuous and slow to recover from disturbance – Limited seed dispersal distances, which constrains recovery and migration opportunities – Low societal value relative to other habitats (e.g., valued less than timberlands, viewed as fire risk from the vantage of community fire protection)

Chaparral shrublands cover an extensive land area in northern California. Chaparral continuity is highest in inland locations, which generally demonstrate higher landscape heterogeneity and greater climate variability. Serpentine chaparral generally has lower habitat continuity than non-serpentine chaparral due to the patchy occurrence of ultramafic substrates. Chaparral shrubland integrity has been affected by fire suppression and land use changes. In general, chaparral shrublands are fairly resilient to disturbances such as drought and infrequent fire, but may be less resilient to increasing disturbance frequency, which promotes non-native plant invasion and facilitates vegetation type conversion. Chaparral recovery from disturbance is slower on serpentine soils and in drier areas. Overall habitat recovery post-disturbance and migration potential is limited by short seed dispersal distances; for serpentine chaparral, habitat migration in the face of climate change will also likely be restricted by larger dispersal distances and the availability of serpentine soils in climatically suitable areas. Relatively high topographical, functional group, and species diversity generally will increase resilience of chaparral to climatic changes, although some species may be particularly vulnerable to climate change (e.g., obligate seeding and obligate resprouting species, endemic and rare plants, serpentine pollinators). Although chaparral provides many critical ecosystem services, it generally has lower societal value relative to other habitats (e.g., when compared to economic value of timberland), although efforts are underway to change this perception.

Sensitivity and Exposure

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

Under some climate and vegetation projections, shrubland extent is projected to decline in northern California and across the state by the end of the century (Lenihan et al. 2008). Under cooler and wetter conditions, forest encroachment may successionally replace shrublands, while under warmer and drier conditions, increasing fire and reductions in effective moisture could drive shrubland replacement by grasslands (Lenihan et al. 2008). In general, fire is expected to play an important role in mediating vegetation response to climate change. For example, fire may accelerate changes from shrub to grassland in drier conditions, particularly at lower elevations (Lenihan et al. 2008; Estes 2013). Under wetter conditions, fire may combat some conifer encroachment into chaparral areas, and may even drive shifts from forests to chaparral at higher elevations, expanding habitat availability in some areas (Lenihan et al. 2008). Studies in adjacent geographies project similar fire-mediated effects. For example, in southern Oregon, a rise in high-severity fire paired with warmer and drier conditions is projected to expand chaparral by replacing some dry and mesic forest areas (Case et al. 2018; Southwest Oregon Adaptation Partnership 2018) .

Recent habitat distribution modeling by Thorne et al. (2016, 2017) projects areas of climatic exposure (i.e., areas likely to experience significant environmental stress and/or experience type conversion) and climatic suitability for both mixed and montane chaparral shrublands by the end of the century under different climate change and emissions scenarios (see table below).[1] Models project warmer and wetter future conditions to create more climatically

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

exposed chaparral areas in northern California than warmer and drier conditions, and for higher emissions scenarios to generate more climatically exposed chaparral habitat than lower emissions scenarios (Thorne et al. 2016, 2017). In northern California, mixed chaparral areas likely to become significantly climatically exposed under most future climate scenarios include the northern Sierra Nevada foothills, northern segments of the Northern California Interior Coast Ranges, and the northwest corner of the Klamath Mountains (see table below; Thorne et al. 2016).[2] For montane chaparral, the only areas projected to become significantly climatically exposed under most future climate scenarios include small portions of the northwest corner of the Northern California Coast Range, and the very northern segments of the northern Sierra Nevada Foothills and Northern California Interior Coast Range (Thorne et al. 2016).

Potential Changes in Habitat Distribution by 2100
• *Mixed chaparral:* 16–42% of the state-wide current vegetation distribution is projected to experience an increase in climatic stress, while 40–66% will remain within climatically suitable areas; 17–47% of the projected climatically suitable habitat will be in newly suitable areas • *Montane chaparral:* 3–18% of the state-wide current vegetation distribution is projected to experience an increase in climatic stress, while 59–86% will remain within climatically suitable areas; 9–26% of the projected climatically suitable habitat will be in newly suitable areas • *Serpentine chaparral:* Serpentine species will have to make large "jumps" to reach future suitable habitat, and will require access to suitable substrate in newly suitable habitats
Source(s): Damschen et al. 2012; Thorne et al. 2016

However, a substantial amount of existing habitat area of mixed and montane chaparral is projected to remain climatically suitable in northern California by the end of the century (see table below), and may represent areas of climatic refugia for these habitat types (Thorne et al. 2016). Under projected future scenarios, areas of major climatic refugia include the central portion of the Northern California Interior Coast Ranges (for mixed chaparral) and the central Klamath Mountains (for montane chaparral); the southern Cascades also represent possible refugia under lower greenhouse gas emissions scenarios (Thorne et al. 2016). In addition, as the climate changes, areas not currently occupied by chaparral shrublands are projected to become newly climatically suitable, representing migration opportunities (Thorne et al. 2016). In northern California, newly suitable areas for mixed chaparral may include higher elevations in the northern Sierra Nevada Foothills, more western and northerly areas along the Northern California Coast and in the Northern California Coast Ranges, and expanded area in the southern Cascades and southern Klamath Mountains (Thorne et al. 2016). Newly climatically-suitable areas for montane chaparral may occur mainly to the east and west of current suitable area in the central Northern California Coast Range (Thorne et al. 2016).

Predicting future persistence and distribution for serpentine chaparral is difficult. As soil specialists, endemic serpentine species are geographically tied to appropriate soils; persistence

[2] Regional areas identified as being climatically exposed, climatically suitable, and/or projected to become climatically suitable in the future were inferred by comparing Thorne et al. (2016, 2017) habitat distribution maps to modified Jepson ecoregions (Hickman 1993).

in existing locations will largely require maintenance of suitable climatic conditions or species capacity to rapidly adapt to changing conditions (Damschen et al. 2012). Migration opportunities to new locations will be limited by soil specificity and the natural isolation and fragmentation of unique ultramafic soil habitats (Damschen et al. 2012). Soil specialists would have to "jump" to new habitat areas as the climate changes, as soil specificity limits their ability to migrate "through" areas (Damschen et al. 2012). Additionally, soil specialists are likely to face larger dispersal distances than soil generalist species, and may not have access to appropriate substrate in areas projected to become climatically suitable (Damschen et al. 2012).

Sensitivity and future exposure to climate and climate-driven factors

Regional experts evaluated chaparral shrublands as having low-moderate sensitivity to climate and climate-driven factors (moderate confidence in evaluation), with an overall moderate-high future exposure to these factors within the study region (moderate confidence). Key climatic factors that affect chaparral shrublands include seasonal precipitation timing, climatic water deficit, soil moisture, air temperature, and heat waves.[3]

<u>Precipitation timing</u>

In general, chaparral shrublands, particularly those in central and southern California, are adapted to seasonal summer drought characteristic of Mediterranean-type climates (Keeley & Davis 2007; Halsey & Keeley 2016). However, shifts in seasonal precipitation timing, especially a contraction of the rainy season (Pierce et al. 2018; Swain et al. 2018), may affect recruitment and community composition by altering soil moisture profiles (Keeley & Davis 2007; Cornwell et al. 2012), especially when co-occurring with other climatic changes that affect watershed hydrology and overall moisture availability (e.g., increased climatic water deficit, reduced snowpack leading to earlier soil drying, and altered runoff and recharge patterns; Knowles et al. 2006; Thorne et al. 2015). Exact impacts will vary depending on individual species' physiological limits, but in general, facultative seeding chaparral species would likely be favored by declines in precipitation relative to obligate seeding or obligate resprouting chaparral species (Cornwell et al. 2012). The most significant impacts at the community level are likely to occur if shifts in precipitation timing enhance summer and prolong drought stress (i.e., if spring precipitation declines and/or last spring rains occur earlier, and fall rains decline and/or first fall rains occur later), as summer drought stress is known to limit chaparral seedling recruitment (Keeley & Davis 2007). In addition, a study in southern California found that chaparral communities were most vulnerable to cavitation (i.e., xylem collapse), and subsequent hydraulic failure in response to moisture stress during the early growing season (Jacobsen et al. 2014).

However, shifts in seasonal precipitation timing that increase moisture levels during the warm season may also have negative impacts on chaparral communities. For example, higher soil moisture during the warm season may increase the vulnerability of some chaparral species to pathogenic soil fungi (e.g., *Phytophthora* spp.; Meentemeyer et al. 2004; Rooney-Latham et al.

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

2015; Swiecki et al. 2017). Higher summer soil moisture may also enhance non-native species invasion and growth (Keeley & Syphard 2015). Shifting precipitation regimes may also alter fire activity by influencing the relative abundance of non-native herbaceous vegetation, with higher precipitation enhancing herbaceous growth and consequently increasing the availability of fine fuels (Keeley & Syphard 2015).

<table>
<tr><th colspan="2">Regional Precipitation Trends[4]</th></tr>
<tr><td>Historical & current trends:
• 7.2–9.4 cm (2.8–3.7 in) increase in mean annual precipitation between 1900 and 2009 for the Northwestern California and Southern Cascade ecoregions (Rapacciuolo et al. 2014)</td><td>Projected future trends:
• 20% decrease to 34% increase in mean annual precipitation by 2100 (compared to 1951–1980) for the North Coast, Northern Coast Range, Northern Interior Coast Range, Klamath Mountain, and Southern Cascade ecoregions (Flint et al. 2013; Flint & Flint 2014)[5]
• Seasonal changes are projected to be more significant as the wet season becomes wetter and shorter (i.e., later onset of fall rains and earlier onset of summer drought) and the dry season becomes drier and longer (Pierce et al. 2018; Swain et al. 2018)
• Overall, interannual variability is expected to increase (Pierce et al. 2018; Swain et al. 2018)</td></tr>
<tr><th colspan="2">Summary of Potential Impacts on Habitat (see text for citations)</th></tr>
<tr><td colspan="2">• Altered species composition, including the potential for increased dominance of facultative seeding species under drier conditions
• Limited recruitment and increased vulnerability to cavitation and hydrologic failure if spring precipitation decreases
• Increased vulnerability to pathogenic soil fungi and non-native species invasion and growth if warm season precipitation increases
• Altered fire risk due to fine fuel growth</td></tr>
</table>

<u>Climatic water deficit and soil moisture</u>

Water availability and corresponding moisture stress affects chaparral community composition, recruitment, plant survival, growth, and phenology (Keeley & Davis 2007; Vasey et al. 2012; Willis et al. 2013; Jacobsen et al. 2014; Jacobsen & Pratt 2018; Small et al. 2018). Northern

[4] Trends in climate factors and natural disturbance regimes presented in this and subsequent summary tables are not habitat-specific; rather, they represent broad trends and future projections for the study region. The precipitation, temperature, climatic water deficit, and snowpack projections for this project are derived from the Basin Characterization Model, which uses modified Jepson ecoregions (Flint et al. 2013; Flint & Flint 2014). Projections for all other factors are based on a review of relevant studies in the scientific literature. For this project, exposure was evaluated by calculating the magnitude and direction of projected change within the modified Jepson ecoregions that include habitat distribution within the study geography.
[5] Projections for changes in annual and seasonal precipitation by ecoregion can be found in the full climate impacts table (https://bit.ly/2LHgZaG).

California is projected to experience significant reductions in climatic water deficit (CWD) over coming century (Flint et al. 2013; Flint & Flint 2014; Thorne et al. 2015). CWD is a "plant-relevant" measurement of moisture stress that takes into account the interaction between water (precipitation) and energy (temperature; Stephenson 1998).[6] The balance between water supply and demand in California shifts over the course of the year, with CWD increasing as soil moisture from the winter rains is depleted by late spring and evapotranspiration increases in warmer months (Stephenson 1998). Due to increased evaporative demand as temperatures rise, even areas where precipitation may increase are expected to see a rise in climatic water deficit under future climate conditions (Thorne et al. 2015; Micheli et al. 2018). However, factors such as soil depth and drainage significantly affect the water-holding capacity of soil, and topographic features such as north-facing slopes can mediate CWD at a site-level scale (Dobrowski 2011; Flint et al. 2013).[7] Local hydrological climate microrefugia may also exist because of localized groundwater flows, but these are difficult to identify based on landscape position alone (McLaughlin et al. 2017).

Changes in moisture availability are likely to alter chaparral landscape distribution and community composition (Keeley & Davis 2007). For example, at the landscape scale, and depending on elevation and fire history, chaparral may be replaced by oak woodlands and coniferous forests on more mesic sites (Keeley & Davis 2007). Soil moisture changes are also likely to change chaparral community composition because soil moisture drives differing species assemblages and functional group dominance in xeric versus more mesic sites (Keeley & Davis 2007). For example, drier areas, including serpentine sites, typically feature obligate seeding chaparral species and facultative seeders (Jacobsen et al. 2007, 2014; Cornwell et al. 2012; Halsey & Keeley 2016). These species have higher cavitation resistance and also tend to have lower specific leaf area, enabling survival during short periods of high water stress (i.e., seasonal drought; Jacobsen et al. 2007, 2014; Anacker et al. 2011), although they may be more susceptible to long-term drought due to shallow rooting systems (see discussion of drought below; Jacobsen & Pratt 2018). Comparatively, moister (more mesic areas) tend to feature obligate resprouting species (Cornwell et al. 2012; Halsey & Keeley 2016), which function as dehydration-avoiders by having deeper root systems that allow enhanced access to water during dry periods (Burk 1978; Meentemeyer & Moody 2002).

Decreased soil moisture may alter chaparral recruitment, as summer drought stress is known to limit seedling recruitment in other parts of California (Keeley & Davis 2007). In southwestern Oregon, higher summer soil moisture (in comparison to southern California) was hypothesized to contribute to the germination and recruitment of buckbrush (*Ceanothus cuneatus*) during fire-free intervals (a previously unknown occurrence; Duren & Muir 2010). However, such

[6] CWD, which is calculated as potential evapotranspiration (PET) minus actual evapotranspiration (AET), measures the degree to which the impact of local atmospheric conditions (particularly air temperature and relative humidity) on plants and soil exceeds available moisture (Stephenson 1998).

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

moisture-mediated recruitment events may become less frequent with future reductions in moisture availability.

Declining moisture availability may also reduce individual plant survival. Water stress as a result of low soil moisture availability can cause xylem cavitation, and persistent water stress (i.e., drought) can lead to hydraulic failure, chaparral dieback, and ultimately, plant mortality (Jacobsen et al. 2014). More mesic chaparral areas, mesic-adapted ecotypes, and shallow-rooted obligate seeding species may be particularly vulnerable to declining soil moisture (Vasey et al. 2012; Jacobsen et al. 2014; Jacobsen & Pratt 2018). For example, in southern California, Jacobsen et al. (2014) found that individuals in mesic chaparral areas were more sensitive to cavitation and subsequent hydraulic failure in response to moisture stress than chaparral communities on more xeric sites. Similarly, obligate-seeding chaparral species in mesic maritime areas along California's Central Coast were found to be more vulnerable to vascular cavitation in response to water stress than both obligate-seeding species found in drier interior areas and facultative seeding species found in the same coastal locations (Vasey et al. 2012). Additionally, shallow-rooting species (e.g., obligate seeders) in southern California, despite their adaptations to short-term moisture stress and tendency to occupy more xeric areas, have been found to be particularly vulnerable to mortality during longer-term, higher-intensity drought events (Jacobsen & Pratt 2018). Cumulatively, moisture stress-induced mortality and recruitment impacts are likely to affect chaparral community composition (Jacobsen & Pratt 2018).

Shifts in soil moisture may also alter plant growth and phenology. Seasonal shrubland growth across California from 2007–2016 was positively correlated with rainfall in the preceding 4–12 months, indicating that plant growth is responsive to moisture availability, and higher moisture translates to higher growth amplitudes (Small et al. 2018). Even following severe, multi-year drought, shrubland plant growth across the state rebounded with the return of higher precipitation, indicating some resilience to drought events (Small et al. 2018). Both overstory and understory plants are likely to be affected by changing moisture availability; for example, chamise has been shown to alter leaf production and maintenance in response to changing moisture conditions (Narog 2008), and understory cover also responds to soil moisture changes, even in mature chaparral stands where understory cover is low (Halsey & Keeley 2016). Soil moisture changes may also alter plant phenology; for example, a study in southern California found chaparral phenology to exhibit high interannual variability in response to precipitation and soil moisture declines (Willis et al. 2013).

Regional Climatic Water Deficit (CWD) & Soil Moisture Trends	
Historical & current trends: • 1.1 cm (0.4 in) decrease to 0.4 cm (0.2 in) increase in average annual CWD between 1900 and 2009 for the Northwestern California and Southern Cascade ecoregions (Rapacciuolo et al. 2014)	*Projected future trends:* • Increases in average annual CWD by 2100 (compared to 1951–1980; Flint et al. 2013; Flint & Flint 2014): ○ 9–29% increase on the North Coast

Regional Climatic Water Deficit (CWD) & Soil Moisture Trends	
• No trends available for soil moisture	○ 7–24% increase in the Northern Coast Range ○ 5–16% increase in the Northern Interior Coast Range ○ 10–32% increase in the Klamath Mountains ○ 16–43% increase in the Southern Cascades • Increased CWD and decreased top-level soil moisture is likely even if precipitation increases due to temperature-related changes in evaporative demand (Thorne et al. 2015; Micheli et al. 2018; Pierce et al. 2018)
Summary of Potential Impacts on Habitat *(see text for citations)*	
• Altered landscape distribution and site-level species composition • Reduced plant growth and recruitment, although plant growth rebounds with precipitation • Altered plant phenology • Increased vulnerability to chaparral dieback and mortality, especially in mesic areas	

Air temperature and heat waves

Air temperature influences chaparral landscape distribution, community composition, and stand development (Keeley & Davis 2007). Adult shrubs are vulnerable to winter freeze injury, which may historically have limited chaparral persistence at higher elevations in northern California (Keeley & Davis 2007). At a local level, species composition is controlled by the freeze tolerances of individual species interacting with patterns of cold air drainage (Keeley & Davis 2007). Increasing air temperatures may facilitate expansion of chaparral to higher elevation locations and/or increase the local abundance of more cold-intolerant chaparral species by creating more mild conditions (Keeley & Davis 2007). In particular, warmer winter air temperatures may increase abundance of facultative seeding chaparral species, which currently have higher abundance in central and southern California (Cornwell et al. 2012). Increasing air temperatures may also accelerate the rate of montane chaparral stand development by increasing favorable growing conditions; stand development has historically been slow due to short growing seasons, cold air, and snow (Risser & Fry 1988). Increasing air temperatures will also drive increased climatic water deficit (Cook et al. 2014; Thorne et al. 2015; McLaughlin et al. 2017).

Heat waves generally exacerbate other climatic conditions that can impact chaparral communities. For example, heat waves dry potential fuels and generally create more extreme fire weather, enhancing wildfire risk (Micheli et al. 2016; Parker et al. 2016).

Regional Air Temperature & Heat Wave Trends	
Historical & current trends: • 0.03°C (0.05°F) decrease to 0.2°C (0.4°F) increase in the average annual temperature	*Projected future trends:* • 2.2–5.5°C (4.0–9.9°F) increase in the average annual temperature by 2100 (compared to

Regional Air Temperature & Heat Wave Trends	
between 1900 and 2009 for the Northwestern California and Southern Cascade ecoregions (Rapacciuolo et al. 2014) 　○ No seasonal temperature trends available • Increase in the frequency of humid nighttime events over the past several decades (Gershunov & Guirguis 2012) • High interannual and interdecadal variability in heat waves (Gershunov & Guirguis 2012)	1951–1980) for the North Coast, Northern Coast Range, Northern Interior Coast Range, Klamath Mountain, and Southern Cascade ecoregions (Flint et al. 2013; Flint & Flint 2014) 　○ 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 • Increased heat waves, with the greatest increase in humid nighttime heat waves and in coastal areas (Gershunov & Guirguis 2012) • 2–6°C (3.6–10.8°F) increase in the temperature of the hottest day of the year by 2100 (Pierce et al. 2018)

Summary of Potential Impacts on Habitat *(see text for citations)*
• Potential habitat expansion to higher elevations • Altered species composition: increased abundance of cold-intolerant species (e.g., facultative seeders) • Accelerated stand development • Enhanced fire risk due to extreme fire weather

Sensitivity and future exposure to changes in natural disturbance regimes

Regional experts evaluated chaparral shrublands as having moderate sensitivity to changes in natural disturbance regimes (moderate confidence in evaluation), with an overall low-moderate future exposure to these stressors within the study region (low confidence). Wildfire and disease are the key natural disturbance regimes that affect chaparral shrublands.[8]

Wildfire

Wildfire exerts a strong influence on vegetation dynamics in chaparral shrublands (Keeley et al. 2005a; Duren & Muir 2010; Odion et al. 2010; Duren et al. 2012; Halsey & Keeley 2016; Airey Lauvaux et al. 2016; Keeley 2018). With adequate fire return intervals, wildfire generally resets stand succession and alters chaparral biodiversity across the landscape (Keeley et al. 2005b). For example, in montane areas, fire creates opportunities for montane chaparral communities to establish from soil seedbanks within forested areas (Knapp et al. 2012), and chaparral can then remain dominant for a long time (about 50 years; Risser & Fry 1988). In the Klamath-Siskiyou and southern Cascade regions of northern California and southwestern Oregon, montane chaparral was an “alternative stable state” under pre-EuroAmerican settlement conditions, where fire, both natural and resulting from Native American management, maintained shrublands as multi-aged systems by preventing colonization by conifers (Duren & Muir 2010; Odion et al. 2010; Duren et al. 2012; Airey Lauvaux et al. 2016).

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

Native American tribes historically used fire to manage chaparral shrublands for a variety of purposes, including increasing the abundance and productivity of harvestable plant species, increasing shrub shoot production for basketry materials, and increasing habitat suitability for game species, among other reasons (Anderson & Rowney 1999; Anderson 2005; Anderson & Rosenthal 2015; Anderson & Keeley 2018). For example, several edible herbaceous chaparral species require fire for germination and flowering, including farewell-to-spring, western redbud (Anderson 2005; Keeley 2018), and blue dicks (*Dichelostemma capitatum*; Anderson & Rowney 1999). Native American tribes also historically used fire to mitigate chaparral encroachment on other habitats that support culturally important species (Anderson 2005; Norgaard et al. 2016). In general, tribes managed cultural burning to maintain chaparral in diverse age and size classes, supporting high spatial, structural, successional, and biotic diversity (Anderson & Keeley 2018).

Chaparral systems today experience both natural and human-ignited fires. Natural lightning ignitions occur primarily in the spring and summer (Jin et al. 2015). Volcanic events can also serve as natural ignitions sources (Vuln. Assessment Reviewers, pers. comm., 2018). However, human ignitions are now the primary ignition source for regional fires, and human settlement patterns will continue to be the most significant determinants of fire risk in chaparral communities in the future (Syphard et al. 2007, 2017; Halsey & Keeley 2016; Balch et al. 2017; CAL FIRE 2018).

Chaparral species are largely fire-regime-adapted, and habitat recovery after fire is typically rapid (Parker & Kelly 1989; Keeley & Davis 2007). In the first year or two, annuals are usually the dominant life-form, including a group of taxa known as pyro-endemics, which are entirely fire-dependent species that occur in the understory and vegetation openings. Example pyro-endemics include whispering bells (*Emmenanthe penduliflora*), scorpionweed (*Phacelia* spp.), dense false gilia (*Allophyllum gilioides*), and fire poppy (*Papaver californicum*; Halsey & Keeley 2016; Keeley 2018). Obligate- and facultative-seeding species, which largely depend on fire for germination (although see Duren & Muir 2010), follow these annual herbaceous species and quickly reestablish across the burned landscape (Keeley et al. 2005a; Keeley & Davis 2007), followed by slower recruitment of obligate resprouters (Keeley et al. 2005b). However, post-fire recovery is also influenced by soil moisture, topography, soil type, and other factors, contributing to variable recovery rates in different geographic locations (Keeley et al. 2005a; Narog 2008). For example, recovery is typically faster in mesic areas (England 1988). Serpentine sites are slower to recover than non-serpentine areas (Safford & Harrison 2004). Based on short-term fire recovery rates in a study in northern California, chaparral community recovery was projected to take 4-5 years on non-serpentine sites and 12–15 years on serpentine sites (Safford & Harrison 2004).

Fires in chaparral systems are typically high-intensity crown fires, with rate of spread depending on wind, weather, topography, and available live and dead fuel, and past burn patterns (Keeley & Davis 2007). There is little consensus on what the 'natural' fire cycle is for chaparral (Barro & Conard 1991). Fire return intervals are difficult to precisely determine due to the stand-replacing nature of fire in this habitat type, but estimated historic fire return intervals in mixed chaparral in most of California range from 30–150+ years depending on location (Safford & Harrison 2004; Keeley et al. 2005b; Nagel & Taylor 2005; Duren & Muir 2010; Van de Water & Safford 2011; Safford & Van de Water 2014; Halsey & Keeley 2016; Airey Lauvaux et al. 2016). Montane chaparral in northern California and southwestern Oregon has been documented as having had longer fire-return intervals than adjacent forests before forest management for fire suppression began (Nagel & Taylor 2005; Duren & Muir 2010; Odion et al. 2010; Duren et al. 2012). Fires in montane chaparral exhibited mean fire-return intervals of approximately 30 years in the northern Sierra Nevada (Nagel and Taylor 2005) and southern Cascades (Airey Lauvaux et al. 2016), but some shrubland stands in southwestern Oregon showed no evidence of fire in more than 130 years (Duren & Muir 2010). In southwestern Oregon, chaparral stands demonstrated heterogeneous age structures due to individual plants that survived one or more previous fires and plants that germinated and grew during fire-free intervals, indicating that montane chaparral can be a multi-aged, stable community type in the region (Duren & Muir 2010). Serpentine chaparral typically experiences longer fire return intervals and more variable fire severity because low productivity soils limit shrub cover, plant productivity, and associated fuel accumulation and connectivity (Safford & Harrison 2004; Keeley & Davis 2007).

Fire intervals are projected to decrease in northern California as a result of climate change (Westerling et al. 2011; Mann et al. 2016; Parks et al. 2016). Changes in the timing and amount of precipitation will alter fuel moisture and the availability and continuity of fine fuel, influencing the likelihood of an ignition becoming a wildfire (Jin et al. 2015). Alterations in wind patterns and velocity, lightening frequency, and relative humidity will also exacerbate ignition risk, rate of fire spread, and area burned from both lightning and human-sourced ignitions (Jin et al. 2015; Keeley & Safford 2016). Warm, dry Diablo winds in northern California, which function similarly to Santa Ana winds in southern California (Miller & Schlegel 2006), contribute to faster fire spread (Jin et al. 2015).

Although chaparral is adapted to fire, too frequent fire can cause shifts in species composition and habitat structure by inhibiting regeneration (Risser & Fry 1988; Haidinger & Keeley 1993; Keeley 1995; Cornwell et al. 2012). For example, shortened fire intervals (<10–20 years) may eliminate obligate seeding species by killing recruits and depleting their seedbank. Fire stimulates dormant seed germination, but if a subsequent fire kills seedlings before they reproduce, obligate seeding chaparral species can be locally extirpated. This reduces biodiversity and results in a shrubland community comprised predominately of obligate resprouting and facultative seeding species (Risser & Fry 1988; Haidinger & Keeley 1993; Keeley & Davis 2007; Cornwell et al. 2012; Halsey & Keeley 2016). Shortened fire return intervals may also limit recruitment of obligate resprouters, which also depends on sufficient fire-free periods (Keeley 1991; Haidinger & Keeley 1993; Keeley et al. 2005b; Cornwell et al. 2012; Keeley & Brennan 2012). Across California, facultative seeding chaparral species have higher relative cover than obligate seeding and obligate resprouting species in areas experiencing higher fire frequencies, indicating their relative dominance in northern California may increase if fire becomes more frequent and other climatic conditions are suitable (e.g., temperature, precipitation; Cornwell et al. 2012).

Impaired native shrub regeneration resulting from more frequent fires may facilitate establishment and eventual dominance of non-native species (Keeley et al. 2011; Keeley & Brennan 2012). For example, non-native medusahead (*Elymus caput-medusae*) and yellow starthistle (*Centaurea solstitialis*) can make up a significant portion of post-fire vegetation assemblages, especially in areas with reduced post-fire establishment of native woody plants (Keeley & Davis 2007; Halsey & Keeley 2016). These and other non-native grass and forb species comprise easily ignited fuels and feature more frequent fire return intervals, creating a positive feedback loop that increases the likelihood of vegetation type conversion (Haidinger & Keeley 1993; Keeley & Davis 2007; Keeley et al. 2011; Keeley & Brennan 2012). Vegetation type conversion from chaparral to non-native annual grassland is widespread (Haidinger & Keeley 1993; Lenihan et al. 2008; Keeley & Brennan 2012; Halsey & Keeley 2016).

Conversely, in some areas, increasing fire frequency and severity in adjacent forest systems may facilitate chaparral dominance in lieu of forest vegetation, including future type conversion from forestlands to montane chaparral (Nagel & Taylor 2005; Keeley & Davis 2007; Cornwell et al. 2012; Airey Lauvaux et al. 2016). Forest types typically burn more frequently, but less severely, than chaparral systems. Interpretation of historical photos and fire-scar evidence indicates that historical landscapes demonstrated high-severity fires in chaparral that eliminated encroaching forest species, while adjacent forests showed more frequent, lower-severity fires that suppressed understory vegetation (Nagel and Taylor 2005; Airey Lauvaux et al. 2016). However, high-severity forest wildfires may allow the colonization or release of shrubs that perpetuate the occurrence of severe fires (Coppoletta et al. 2016), potentially contributing to vegetation type conversion from forest to chaparral (Crotteau et al. 2013; Airey Lauvaux et al. 2016). This process has been observed in several locations in northern California, including with mixed conifer and montane chaparral systems in the Lake Tahoe basin (Nagel & Taylor 2005), Lassen National Park (Airey Lauvaux et al. 2016), and in Lassen National Forest (Crotteau et al. 2013). In the northern Sierra Nevada, high-severity fire that favors shrub development often leads to subsequent high-severity fire, maintaining landscapes as shrublands (Coppoletta et al. 2016). Vegetation transitions such as these will alter the type and availability of tribal food and material resources by altering the relative abundance of different montane habitat types (Norgaard et al. 2016).

Faunal responses to wildfire in chaparral have received little direct study (Halsey & Keeley 2016). Many bird species respond positively to shrub-related habitat elements. Studies in the shrub-enriched forestlands of the Klamath-Siskiyou region demonstrate that abundant post-fire shrub regrowth is important for maintaining habitat values for shrub-associated birds (Fontaine et al. 2009; Fontaine & Kennedy 2012). Similarly, some wildlife species (e.g., deer) respond positively to mixed severity fire regimes. Fire plays an important role in creating early-successional habitat with higher abundance and diversity of regenerating woody shrubs and understory herbaceous species to browse, as well as creating mosaic habitat conditions at the landscape scale (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008; Innes 2011). However, increasing fire frequency and size and associated wildfire-driven shrub loss or type conversion may adversely affect habitat quality for some chaparral wildlife species (Innes 2011, 2013; Jennings 2018).

<table>
<tr><th colspan="2">Regional Wildfire Trends</th></tr>
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Historical & current trends:

- 85% of U.S. Forest Service lands in northern California are burning less frequently compared to pre-1850 fire return intervals, largely due to fire suppression (Safford & Van de Water 2014)
- Fire size and total area burned increased on U.S. Forest Service lands in northwestern California between 1910-2008, with the highest values occurring after 2000 (Miller et al. 2012)
- Changes in large fires (over 400 ha) in the inland northern California/Sierra Nevada region since the 1970s (Westerling 2016):
  - 184–274% increase in frequency
  - 270–492% increase in total area burned
  - 215% increase in length of the fire season
- Changes in fire size, area burned, and fire frequency over the past several decades remain well below historical tribally-influenced frequency and extent of burning in California (Stephens et al. 2007)
- No significant trends in the average areal proportion of high-severity fire were documented in northwestern CA from 1984–2008 (Miller et al. 2012; Parks et al. 2015; Law & Waring 2015; Keyser & Westerling 2017)
  - The relatively short period of record for fire severity data may obscure long-term trends
  - To date, there are no peer-reviewed studies on trends in northern California fire severity that include data from the last ten years

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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)
  - Greatest increases in burned area (up to 400%) occur in montane forested areas in northern California (Westerling et al. 2011; Westerling 2018)
  - Less significant increases or possible decrease along the North Coast (Westerling et al. 2011)
- Little projected change in fire severity in northwestern California by 2050 in models based solely on historical fire-climate relationships (Parks et al. 2016)
  - However, human activity and fuel buildup from decades of fire suppression have altered historical fire-climate relationships (Taylor et al. 2016; Syphard et al. 2017; Wahl et al. 2019), and projections that incorporate these factors suggest that more significant increases in fire severity and size may occur (Mann et al. 2016; Wahl et al. 2019)
- The majority of impacts to natural and human ecosystems come from extreme fire events (i.e., fires that have a low probability of occurring in any given place and time), which are likely to increase over the coming century (Westerling 2018)
  - Generally, these patterns are not well-represented in studies that evaluate indices of mean fire size, intensity/severity, etc.

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<tr><th colspan="2">Summary of Potential Impacts on Habitat (see text for citations)</th></tr>
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- *Immediate*:
  - Resets existing stand succession
  - Creates opportunities for chaparral to develop from soil seedbanks in forested areas
- *Short-term (~2-year)*:
  - Increased dominance of herbaceous species for 1–2 years, eventually followed by obligate-seeding and obligate-resprouting species

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Regional Wildfire Trends
○ Increased stand biodiversity due to abundance of annual species ○ Increased stand biodiversity due to abundance of annual species • *Long-term*: ○ Rapid recovery of communities with sufficient fire return intervals, though recovery is slower on drier sites and serpentine areas

Disease

Disease is not currently known to be a major disturbance regime in northern California chaparral habitats (Vuln. Assessment Workshop, pers. comm., 2017), but climate change may alter disease incidence and/or severity (Kliejunas 2011). Some chaparral species including pallid manzanita (*Arctostaphylos pallida*), common manzanita (*A. manzanita*), toyon, and coffeeberry are vulnerable to soil-borne *Phytophthora* pathogens (e.g., *P. ramorum, P. tentaculata*), which cause plant mortality via root and crown rot (Meentemeyer et al. 2004; Rooney-Latham et al. 2015; Swiecki et al. 2017). *Phytophthora* pathogens are often found in nursery-grown stock used for restoration efforts and can be spread via contaminated soil (Rooney-Latham et al. 2015; Swiecki et al. 2017). Climate change may create more favorable conditions for these pathogens. For example, warmer temperatures may enhance *Phytophthora* germination rates, although germination rates are also controlled by moisture (Kliejunas 2011). Exact impacts of climate- and human-mediated increases in disease incidence or severity in northern California are unknown, but would likely include elevated mortality in some chaparral species, altering chaparral species composition, stand structure (Kliejunas 2011; Swiecki et al. 2017), and potentially, ecosystem functioning (Avila et al. 2019).

Regional Disease Trends	
Historical & current trends: • No disease trends are available	*Projected future trends:* • Altered disease incidence and/or severity in pathogens such as *Phytophthora* (Kliejunas 2011)
Summary of Potential Impacts on Habitat *(see text for citations)*	
• Increased mortality if climate change or human introductions increase disease frequency or severity • Increased disease mortality would likely alter species composition, stand structure, and ecosystem functioning	

Sensitivity and current exposure to non-climate stressors

Regional experts evaluated chaparral shrublands as having low-moderate sensitivity to non-climate stressors (high confidence in evaluation), with an overall low-moderate current exposure to these stressors within the study region (high confidence). Key non-climate stressors that affect chaparral shrublands include fire suppression activities and roads/highways/trails.[9]

Fire suppression activities

Chaparral is widely regarded as a hazardous land cover type because it typically burns with high energy release (Parker et al. 2016). In consequence, chaparral is often identified as a target for significant fire suppression and fuel treatment activities (Potts & Stephens 2009; Potts et al. 2010; Parker et al. 2016; Wilkin et al. 2017; Safford et al. 2018a). Activities vary according to setting, but many wildland treatments consist primarily of fuelbreaks, as well as prescribed fire and mastication (Potts & Stephens 2009; Potts et al. 2010; Wilkin et al. 2017). Effects of these treatments on the structure and composition of northern California chaparral ecosystems differ only slightly (e.g., relative to mastication, prescribed fire may result in initially lower cover of non-native annual grasses). All of the treatments affect the biotic structure of the chaparral community without substantially reducing long-term fuel availability (Potts & Stephens 2009; Potts et al. 2010; Wilkin et al. 2017). Maintaining long-term fuel reduction benefits (i.e., to reduce fire risk to adjacent wildland-urban interface structures) would likely require re-treatment intervals (i.e., disturbance intervals) that are too frequent to avoid detrimental ecological impacts such as vegetation type conversion and/or significant species composition changes (as discussed above with too-frequent fire; Wilkin et al. 2017).[10]

Fire suppression has reduced the occurrence and extent of fire across large portions of the landscape in northern California (Safford & Van de Water 2014; Steel et al. 2015; Norgaard et al. 2016). Relative to forested habitats, a prolonged lack of fire has not significantly altered chaparral stand structure in northern California or southwestern Oregon (Duren & Muir 2010). Similarly, in southern Sierra Nevada foothill chaparral, fire exclusion contributed to only minor structural stand changes via the loss of mature obligate-seeding buckbrush (Keeley et al. 2005b). In general, chaparral is thought to be resilient to long fire-free periods, although some soil seed bank deterioration may occur, affecting the abundance of post-fire recruits (Keeley et al. 2005b). Additionally, very old stands may be more vulnerable to non-native annual invasion if any key shrub canopy species senesce and die (i.e., as occurred with buckbrush in the southern Sierra Nevada; Keeley et al. 2005b). Fire exclusion can also lead to declines in stand diversity through natural successional processes, as herbaceous and subshrub species are excluded by increased shade from the development of a closed canopy (England 1988). In general, changes in vegetation composition, structure, productivity, and nutrient availability in the absence of fire may affect chaparral wildlife populations, including deer, small mammals, birds, and reptiles (Humple & Burnett 2010; Innes 2013). Additionally, fire exclusion has reduced the value of chaparral stands to Native American tribes by affecting the availability and

[9] Non-climate stressors presented are those ranked as having a moderate or higher impact on this habitat type; additional non-climate stressors that may influence the habitat to a lesser degree include residential and commercial development, invasive and other problematic species, pollution (i.e., nitrogen deposition), and livestock grazing.

[10] Refer to the section on management capacity for a more complete discussion of future climate-informed management opportunities related to landscape development patterns, prescribed fire, and other fuel treatments for chaparral habitats.

quality of food, medicinal, and material resources (Norgaard et al. 2016; Anderson & Keeley 2018).

Fire suppression may also change chaparral distribution on the landscape through successional mechanisms. For example, the absence of fire in grassland systems may increase shrubland extent by allowing woody vegetation encroachment, particularly if occurring in tandem with wet years (Cornwell et al. 2012). Alternatively, fire exclusion may cause chaparral stands to be colonized by conifers or oak woodlands; as a result, chaparral may remain as a patchy understory or be eliminated by shading from overtopping trees (Risser & Fry 1988; Cornwell et al. 2012). This process has occurred in areas of montane chaparral in northern California, resulting in successional replacement of montane chaparral by mixed conifer forest (Airey Lauvaux et al. 2016). However, fire suppression may also lead to substantial fuel buildup in adjacent forest systems (Airey Lauvaux et al. 2016), increasing risk of high severity fire, which can maintain chaparral in lieu of forest vegetation (Crotteau et al. 2013; Airey Lauvaux et al. 2016; Coppoletta et al. 2016).

Roads, highways, and trails

Roads, highways, and trails cause a variety of impacts in chaparral systems. They fragment existing habitat areas (Coffin 2007) and act as a barrier to wildlife movement (Jennings 2018), particularly small mammals (Sauvajot et al. 1998), which can limit seed transport and dispersal and ultimately influence species composition and habitat recovery following disturbance (Vuln. Assessment Workshop, pers. comm., 2017). Transportation corridors also facilitate invasive plant spread, both by increasing non-native seed transport (e.g., on vehicles) and through construction activities which increase disturbance and provide colonization opportunities (Coffin 2007). Following canopy disturbance, chaparral systems are vulnerable to non-native species such as *Bromus* spp. and *Centaurea* spp., which invade and displace understory vegetation (Keeley et al. 2011; Keeley & Brennan 2012). Vehicular travel also promotes nitrogen deposition, which may further facilitate invasive species establishment following disturbance that opens the chaparral canopy (Keeley et al. 2011; Keeley & Brennan 2012), especially in serpentine habitats (Fenn et al. 2010).

Sensitivity to other critical factors

Habitat experts evaluated chaparral shrublands as having moderate-high sensitivity (moderate confidence in evaluation) to the potential decoupling of plant phenologies and pollinator migrations and/or life cycles as a result of climate change. Chaparral plants are generally insect-pollinated (Parker et al. 2016). Climate change may cause a mismatch in plant flowering timing and insect migrations or life cycles, which can reduce seed set and successful recruitment for insect-pollinated species (Memmott et al. 2007).

Adaptive Capacity

Chaparral shrublands were evaluated by regional experts as having moderate overall adaptive capacity (high confidence in evaluation).

Habitat extent, integrity, continuity, and permeability

Regional experts evaluated chaparral shrublands as having a high geographic extent (high confidence in evaluation) and moderate-high structural and functional integrity (moderate confidence). Habitat continuity and landscape permeability varies by substrate type: non-serpentine chaparral was evaluated as having moderate continuity (high confidence) and low-moderate landscape permeability (high confidence), while serpentine chaparral was evaluated as having low continuity (high confidence) but moderate landscape permeability (high confidence). Geologic features (i.e., soil types), roads/highways/trails, and invasive/problematic species were identified as the primary barriers to habitat continuity and dispersal for both substrate types across the study region (Vuln. Assessment Workshop, pers. comm., 2017).[11] Geologic features (i.e., soil types) and invasive species impact where chaparral can establish, and roads, highways and trails increase fragmentation and limit seed dispersal by wildlife (Vuln. Assessment Workshop, pers. comm., 2017). Additionally, agricultural land-use conversion (i.e., for vineyards, cannabis (*Cannabis sativa* or *C. indica*) cultivation, and/or feral livestock grazing) acts as a dispersal barrier for non-serpentine chaparral (Vuln. Assessment Workshop, pers. comm., 2017).

Chaparral shrublands are the most abundant vegetation type in California, comprising 9% of wildland vegetation in the state (Parker et al. 2016). Chaparral shrublands also occur beyond California, ranging from Baja, California north to south-central Oregon, and in disjunct patches as far north as Washington and as far east as Arizona (Keeley & Davis 2007). In northern California, various combinations of chaparral species occur from near sea level to near treeline in the Sierra Nevada, the Cascades, the Klamath Mountains, and the Coast Range (Keeley & Davis 2007; Parker et al. 2016; Vuln. Assessment Reviewers, pers. comm., 2018). Chaparral is more continuous in inland locations (England 1988; Keeley & Davis 2007), which generally demonstrate higher landscape heterogeneity and greater climate variability (Vuln. Assessment Reviewers, pers. comm., 2018). Serpentine chaparral distribution is controlled by substrate type, resulting in it being more discontinuous (i.e., patchy) and less widespread than non-serpentine chaparral, which will likely limit migration opportunities in the face of climate change (Damschen et al. 2012). In general, chaparral shrublands are intermixed with other habitat types such as oak woodlands, coniferous forest, and grasslands; chaparral plant species often co-occur in suitable habitats with taller woody vegetation (Keeley & Davis 2007). Trees in oak woodlands and coniferous forests typically dominate more mesic landscape areas unless limited by fire or other ecological constraints, while grasslands dominate in more fire- and disturbance-prone areas (Keeley & Davis 2007).

Similar to other northern California habitats, chaparral integrity in northern California has been affected by fire suppression (Potts & Stephens 2009; Potts et al. 2010; Steel et al. 2015; Norgaard et al. 2016; Wilkin et al. 2017) and regional land use changes (Vuln. Assessment Workshop, pers. comm., 2017). Future human population growth and associated urban and low density exurban development may contribute to regional chaparral shrubland loss,

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

fragmentation, and degradation (e.g., through increased wildfire ignitions and non-native annual invasion); links between development and chaparral shrubland impacts have been well documented in southern California (Syphard et al. 2018).

Habitat diversity

Regional experts evaluated non-serpentine chaparral shrublands as having high physical and topographical diversity (high confidence in evaluation), moderate-high component species diversity (high confidence), and moderate-high functional group diversity (high confidence). Regional experts evaluated serpentine chaparral shrublands as having low physical and topographical diversity (high confidence in evaluation), low component species diversity (no confidence score provided), and low-moderate functional diversity (moderate confidence).

Chaparral shrublands exhibit high physical and topographic diversity throughout northern California, with species composition varying within and between sites based on topography, aspect, elevation, distance from the coast, climate, disturbance history, soil type, and soil miosture (England 1988; Risser & Fry 1988; Keeley & Davis 2007; Halsey & Keeley 2016; Keeley 2018). For example, elevation is associated with differences in species composition between montane and mixed chaparral communities (England 1988; Risser & Fry 1988), and serpentine soils support different chaparral species and different canopy structure than non-serpentine soils (England 1988; Risser & Fry 1988; Keeley & Davis 2007).

Chaparral shrublands support several functional groups, although functional group diversity is lower in California chaparral communities than interior U.S. chaparral communities (e.g., Arizona; Keeley et al. 2012). Shrubs and sub-shrubs are the dominant functional group, but chaparral habitats also support small trees and understory annual and perennial herbaceous species (Keeley & Davis 2007; Keeley 2018). Understory vegetation cover declines in more mature stands as the shrub canopy closes , and fluctuates annually with changes in precipitation (Halsey & Keeley 2016; Keeley 2018). Plants within these groups also display diverse functional traits (Hidalgo-Triana et al. 2018) and life histories (e.g., obligate seeding or resprouting, facultative seeding), further contributing to community diversity (Keeley 2018).

Across its geographic range, chaparral has high species richness and diversity as a result of variable topography, climates, disturbance histories, and soil types (England 1988; Risser & Fry 1988; Keeley & Davis 2007; USDA Forest Service 2009a, 2009b; Halsey & Keeley 2016; Keeley 2018; Hidalgo-Triana et al. 2018). Additionally, many dominant shrub genera (e.g., *Ceanothus*, *Arctostaphylos*) exhibit high speciation, contributing to high regional species diversity (Halsey & Keeley 2016; Keeley 2018). Vegetative species richness is highest in early successional, post-fire chaparral stands, and declines over time as the shrub canopy closes (Keeley & Davis 2007; Halsey & Keeley 2016; Keeley 2018). Chaparral shrublands support a variety of rare and endemic species, and also support a variety of wildlife, including deer, rodents, birds, reptiles, amphibians, and invertebrates (Risser & Fry 1988; Keeley & Davis 2007; Estes 2013; Halsey & Keeley 2016; Jennings 2018). For example, montane chaparral is known to support many shrubdependent bird species that are otherwise rare in adjacent coniferous habitats (Humple & Burnett 2010).

Below, major plant species are listed and other notable diversity concepts are discussed for montane chaparral and mixed chaparral, as well as serpentine areas found within these major chaparral types. A more detailed description of various chaparral alliances within these types can be found in USDA Forest Service (2009a, 2009b).

- *Montane chaparral*: Montane chaparral is typically characterized by one or more of the following species: whitethorn ceanothus (*Ceanothus cordulatus*), snowbrush ceanothus (*C. velutinus*), greenleaf manzanita (*Arctostaphylos patula*), whiteleaf manzanita (*A. viscida*), pinemat manzanita (*A. nevadensis*), hoary manzanita (*A. canescens*), bitter cherry (*Prunus emarginata*), huckleberry oak (*Quercus vaccinifolia*), bush chinquapin (*Chrysolepis sempervirens*), serviceberry, Fremont silktassel (*Garrya fremontii*), and mountain mahogany (*Cercocarpus ledifolius*; Risser & Fry 1988; USDA Forest Service 2009a, 2009b).
- *Mixed chaparral*: Depending on location, mixed chaparral stands may feature shrub oaks (e.g., scrub oak [*Quercus berberidifolia*], huckleberry oak, Sadler oak [*Q. sadleriana*], canyon live oak [*Q. chrysolepis* var. *nana*] and interior live oak [*Q. wislizeni* var. *frutescens*]) and a variety of different *Ceanothus* and manzanita species, as well as shrubs such as chamise, toyon, California bay (*Umbellularia californica*), coyote brush (*Baccharis pilularis*), birchleaf mountain mahogany (*Cercocarpus betuloides*), silktassel (*Garrya* spp.), California buckeye (*Aesculus californica*), and poison oak (*Toxicodendron diversilobum*; England 1988; USDA Forest Service 2009a, 2009b).
- *Serpentine chaparral*: Serpentine soils generally support endemic chaparral species (Keeley & Davis 2007), such as Jepson ceanothus (*C. jepsonii*), coyote ceanothus (*C. ferrisiae*), dwarf ceanothus (*C. pumilus*), whiteleaf manzanita, and leather oak (*Quercus durata*; England 1988; USDA Forest Service 2009a, 2009b; Hidalgo-Triana et al. 2018). Serpentine chaparral also supports serotinous conifer species such as knobcone pine (*Pinus attenuata*), Coulter pine (*P. coulteri*), foothill pine (*P. sabiniana*), incense cedar (*Calocedrus decurrens*) and cypress (*Hesperocyperis* spp.; England 1988; Safford & Harrison 2004; Hidalgo-Triana et al. 2018). The patchy nature of serpentine soils contributes to high regional diversity (i.e., gamma diversity) as well as high diversity and species composition differentiation between sites (i.e., beta diversity) by increasing edge effects and driving local extinction and recolonization dynamics on individual patches (Harrison 1997).

Some chaparral species may be particularly vulnerable to climate change. For example, both obligate-seeding and obligate-resprouting species are vulnerable to increasingly frequent fires (Risser & Fry 1988; Haidinger & Keeley 1993; Keeley & Davis 2007; Cornwell et al. 2012). Serpentine pollinators and endemic and rare plants may also be particularly sensitive to climate-related extirpation due to their limited distribution (Vuln. Assessment Workshop, pers. comm., 2017).

Resistance and recovery

Regional experts evaluated non-serpentine chaparral shrublands as having moderate-high resistance to climate stressors and natural disturbance regimes (high confidence in evaluation), while recovery potential was evaluated as moderate (moderate confidence). For serpentine chaparral shrublands, resistance was evaluated as high (moderate confidence) and recovery potential was evaluated as moderate-high (moderate confidence).

Chaparral shrublands are resilient to many disturbances, including drought, wildfire (with adequate return intervals), and non-native species invasion (Keeley & Davis 2007; Small et al. 2018). For example, component species feature many adaptations that allow them to withstand seasonal drought (e.g., waxed leaf coatings, thick cell layers, recessed stomata to reduce evaporative moisture loss; Keeley & Davis 2007). Chaparral plant associations generally recover quickly post-fire (Keeley & Davis 2007), and persistent soil seedbanks maintain reestablishment opportunities after fire in areas where chaparral has been successionally replaced by conifers (Knapp et al. 2012). Additionally, chaparral shrublands with intact canopies are highly resistant to invasion by annual grasses, with serpentine sites even more so (Keeley & Brennan 2012). However, chaparral is less resilient to increasing disturbance frequency. For example, increasing fire frequency may increase the likelihood of type conversion to non-native annual grassland or degraded shrubland (Haidinger & Keeley 1993; Keeley & Davis 2007; Keeley et al. 2011; Keeley & Brennan 2012; Halsey & Keeley 2016) or increase barren area (Vuln. Assessment Workshop, pers. comm., 2017).

Chaparral recovery following disturbance depends on a variety of factors, including the severity of the stressor (Vuln. Assessment Workshop, pers. comm., 2017), substrate type, topographic location, and moisture availability (England 1988). Based on a review of 25 studies in California, Bohlman et al. (2018) found that live aboveground biomass of mixed chaparral increased rapidly up to 30 years (to over 4,500 g/m^2), before declining. In general, community development and recovery following disturbance tends to be much slower in drier areas (England 1988) and areas with shallow soils (Risser & Fry 1988; Safford & Harrison 2004). In addition, most chaparral species have a limited dispersal range and are slow growing, which can limit recovery potential (Keeley & Davis 2007) and migration in response to climate change, especially if habitat fragmentation increases in the region as a result of development and human landscape alterations (Beltrán et al. 2014). In general, obligate seeding and obligate resprouting species appear to be less resilient to more severe climatic conditions and enhanced disturbance than facultative seeders (Keeley 1991; Cornwell et al. 2012).

Serpentine chaparral may have higher resistance to, but slower recovery from, climatic changes and enhanced disturbance. Harsh growing conditions associated with these low-productivity soils limit non-native invasion success and contribute to specialized flora with stress-tolerant functional traits (e.g., low specific leaf area), enhancing community resilience to increasing aridity and other stressors (Anacker et al. 2011; Keeley & Brennan 2012; Damschen et al. 2012; Harrison et al. 2015). However, community development and recovery following disturbance tend to be much slower in serpentine areas (Safford & Harrison 2004). Additionally, habitat areas with suitable substrate are small and spatially isolated, which can limit serpentine species' ability to track climate changes (Damschen et al. 2012).

Management potential

Public and societal value

Regional experts evaluated chaparral shrublands as having low public and societal value (high confidence in evaluation). Chaparral areas are valued for hunting, birding, grazing, and off-highway vehicle recreation use (Vuln. Assessment Workshop, pers. comm., 2017). They also are highly valued by regional Native American tribes (see discussion of ecosystem services below). Chaparral shrublands are also valued in the scientific community for ecosystem research opportunities (Vuln. Assessment Workshop, pers. comm., 2017). There is some societal support for habitat management from hunting groups and the California Mule Deer Foundation, which supports protecting habitat for trophy mule deer (Vuln. Assessment Workshop, pers. comm., 2017).

Despite its ecological value and the many ecosystem services it provides, chaparral is often valued less than other vegetation types (Parker et al. 2016). Historically, this was because chaparral was seen as less valuable for grazing and timber (Parker et al. 2016), and because it was thought to be using water that could otherwise be used for agriculture (Safford et al. 2018b). Chaparral is also often viewed as a fire risk due to its propensity for high severity fire (Parker et al. 2016). For example, there is regional societal support for eradicating or suppressing chaparral to reduce fire risk to human structures, and this support would likely increase in the wake of extreme fire events (Vuln. Assessment Workshop, pers. comm., 2017).

Despite these views, recent education and scientist-resource manager knowledge exchange efforts are attempting address common misperceptions involving chaparral ecosystems and increase appreciation and value of the critical functions these ecosystems provide (Underwood et al. 2018b, 2018a). For example, there is emerging literature about how fuel treatments (e.g., prescribed fire, fuel mastication) can be used to reduce fire risk while maintaining the ecological integrity of chaparral (see discussion below; Le Fer & Parker 2005; Potts & Stephens 2009; Potts et al. 2010; Wilkin et al. 2017).

Management capacity and ability to alleviate impacts[12]

Regional experts evaluated the potential for reducing climate impacts on chaparral shrublands as low-moderate (high confidence in evaluation). Regional experts identified use conflicts and/or competing interests for chaparral shrublands as residential and commercial development (particularly in backcountry areas adjacent to public land), viticulture, and off-highway vehicle use (Vuln. Assessment Workshop, pers. comm., 2017).

Many climate-related management opportunities for chaparral identified in the scientific literature revolve around managing fire regimes. In general, interactions between housing

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

development, fire ignitions, non-native grasses, and roads make fire management complex (Parker et al. 2016). Land use planning that limits exurban sprawl and/or development adjacent to chaparral could reduce both anthropogenic ignitions and the need for fuel reduction (Syphard et al. 2018). Prescribed fire can be used to help restore natural fire regimes and maintain chaparral benefits for wildlife (Potts et al. 2010), but requires careful management to avoid detrimental impacts on chaparral recruitment, recovery, and vulnerability to non-native invasion, particularly as climate change progresses (Le Fer & Parker 2005; Perchemlides et al. 2008; Potts & Stephens 2009; Potts et al. 2010; Wilkin et al. 2017). For example, spring prescribed burns, while attractive to managers due to lower risk of fire spread, may be detrimental to chaparral seedling germination and recruitment (Le Fer & Parker 2005; Potts et al. 2010; Wilkin et al. 2017). Native American tribes historically used fire to increase abundance of important chaparral species and manage chaparral encroachment on other ecosystem types, presenting potential opportunities for integrating traditional ecological knowledge into agency natural resources management and decision-making (Anderson & Rowney 1999; Anderson 2005; Anderson & Rosenthal 2015; Norgaard et al. 2016; Anderson & Keeley 2018). Prescribed and wildland fire will also have to be managed at a landscape scale to avoid unintended type conversions of forested vegetation (Airey Lauvaux et al. 2016) while also minimizing losses of intact chaparral to fuelbreak creation efforts (Parker et al. 2016; Safford et al. 2018a).

There may also be some climate-informed chaparral restoration opportunities. For example, there are seed collections in conservation storage available for restoration use, depending on the scale of treatment (Vuln. Assessment Workshop, pers. comm., 2017), although restoration efforts will need to mitigate risk of disease spread via contaminated soil sometimes found in nursery-grown restoration stock (Parker et al. 2016; Swiecki et al. 2017; Underwood et al. 2018a). Additionally, managers may want to consider preserving hydraulic trait diversity among chaparral species and populations to maintain chaparral shrubland resilience to drought and moisture stress (Jacobsen et al. 2014).

Ecosystem services

Chaparral shrublands provide a variety of ecosystem services, including:

- Provisioning of food, fiber, fuel, genetic resources, natural medicines, ornamental resources, and fresh water;
- Regulation of air quality, climate/microenvironments, flood and erosion control, water purification, pollination, and carbon sequestration;
- Support of primary production, soil formation and retention, nutrient cycling, and water cycling; and
- Cultural and tribal uses for spiritual and religious purposes, knowledge systems, educational values, aesthetic values, social relations, sense of place, cultural heritage, inspiration, and recreation (Vuln. Assessment Workshop, pers. comm., 2017).

Chaparral habitats are increasingly being recognized for the critical ecosystem services they provide, including flood and erosion control, water filtration, carbon sequestration, support for native pollinator communities, and recreation (Kremen et al. 2002, 2004; Parker et al. 2016, 2016; Safford et al. 2018a; Wohlgemuth & Lilley 2018). For example, chaparral cover provides sediment retention services and protects the soil from erosion and overland flow, especially after heavy rains (Wohlgemuth & Lilley 2018), and intact chaparral provides water filtration for areas downstream in the watershed (Parker et al. 2016). Healthy chaparral and other native vegetation in close proximity to cropland and orchards provides important pollination services (Kremen et al. 2002, 2004). Chaparral communities also sequester more carbon than the non-native grasslands they can become converted to with too frequent fire (e.g., Bohlman et al. 2018). Chaparral, especially old-growth chaparral, also supports biodiversity by providing important habitat for many wildlife species (Halsey & Keeley 2016).

Chaparral is also recognized for providing significant cultural ecosystem services, including recreation (e.g., hunting, backpacking, hiking) and various tribal uses by Native Americans (Anderson 2005; Parker et al. 2016, 2016). In northern California, tribes have long relied on and managed chaparral shrublands via fire to provide important material, food, and medicinal resources, including basketry materials, utensils, berries, seeds, and medicinal teas (Anderson 2005; Anderson & Rosenthal 2015).

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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),[13] 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.[14]

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

[14] 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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