Oak Savannas and Open Woodlands

Northern California Climate Change Vulnerability Assessment Synthesis

An Important Note About this Document: *This document represents an initial evaluation of vulnerability for savanna and foothill oak woodlands 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 (University of California Cooperative Extension; brief comments only), Earl Crosby (Karuk Tribe); Frank Lake (U.S. Forest Service), Kari Norgaard (University of Oregon), Chad Roberts (Tuleyome), Bruce Taylor (Pacific Birds Habitat Joint Venture), and Jim Weigand (Bureau of Land Management). Vulnerability scores were provided by Redding workshop participants. Upper Lake workshop participants provided additional comments on the climate change vulnerability of this habitat.

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Habitat Description

Oak-dominated habitats within northern California comprise a diverse set of communities and species assemblages, with habitat type and structure corresponding to the elevation, geographic range, geology/soils, and ecological needs of the dominant oak species (Pavlik et al. 1991; Jimerson & Carothers 2002; Altman & Stephens 2012). This assessment will focus on habitats dominated by blue oak (*Q. douglasii*), valley oak (*Q. lobata*), interior live oak (*Q. wislizeni* var. *wislizeni*), and Oregon white oak (*Q. garryana*) that fall into the following structural categories: oak savanna (scattered trees with <25% cover), open oak woodlands (25–50% cover), and riparian oak woodlands (located adjacent to water bodies).[1]

Savannas and woodlands dominated by blue oak, valley oak, and interior live oak are extensive within the interior Coast Range and Sacramento Valley, often with foothill pine (*Pinus sabiniana*; also called gray pine) as a co-dominant species (Davis et al. 2016b; CNPS 2019). Blue oak savannas and woodlands are the most widespread types and are generally found on warmer, drier sites, where they occur with a mixture of other hardwoods and conifers (CNPS 2019). Interior live oak is most common in the southern Klamath Mountains and Coast Ranges, and is often associated with blue oak woodlands and chaparral (CNPS 2019). The shrubby form of interior live oak (*Q. w.* var. *frutescens*) is also found within the region, generally as an element of mixed chaparral and other shrublands (Thorne et al. 2004). Finally, valley oak woodlands occur in riparian areas and floodplains with deep alluvial soils up to 775 m (2,500 ft) in elevation, primarily in the Sacramento Valley and Coast Ranges (CNPS 2019).

Oregon white oak habitats are scattered throughout the coastal forest zone, along river corridors, and within lower montane forests in the Klamath region (Gaman & Firman 2006; Stuart & Stephens 2006; CNPS 2019). This species is distributed across a wide elevational range in California (16-2,700 m [52-8,860 ft]) (Calflora 2019), but is typically found between 60 and 1,800 m (200-5,900 ft; CNPS 2018). Stands of up to several hundred hectares often occur on drier, poorly-drained, and/or frequently disturbed sites, and habitat structure can range from open savannas to closed-canopy stands (Stuart & Stephens 2006; Schriver 2015; CNPS 2019). Oregon white oak can also occur as a dominant or co-dominant species with conifers and other hardwoods in a variety of other forest types (Calflora 2019; CNPS 2019).

Oak savannas and open woodlands are highly valued by northern California tribes, and acorns are harvested by most or all tribes in the region (Anderson 2005; Mensing 2006), in part because they have relatively high fat and protein content (Pavlik et al. 1991). Blue oak and interior live oak are also manipulated through cultural burning practices to produce long, straight epicormic branches used for weaving baskets and cradleboards (Anderson 1993, 2005; Mensing 2006). Oak-dominated ecosystems support diverse wildlife and understory plant species (Grivet et al. 2008; Davis et al. 2016b), including many shrubs, grasses, and forbs that are culturally valued by area tribes and provide food (e.g., seeds, berries, mushrooms), medicine, fiber for basketry, and forage for animals that are hunted (Anderson 1993, 2005; Shebitz 2005; Hankins 2015). Contemporary oak savannas and open woodlands have existed since the last glacial period (Mensing 2005), and their structure and distribution has been enhanced and maintained by routine cultural burning (Biswell 1999; Anderson 2005; Mensing 2006; Hankins 2015), which is considered a "keystone process" that creates heterogeneity of species and habitats (Hankins 2015).

[1] Oak-dominated habitat structures are modeled after Altman and Stephens (2012), which primarily distinguish habitat structure categories based on canopy cover.

Executive Summary

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

Oak Savannas and Open Woodlands	Rank	Confidence
Sensitivity	Moderate-High	High
Future Exposure	Moderate	Moderate
Adaptive Capacity	Moderate	High
Vulnerability	**Moderate-High**	**High**

Sensitivity & Exposure Summary	
	Climate and climate-driven factors: • Climatic water deficit, soil moisture, precipitation amount and timing, drought *Disturbance regimes*: • Wildfire *Non-climate stressors*: • Fire suppression, understory fuel management, livestock grazing, recreation and trails, timber harvest, residential and commercial development, roads/highways/powerlines

Oak savannas and open woodlands are primarily sensitive to changes in climate stressors that alter water availability, including changes in precipitation amount and timing, reduced soil moisture, and increased drought. These changes can impact acorn germination and seedling/sapling growth and survival, ultimately determining oak recruitment rates and habitat distribution. Increased water stress also reduces tree vigor, enhancing oak vulnerability to disturbance-related mortality, and alters the availability and moisture level of potential wildfire fuels.

Oaks are well-adapted to low- and moderate-intensity fires, which act as a critical disturbance regime for savannas and open woodlands. However, changes in the frequency and intensity of fires may increase rates of tree mortality, preventing successful oak sapling recruitment and subsequent acorn production and potentially leading to the conversion of oak woodlands to chaparral and grasslands. Non-climate stressors have also significantly contributed to the loss and degradation of oak savannas and open woodlands across the region, and are likely to exacerbate the negative impacts of climate change. Fire suppression, in particular, has resulted in conifer encroachment and subsequent declines in habitat extent in Oregon white oak woodlands across the region. Other non-climate stressors that impact northern California savannas and open woodlands include understory fuel management, livestock grazing, recreation and trails, timber harvest, residential and commercial development, and the presence and use of roads, highways, and powerlines. These anthropogenic stressors can degrade savannas and open woodlands by contributing to the spread of invasive species, increasing the frequency of high-severity wildfire, fragmenting habitats, and altering ecosystem processes.

Adaptive Capacity Summary	
	Factors that enhance adaptive capacity: + Long lifespan and mast seeding strategy increases resilience to changing conditions + Adapted to drought and rapid recovery following wildfire (depending on intensity) + Historically high understory species diversity, which supports a broad array of wildlife + Holds significant cultural and public value *Factors that undermine adaptive capacity*: – Significant loss of oak savannas and open woodlands due to land-use conversion (e.g., to development, agricultural crops, livestock) and conifer encroachment associated with fire suppression – Recruitment limited by low pollen production and short acorn dispersal distances

Although oak savannas and open woodlands remain widely distributed in northwestern California, their historical extent has been significantly reduced by development, agriculture, fire exclusion, and other anthropogenic stressors, and many of the remaining areas have been degraded and/or fragmented. High topographic diversity within the region has contributed to the wide distribution and heterogeneous composition of oak savannas and open woodlands, which support very high biodiversity within understory vegetation and wildlife communities. Additionally, oaks are adapted to survive drought and wildfire (depending on intensity), and suitable microclimates may allow the persistence and regeneration of oaks in drier areas. However, genetic exchange and successful oak recruitment under changing climate conditions may be limited due to low pollen production and short acorn dispersal distances. Additionally, low recruitment rates observed in blue oak and valley oak are likely to limit their ability to recover from future disturbances and stressors. Oaks have particularly high value for tribal communities in the region, and the continued decline of savannas and open woodlands would result in the loss of many ecosystem services provided by these habitats. Management strategies that may increase the resilience of oak-dominated habitats to climate impacts include removal of encroaching conifers and reintroduction of frequent low-intensity fire to Oregon white oak woodlands, management of grazing impacts, and restoration plantings.

Sensitivity and Exposure

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

Climate changes are projected to alter the distribution of oak savannas and open woodlands across the state by the end of the century, primarily due to warmer temperatures, decreased moisture availability, and increases in wildfire activity (Shafer et al. 2001; Kueppers et al. 2005; Lenihan et al. 2008; Pellatt et al. 2012; McLaughlin & Zavaleta 2012; Serra-Diaz et al. 2016; Thorne et al. 2016, 2017). Several studies project that suitable habitat area for blue oak and valley oak will generally shift north, with some expansion occurring in northwestern California due to conversion of other forest types to oak-dominated habitats (Kueppers et al. 2005;

Lenihan et al. 2008; Thorne et al. 2016, 2017). For California foothill and valley forests and woodlands, a group dominated by oaks including blue, valley, and interior live oak, 17-54% of the current mapped habitat area state-wide is projected to experience a significant increase in environmental stress and/or to become climatically unsuitable by the end of the century (Thorne et al. 2016).[2] Species-specific projections for blue oak and valley oak suggest that, state-wide, climatically suitable habitat may contract to 59-81% of the currently suitable area for blue oak and 54-73% of the currently suitable area for valley oak (Kueppers et al. 2005). For valley oak, this area may be further reduced by up to 15%, due primarily to drier conditions that limit recruitment (McLaughlin & Zavaleta 2012). The primary areas of range contraction for foothill oak woodlands in northern California are likely to occur in the foothills of the Coast Range surrounding the Central Valley and, to a lesser degree, in the Siskiyou Mountains (Kueppers et al. 2005; Thorne et al. 2016). Comparatively, 30-69% of mapped foothill and valley forests and woodlands will remain climatically suitable, representing climatic refugia for these habitat types; in northern California, most of this area is around Mendocino National Forest and Berryessa Snow Mountain National Monument (Thorne et al. 2016). Finally, by the end of the century, 11-34% of the projected climatically suitable area for this habitat type will be in newly suitable areas (Thorne et al. 2016), which are primarily located in northwestern California (Kueppers et al. 2005; Thorne et al. 2016).

Slow-growing oaks with limited dispersal distances may have difficulty keeping pace with modeled shifts in climate (Kueppers et al. 2005; Sork et al. 2010). Thus, changes in habitat distribution may primarily involve spatial rearrangement of recruitment patterns within areas where oaks are currently distributed (McLaughlin et al. 2014). For instance, a study of valley oak in southern and central California found that sites with sapling recruitment were concentrated near areas where suitable habitat is projected to expand, suggesting that some distributional shifts are already occurring (McLaughlin & Zavaleta 2012). Patterns of regeneration also suggest that interior live oak regeneration is becoming more concentrated at higher elevations (i.e., in cooler, wetter conditions; Serra-

Potential Changes in Habitat Distribution by 2100

• *Foothill oak woodlands (dominated by blue, valley, and interior live oak):* 17-54% of the state-wide current vegetation distribution is projected to experience an increase in climatic stress, while 30-69% will remain within climatically suitable areas; 11-34% of the projected climatically suitable habitat will be in newly suitable areas
  ○ *Blue oak:* Range contraction to 59-81% of climatically suitable habitat area state-wide
  ○ *Valley oak:* Range contraction to 54-73% of climatically suitable habitat area state-wide
• Generally, oak savannas and open woodlands are expected to shift northwards, with potential areas of expansion in northwestern California
• Possible refugia include north-facing slopes and areas near sources of surface water or easily accessible groundwater

Source(s): Kueppers et al. 2005; McLaughlin et al. 2014, 2017; Thorne et al. 2016

[2] 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).

Diaz et al. 2016). In habitat projected to become less suitable, valley and blue oak seedlings and saplings have begun to exhibit clustering within hydrologic microrefugia, which include shaded areas, north-facing slopes, and areas near sources of surface water or easily accessible groundwater (McLaughlin & Zavaleta 2012; McLaughlin et al. 2014). Hydrologic refugia such as these are often decoupled from climate-driven changes in hydrology, buffering them from rapid change and allowing them to sustain oaks even where the climate may become less suitable (McLaughlin et al. 2017).

Sensitivity and future exposure to climate and climate-driven factors

Regional experts evaluated oak savannas and open woodlands as having high sensitivity to climate and climate-driven factors (high confidence in evaluation), with an overall moderate-high future exposure to these factors within the study region (low confidence). Key climatic factors that affect oak savannas and open woodlands include climatic water deficit, soil moisture, precipitation amount and timing, and drought.[3]

Climatic water deficit, soil moisture, and precipitation amount and timing

Moisture stress primarily impacts tree growth and seedling establishment during the summer months, strongly influencing oak savanna and open woodland species composition and habitat distribution (Jimerson & Carothers 2002; Kueppers et al. 2005; McLaughlin & Zavaleta 2012; Davis et al. 2016b, 2016a; Hahm et al. 2018). One particularly useful way to measure moisture stress for oak-dominated habitats is climatic water deficit (CWD), which is strongly correlated with oak species distribution and seedling recruitment (Davis et al. 2016b, 2016a).[4] In California, where the winter rains provide the majority of moisture between December and March, the balance between water supply and water demand for plants shifts over the course of the year, with CWD increasing as soil moisture from winter rains is depleted by late spring and evapotranspiration increases in warmer months (Stephenson 1998; Davis et al. 2016b). For deciduous oaks, this means that moisture is highest before trees have fully leafed out (Davis et al. 2016b; Hahm et al. 2018). Due to increased evaporative demand as air temperatures rise, even areas where precipitation may increase are expected to see a rise in CWD under future climate conditions (McLaughlin & Zavaleta 2012, 2013; Thorne et al. 2015; Micheli et al. 2018), likely impacting acorn germination, seedling recruitment, and tree growth (Tyler et al. 2006; McLaughlin et al. 2014; Davis et al. 2016a).

All oak savannas and open woodlands experience high year-to-year variability in rainfall, which contributes to variable acorn production (Koenig et al. 1994) and episodic seedling recruitment events (Gilligan & Muir 2011; Davis et al. 2011). Weather explains a significant portion of the interannual variability in acorn production, particularly annual rainfall during the year(s) that acorns are maturing (Koenig et al. 1996). Germination is dependent on the timing and amount

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

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

of spring rainfall, and acorns left lying on the soil surface are prone to desiccation (Griffin 1971; Tietje et al. 1991; Tyler et al. 2006). The highest rates of seedling recruitment occur when mast years are followed by high winter and spring rainfall (Griffin 1971).

Severe soil moisture deficits during the growing season cause decreased growth rates and mortality in oaks (Mahall et al. 2009; McLaughlin & Zavaleta 2012), especially where competition with annual plants (e.g., invasive grasses) further reduces available moisture (Gordon et al. 1989; Gordon & Rice 1993; Tyler et al. 2006). However, several oak species typically found on warm, dry sites have adaptations that allow them to minimize water loss and/or maximize water use efficiency (Osuna et al. 2015; Merz et al. 2017; Hahm et al. 2018). For instance, blue oak has deep root systems that can access groundwater during periods of very low soil moisture (Miller et al. 2010; Gou & Miller 2014; Osuna et al. 2015). One study found that groundwater accounted for 80% of blue oak evapotranspiration from June-August, allowing them to survive the summer dry period (Miller et al. 2010). Oregon white oak is also extremely tolerant of water stress, and is able to sustain hydraulic function during periods when water availability is very low and evaporative demands are high (Hahm et al. 2018). Since the 1930s, increases in CWD have been associated with increases in the dominance of oaks across California (McIntyre et al. 2015), suggesting that the high tolerance of water stress in oaks compared to pines and other conifers may give them a competitive advantage (McIntyre et al. 2015; Hahm et al. 2018).

Oak seedling survival and growth are higher on moister sites or where seedling irrigation occurs due to patterns of precipitation, topography, and access to surface water sources and groundwater (Kertis et al. 1993; Bakker et al. 2012; McLaughlin et al. 2014; Davis et al. 2016a, 2016b). In general, oak seedlings and saplings are more sensitive to water stress than adults (Matzner et al. 2003; Mahall et al. 2009; McLaughlin & Zavaleta 2012), in part because their root systems are not able to reach the groundwater sources that often support mature oaks (Mahall et al. 2009). Studies of valley oak found that saplings had a narrower range of suitable climate conditions compared to adult trees, suggesting that model projections may overestimate future suitable habitat distribution based on climate thresholds that allow adult trees to persist but do not support tree recruitment (McLaughlin & Zavaleta 2012). In areas where range contractions are projected to occur, researchers have observed saplings clustered around hydrological refugia (e.g., water bodies or easily accessible groundwater) that mediated the impacts of a warmer, drier climate; the same clustering around hydrological refugia did not occur in areas where oak persistence or expansion is projected (McLaughlin & Zavaleta 2012).

Although CWD is projected to increase even in areas that receive more rain (Thorne et al. 2015), increases in precipitation could benefit oak savannas and open woodlands by enhancing productivity (winter rainfall) and/or seedling survival (summer rainfall; Zavaleta et al. 2007). As a result, increases in precipitation could potentially moderate some of the impacts of increasing temperatures on areas of climatically suitable habitat (Kueppers et al. 2005; Pellatt et al. 2012; Thorne et al. 2016), allowing greater expansion in some areas (Kueppers et al. 2005). However, greater soil moisture availability may also accelerate conifer encroachment into Oregon white oak woodlands (Lenihan et al. 2008; Hahm et al. 2018).

Regional Climatic Water Deficit (CWD), Soil Moisture, & Precipitation Trends[5]	
Historical & current trends: • 0.4 cm (0.2 in) increase in average annual CWD and 2.6–7.2 cm (1.0–2.8 in) increase in mean annual precipitation between 1900 and 2009 for the Northwestern California and Great Valley ecoregions (Rapacciuolo et al. 2014) • No trends available for soil moisture	*Projected future trends:* • 23% decrease to 38% increase in mean annual precipitation by 2100 (compared to 1951–1980) for the Northern Coast Range, Northern Interior Coast Range, Klamath Mountain, and Great Valley ecoregions (Flint et al. 2013; Flint & Flint 2014)[6] • Increases in average annual CWD by 2100 (compared to 1951–1980; Flint et al. 2013; Flint & Flint 2014): ○ 7–24% increase in the Northern Coast Range ○ 5–16% increase in the Northern Interior Coast Range ○ 10–32% increase in the Klamath Mountains ○ 4–19% increase in the Great Valley • 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) • 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)*	
• Changes in oak savanna and woodland species composition and habitat distribution based on patterns of water availability ○ Shifts in species distribution towards moister microsites, with overall declines in areas that are climatically suitable for seedling establishment ○ Possible increased dominance and/or expansion into areas currently occupied by conifers	

[5] 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.

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

Regional Climatic Water Deficit (CWD), Soil Moisture, & Precipitation Trends[5]
• Decreased acorn germination, seedling survival, and tree growth on dry sites, although oaks may have a competitive advantage compared to conifers that are less tolerant of water stress

Drought

Oaks are adapted to periods of summer drought from April to October, and many species have strategies that allow them to survive longer periods of drought (Fuchs 2001; Matzner et al. 2003; Miller et al. 2010; McLaughlin & Zavaleta 2012; Hahm et al. 2018). For instance, adult blue and valley oaks are able to withstand drought by accessing groundwater with their deep root systems (Miller et al. 2010; McLaughlin & Zavaleta 2012), although, of the two species, valley oak is more vulnerable to drought (McLaughlin & Zavaleta 2012). Blue oaks may also drop their leaves during periods of particularly severe drought, allowing them to survive by limiting water loss (Matzner et al. 2003). Oregon white oaks are one of the most drought-tolerant species in the region, giving them a competitive advantage over species more prone to drought stress, including Douglas-fir (*Pseudotsuga menziesii*) and most other conifers (Hahm et al. 2018).

Periods of severe drought may still cause oak mortality, especially in sensitive seedlings and saplings (Matzner et al. 2003; Tyler et al. 2006; Mahall et al. 2009; McLaughlin et al. 2014; Young et al. 2017). Mortality is most likely to occur on drier sites and during long droughts, particularly where higher tree density increases competition for soil moisture resources (Young et al. 2017). Drought-stressed trees may also be more vulnerable to mortality from large-scale disturbances such as wildfire, outbreaks of insect pests, and disease, especially as hotter droughts become more common (Millar & Stephenson 2015). In some cases, secondary stressors such as fire, insects, or disease may be the primary driver of shifts in habitat composition and structure (Millar & Stephenson 2015).

Severe drought conditions are likely to increasingly constrict valley and blue oak seedling recruitment to hydrologic refugia that allow access to surface water or groundwater sources (McLaughlin & Zavaleta 2012; McLaughlin et al. 2014). This restriction may limit the future distribution of oak-domianted habitats to areas that fall within the range of conditions suitable for seedling establishment and survival, a climatic envelope that is considerably narrower than the conditions under which adult trees can persist (McLaughlin & Zavaleta 2012).

Regional Drought Trends	
Historical & current trends: • Drought years have occurred twice as often over the last two decades compared to the previous century (Diffenbaugh et al. 2015) • 2012–2014 drought set records for lowest precipitation, highest temperatures, and most extreme drought indicators on record (Griffin & Anchukaitis 2014; Diffenbaugh et al. 2015)	*Projected future trends:* • Drought years are twice as likely to occur over the next several decades due to increased co-occurrence of dry years with very warm years (Cook et al. 2015) • 80% chance of multi-decadal drought by 2100 under a high-emissions scenario (Cook et al. 2015)

Regional Drought Trends	
	• Severe droughts that now occur once every 20 years will occur once every 10 years by 2100 and once-in-a-century drought will occur once every 20 years (Pierce et al. 2018)
Summary of Potential Impacts on Habitat *(see text for citations)*	
• Increased mortality, especially in seedlings and saplings • Greater vulnerability to additional stressors (e.g., wildfire, disease, insects) • Constriction of valley oak and blue oak recruitment to hydrologic refugia, limiting future habitat distribution	

Sensitivity and future exposure to changes in natural disturbance regimes

Regional experts evaluated oak savannas and open woodlands as having moderate-high sensitivity to changes in natural disturbance regimes (high confidence in evaluation), with an overall moderate future exposure to these stressors within the study region (moderate confidence). The key natural disturbance regime that affects oak savannas and open woodlands include wildfire.

Wildfire

Wildfire is a critical disturbance regime in oak savannas and open woodlands, where it affects succession by influencing species composition and habitat structure (Martin & Sapsis 1992; Agee 1996; McCreary 2004; Mensing 2005, 2006; Skinner et al. 2006; Stuart & Stephens 2006). Oak-dominated habitats are adapted to the frequent occurrence of relatively low-intensity fire from human and natural ignitions (Biswell 1999; Mensing 2006; Perry et al. 2011), and indigenous cultural burning practices have supported the persistence of oak savannas and open woodlands on the landscape (Anderson 2005; Mensing 2005, 2006; Hankins 2015). At lower elevations where lightning ignitions are rare, human ignitions usually dominated fire regimes in California wildlands (van Wagtendonk & Cayan 2008; Keeley & Syphard 2017). Historical fire return intervals (due to both naturally occurring and human-ignited fires) were typically 5-45 years in lower-elevation woodlands comprised of species such as blue oak, valley oak, and Oregon white oak (Van de Water & Safford 2011). However, some stands of blue and interior live oak managed for the production of long, straight shoots were burned annually or semi-annually (Mensing 2006). Following the removal of tribes and introduction of widespread fire suppression in the early 1900s, fire return intervals in oak savannas and open woodlands lengthened considerably (Skinner et al. 2009; Safford & Van de Water 2014; Steel et al. 2015; Taylor et al. 2016), leading to shifts in species composition, habitat structure, and, ultimately, the loss of many oak-dominated habitats across the region (Jimerson & Carothers 2002; Skinner & Taylor 2006; Stuart & Stephens 2006; Anderson 2007).[7]

[7] Refer to the section on non-climate stressors for a more complete discussion of the impacts of fire suppression on savanna oak woodlands.

Periodic fire limits encroaching shrub and conifer species and opens up the canopy, allowing shade-intolerant species such as oaks to persist in the canopy without being overtopped and enhancing oak regeneration and growth (Agee 1996; Regan & Agee 2004; Holmes et al. 2008; Engber et al. 2011; Perry et al. 2011). Although fire sensitivity varies by species, most mature oaks have thick bark and are relatively resistant to low- and moderate-intensity fires that kill the seedlings of encroaching conifers (Holmes et al. 2008; Engber et al. 2011). Oaks are also well-adapted to post-fire recovery and most species sprout prolifically from their root collars following topkill during more severe fires (Plumb 1980; Swiecki & Bernhardt 1998; Horney et al. 2002; McCreary 2004; Holmes et al. 2008; Fry 2008; Arévalo et al. 2009). Oaks that regenerate by resprouting are able to produce acorns more rapidly than trees starting out as seedlings, likely because they already have an established root system (Swiecki & Bernhardt 1998; Holmes et al. 2008).

Low-intensity fire, such as occurs during prescribed or cultural burning activities, also has numerous benefits for oak savannas and open woodlands beyond promoting regeneration (Martin & Sapsis 1992; Biswell 1999; Anderson 2005; Arno & Allison-Bunnell 2013; Hankins 2015). For instance, low-intensity fire reduces insect pests that affect tree health and acorn quality while protecting large acorn-bearing trees (Anderson 2005, 2007). Periodic low-intensity fire can increase understory plant and fungi diversity (Martin & Sapsis 1992; Hankins 2015), prevent the accumulation of high fuel levels that could lead to more frequent high-severity fires (Underwood et al. 2003), and break down organic matter, releasing nutrients to the soils where they become available for plant use (Anderson 1993, 2005; Neary et al. 1999). Many wildlife species found within oak savannas and open woodlands also benefit from fire; for instance, the black-tailed deer (*Odocoileus hemionus columbianus*) depends on a mosaic of burned and unburned habitat (Innes 2013).

The impacts of higher-intensity fire are complex, and the severity of these fires varies across space and time depending on factors such as fire frequency and preexisting condition (Neary et al. 1999; Perry et al. 2011). Tree injuries and/or complete mortality during wildfires are most likely when fires burn at high intensity, and seedlings/saplings and smaller trees are significantly more likely to be injured or killed (Swiecki & Bernhardt 1998; Holmes et al. 2008). Oaks are more vulnerable to high-severity fire where fire suppression has allowed shrub and/or conifer encroachment to occur, contributing to denser woodlands with large amounts of available fuel (Holmes et al. 2008). The establishment and spread of annual grasses (e.g., *Bromus* spp.) also increases available fine fuels and enhances fuel continuity, contributing to more severe wildfires that spread rapidly across larger areas (Jimerson & Carothers 2002).

Changes in wildfire regimes that are outside the historical range of variation may increase tree injury and mortality (Jimerson & Carothers 2002; Holmes et al. 2008). In particular, more frequent high-severity fire may negatively impact the persistence of oak-dominated habitats where seedlings and sprouted trees are unable to mature and produce acorns before the next fire (Holmes et al. 2008). Repeated high-severity fire can alter woodland structure by transitioning mature oak woodlands dominated by large-diameter trees to shrubby, multi-stemmed growth forms (Barton & Poulos 2018), and can potentially convert oak woodlands to chaparral or grasslands over time (George & Alonso 2008) . High-severity fire also affects belowground processes and structures directly (i.e., through high soil temperatures) and/or indirectly (i.e., through changes in aboveground vegetation), potentially impacting fungal and bacterial community composition, soil properties (e.g., chemistry and physical structure), and other factors (Neary et al. 1999; Graham et al. 2016; Smith et al. 2017). These changes may alter nutrient and water cycling, mineralization rates, invertebrate soil fauna, and plant establishment and composition following fire (Neary et al. 1999; Cowan et al. 2016; Vuln. Assessment Reviewer, pers. comm., 2018).

Regional Wildfire Trends	

Historical & current trends:
• 85% of U.S. Forest Service lands in northern California are burning less frequently compared to pre-1850 fire return intervals, largely due to fire suppression (Safford & Van de Water 2014)
• Fire size and total area burned increased on U.S. Forest Service lands in northwestern California between 1910-2008, with the highest values occurring after 2000 (Miller et al. 2012)
• Changes in large fires (over 400 ha) in the inland northern California/Sierra Nevada region since the 1970s (Westerling 2016):
  ○ 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
	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.

Regional Wildfire Trends
Summary of Potential Impacts on Habitat *(see text for citations)*

- *Immediate*:
  - Reduced shrub and conifer encroachment following low- to moderate-intensity fire, allowing oak regeneration
  - Reduced insect pests that affect tree health and acorn quality
  - Higher rates of tree injury and mortality following more intense fires
- *Short-term (~2-year)*:
  - Enhanced reproduction, primarily via root crown sprouting
  - Reduced fuel accumulation following low-intensity fires, reducing the future risk of severe fire
  - Increased plant diversity following low-intensity fires, which creates a mosaic of burned and unburned habitat that also supports many wildlife species
- *Long-term*:
  - Enhanced regeneration and persistence of oak savannas and open woodlands due to loss of competing shrubs and conifers
  - Possible type conversion or transition to shrubby, multi-stemmed growth forms in areas burned by repeated high-severity fires
  - Loss of seed sources where high-severity fires are too spatially extensive and frequent to allow maturation of acorn-producing trees

Sensitivity and current exposure to non-climate stressors

Regional experts evaluated oak savannas and open woodlands as having high sensitivity to non-climate stressors (high confidence in evaluation), with an overall moderate-high current exposure to these stressors within the study region (high confidence). Key non-climate stressors that affect oak savannas and open woodlands include fire suppression, understory fuels management, livestock grazing, recreation and trails, timber harvest, residential and commercial development, roads/highways/powerlines.[8]

Fire suppression

Federal and state governments have managed forests by excluding fire in large portions of the landscape over the past century (Skinner et al. 2006, 2009; Stuart & Stephens 2006; Steel et al. 2015). As a result, Oregon white oak savannas and woodlands have become increasingly invaded by shade-tolerant, fire-sensitive conifers such as Douglas-fir, white fir (*Abies concolor*), and incense-cedar (*Calocedrus decurrens*; Jimerson & Carothers 2002; Engber et al. 2011).[9] After overtopping the oak canopy, these shade-tolerant species can cause crown dieback and mortality in oaks and other hardwoods as they compete light, water, nutrients, and space (Jimerson & Carothers 2002; Engber et al. 2011; Gilligan & Muir 2011). Changes in fuel type, structure, and availability associated with fire suppression and subsequent conifer encroachment can also contribute to shifts in wildfire regimes (Engber et al. 2011). Thus, deficits of fire on the landscape due to the cessation of cultural burning and implementation of

[8] All non-climate stressors presented were ranked as having a moderate or higher impact on this habitat type.
[9] Conifer invasion is less relevant for low-elevation foothill woodlands, which are generally too dry in the summer for conifers other than the gray pine to survive (Vuln. Assessment Reviewer, pers. comm., 2019).

fire suppression can alter preexisting fire-climate relationships (Taylor et al. 2016), increasing the vulnerability of oak-dominated ecosystems to uncharacteristically severe and/or large-scale disturbances that can undermine the integrity and persistence of the system (Jimerson & Carothers 2002; Engber et al. 2011; Norgaard et al. 2016).

On privately-owned and state-owned land, which comprises the majority of oak woodlands in northern California (Huntsinger & Fortmann 1990), the responsibility for wildfire suppression is shared between local governments and the California Department of Forestry and Fire Protection (CAL FIRE) (Vuln. Assessment Reviewer, pers. comm., 2018). In these areas, low-intensity fires are often allowed to burn unless people or property are at risk or are directly threatened (Vuln. Assessment Reviewer, pers. comm., 2018). The impact of containment actions (e.g., firebreaks cut by heavy equipment) are the primary ecological concern on privately-owned and state-owned land (Vuln. Assessment Reviewer, pers. comm., 2018).

Understory fuel management

Fuel management activities focused on the reduction of understory manzanita (*Arctostaphylos* spp.) and ceanothus (*Ceanothus* spp.) as potential fire hazards can impact the structure and composition of oak-dominated ecosystems (Perchemlides et al. 2008). One of the primary impacts of fuel treatment is the establishment and/or expansion of invasive annual grasses, which compete with native understory species and further alter fuel profiles (Perchemlides et al. 2008). Fuel treatments can also affect the suitability of oak-dominated habitats for shrub- and edge-associated birds such as the Bewick's wren (*Thryomanes bewickii*) and wrentit (*Chamaea fasciata*), especially in areas where large-scale treatments are completed (Seavy et al. 2008). Because understory fuel management is focused on reducing fire risk, decreased regeneration may occur in fire-dependent tree and shrub species, including oaks (Perchemlides et al. 2008).

Livestock grazing

More than three-quarters of oak woodlands within the state have been grazed over the past several decades (Huntsinger & Fortmann 1990), and many federal, state, and private owners of forested land would commonly remove or reduce the density of oaks through both mechanical and chemical (e.g., herbicide) means in order to favor non-woody plants that provided forage for livestock (Murphy et al. 1976; Murphy 1980). The direct impacts of livestock grazing on oak-dominated ecosystems vary based on grazing intensity, timing, and vegetation composition (Tyler et al. 2006), with the greatest adverse effects generally occurring on drier sites and in more open areas (Swiecki & Bernhardt 1998). Top browsing by both cattle and wild ungulates (e.g., deer) decreases oak seedling/sapling growth and survival to adulthood (Swiecki & Bernhardt 1998; Tyler et al. 2006; Arévalo et al. 2009; Davis et al. 2011). Livestock grazing may also cause damage to understory shrubs, limiting nurse plants that protect fallen acorns and oak seedlings from high solar radiation, desiccation, and herbivory (Tyler et al. 2006). Livestock can alter oak woodlands by compacting soils, impacting patterns of runoff and erosion (Jimerson & Carothers 2002).

Other consumers, especially small mammals (e.g., rodents), also contribute to rates of seed predation and seedling/sapling damage and mortality (MacDougall et al. 2010; Davis et al. 2011). The combined impact of livestock grazing and native ungulate and rodent herbivory may be the primary factor limiting oak regeneration in some areas (Davis et al. 2011). Exclusion of livestock/native ungulates and rodents can result in the recovery of oak recruitment within a decade (Davis et al. 2011).

Recreation and trails

Most of the impacts of recreation are concentrated near urban areas and on publicly-owned lands (Vuln. Assessment Reviewer, pers. comm., 2018). Increased human activity within these areas is associated with more wildfire ignitions, especially within the WUI and popular recreation areas (Syphard et al. 2007, 2009; Mann et al. 2016). Soil erosion may also occur along trails, especially those used by off-highway vehicles (OHVs; Ouren et al. 2007). OHVs compact soil, reduce water infiltration, increase runoff, and spread invasive plants (Ouren et al. 2007).

Timber harvest

In the second half of the 20th century, selective harvesting of conifers may have benefitted Oregon white oak by reducing conifer encroachment in areas that had experienced fire suppression since the early 1900s (Bolsinger 1988; Valachovic et al. 2015). However, oaks could also be seen as a nuisance because they occupied space where more commercially valuable conifers could grow, and they were commonly removed in commercial timberlands (Bolsinger 1988; Huntsinger & Fortmann 1990). In savannas and open woodlands, removal of many large oak trees for “range improvements” incentivized by the State of California provided fuelwood as byproduct (Bolsinger 1988; Holmes 1990; Huntsinger & Fortmann 1990).

Timber harvesting of Oregon white oak is regulated by the California Department of Forestry and Fire Protection (CAL FIRE) and can still occur on private lands (CAL FIRE 2017), where the majority of these woodland types are located (Huntsinger & Fortmann 1990; Valachovic et al. 2015). Prior to 2017, regulations disincentivized active management of encroaching conifers, requiring reforestation of areas dominated by conifers (considered “Class A” species), as well as the maintenance of the proportionality of conifers to hardwoods (considered “Class B” species) (Valachovic et al. 2015). However, a 2017 amendment allows a special prescription for Oregon white oak woodland restoration in historically oak-dominated stands (CAL FIRE 2017).

Residential and commercial development

The extent and continuity of oak savannas and open woodlands has been impacted by development, with the greatest impacts occurring near populated areas in and around the Central Valley and on the North Coast (Bolsinger 1988; Holmes 1990; Huntsinger & Fortmann 1990; Heise & Brooks 1998; Jimerson & Carothers 2002; Easterday et al. 2016). Over the past century, a million acres of oak woodlands have been developed across the state (Gaman & Firman 2006), with valley oak woodlands experiencing the greatest losses due to their distribution in valleys and along riparian corridors where the value of agricultural lands is high (Bolsinger 1988). Almost a quarter of remaining oak woodlands in California will be at risk of development by 2040, with the greatest risk associated with the Central Valley region (Gaman & Firman 2006).

Oak savanna and woodland loss due to development reduces genetic exchange and tree recruitment in fragmented populations (Sork et al. 2002, 2010; Grivet et al. 2008). Additionally, biodiversity is reduced in fragmented areas and plant and wildlife species composition are altered (Heise & Brooks 1998). Development may also cause the loss of microrefugia that support oak regeneration, especially where human water use lowers groundwater tables (McLaughlin et al. 2014, 2017). Finally, development and associated human activity may increase wildfire ignitions, exacerbating climate-driven shifts in wildfire regimes (Syphard et al. 2007; Keeley & Syphard 2016).

Roads, highways, and powerlines

Transportation and energy transmission corridors increase the spatial scale across which habitat loss and fragmentation of oak savannas and woodlands occurs (Bolsinger 1988; Holmes 1990). Road construction and the clearance of associated right-of-way corridors has resulted in reduced oak density in these areas (Vuln. Assessment Reviewer, pers. comm., 2018). Roads also contribute to the spread of invasive plants whose seeds may be carried on vehicles (Coffin 2007). Invasive grasses, in particular, thrive in roadside environments (Coffin 2007), and can contribute to the alteration of fire regimes by providing continuous fine fuels for wildfire (Brooks et al. 2004).

Wildfire ignitions can occur where oaks make contact with powerlines, and inadequate vegetation maintenance around power lines has been implicated in many large wildfires across the state (Paschal 2014). Horticultural tree trimming that attempts to reduce the contact of oak limbs on powerlines can also disfigure trees and reduce the crown area or size of mature older oaks (Vuln. Assessment Reviewer, pers. comm., 2018).

Adaptive Capacity

Oak savannas and open woodlands were evaluated by regional experts as having moderate overall adaptive capacity (high confidence in evaluation).

Habitat extent, integrity, continuity, and permeability

Regional experts evaluated oak savannas and open woodlands as having a high geographic extent (high confidence in evaluation), low-moderate structural and functional integrity (high confidence), and moderate-high continuity (high confidence).

Landscape permeability for oak savannas and open woodlands was evaluated as low-moderate (high confidence). Roads/highways/trails, land-use conversion, and timber harvest/clear-cuts were identified as the primary barriers to habitat continuity and dispersal across the study region. Understory fuel management has also impacted savanna and woodland habitat permeability, though to a lesser degree and/or at a more localized scale (Vuln. Assessment Workshop, pers. comm., 2017).[10]

Savannas and open woodlands dominated by blue oak, valley oak, and interior live oak currently occupy over 5 million acres within the state, or 60% of the current distribution, and the majority of these are dominated by blue oak (Davis et al. 2016b). While Oregon white oak savannas and woodlands have been described as particularly extensive within Humboldt County (Green & Magnuson 2011), stands are widespread throughout most of northwestern California (Griffin & Critchfield 1972; Davis et al. 2016b; Easterday et al. 2016). However, the extent of oak woodland habitats in northern California has significantly declined over the past century (Swiecki & Bernhardt 1998; Kueppers et al. 2005; Grivet et al. 2008; Sork et al. 2010; Easterday et al. 2016). Changes in habitat structure and species composition have resulted in reduced functional integrity within many remaining savanna and woodland areas (Swiecki & Bernhardt 1998; Jimerson & Carothers 2002; Engber et al. 2011; Green & Magnuson 2011). Fire suppression and associated encroachment of fire-intolerant tree species have been the primary causes of degradation and loss, particularly in Oregon white oak woodlands and savannas (Agee 1996; Engber et al. 2011; Green & Magnuson 2011). For instance, within the Bald Hills of northern Humboldt County it is likely that 30% of habitat dominated by Oregon white oak was lost through conversion to Douglas-fir forest types between 1850 and 1983 (Sugihara et al. 1987).

In lower-elevation oak savannas and woodlands, land-use conversion adjacent to urban/suburban development and agricultural areas has also contributed to declines in habitat extent (Swiecki & Bernhardt 1998; Grivet et al. 2008; Sork et al. 2010; Easterday et al. 2016). Savannas and woodlands that occur on privately-owned land may be particularly vulnerable to land use conversion that fragments remaining habitat areas (Bolsinger 1988; Huntsinger & Fortmann 1990; Huntsinger et al. 1997; Heise & Brooks 1998; Green & Magnuson 2011). For instance, some Oregon white oak woodlands are being converted to vineyards and other more intensive forms of agriculture (Heaton & Merenlender 2000), including areas in eastern Humboldt County (Vuln. Assessment Reviewer, pers. comm., 2018). Relatively short pollen and acorn dispersal distances exacerbate the effect of habitat fragmentation by limiting pollination and genetic exchange (Knapp et al. 2001; Sork et al. 2002, 2010; Grivet et al. 2008), which may result in limited ability of blue oak and valley oak to migrate in response to climate change (Kueppers et al. 2005; Sork et al. 2010; Barbour & Kueppers 2012).

State-wide, low recruitment is leading to population declines in both valley oak and blue oak (Tyler et al. 2006; Zavaleta et al. 2007). Valley oak, in particular, has very low rates of seedling and sapling recruitment (Zavaleta et al. 2007). Although more sites across the state show evidence of regeneration for blue oak compared to valley oak (79% compared to 48% in valley oak; Zavaleta et al. 2007), regeneration rates are typically still too low for adequate population replacement (Zavaleta et al. 2007).

[10] Barriers to habitat continuity and dispersal presented were all ranked as having a moderate or higher impact on this habitat type.

Habitat diversity

Regional experts evaluated oak savannas and open woodlands as having moderate-high physical and topographical diversity (high confidence in evaluation), moderate component species diversity (high confidence), and high functional diversity (high confidence).

Oregon white oak, blue oak, valley oak, and interior live oak act as keystone species within oak savannas and open woodlands in northern California (Jimerson & Carothers 2002; CNPS 2019). High topographic diversity in the region created by precipitation and soil moisture gradients, slope and elevational changes, and varying soil types facilitates successful oak reproduction and contributes to the persistence of oak-dominated habitats (McLaughlin & Zavaleta 2012, 2013; McLaughlin et al. 2014; Davis et al. 2016b). These factors contributed to historically high genetic diversity across the range of most oak species (Grivet et al. 2008).

Foothill oak woodlands are often relatively open (10-60% canopy cover) with an understory comprised of native and non-native forbs, exotic annual grasses, and scattered clumps of shrubs (Davis et al. 2016b). Blue oak savannas and woodlands are found in upland areas, valley bottoms, and terraces (Pavlik et al. 1991; Davis et al. 2016b), and can be exclusively blue oak or can include a mixture with other hardwoods and conifers, including California buckeye (*Aesculus californica*), foothill pine, valley oak, and interior live oak (CNPS 2019). Stands with a well-developed shrub layer may also include common manzanita (*Arctostaphylos manzanita*), *Ceanothus* spp., birch leaf mountain-mahogany (*Cercocarpus betuloides*), and poison oak (*Toxicodendron diversilobum*), among others (Calflora 2019; CNPS 2019).

Compared to blue oak, valley oak woodlands occur on more mesic sites in foothill and valley riparian areas in the Central Valley and North Coast Range, where they are associated with deep alluvial soils (CNPS 2019). Other associated species can include boxelder (*Acer negundo*), white alder (*Alnus rhombifolia*), Oregon ash (*Fraxinus latifolia*), northern California black walnut (*Juglans hindsii*), California sycamore (*Platanus racemosa*), Fremont cottonwood (*Populus fremontii*), Gooding's black willow (*Salix gooddingii*), blue oak, California black oak (*Q. kelloggii*), and Oregon white oak (Calflora 2019; CNPS 2019). Understory shrub species may include California pinevine (*Aristolochia californica*), California wild grape (*Vitus californica*), and poison oak, among others (Calflora 2019; CNPS 2019).

Interior live oak is frequently associated with blue oak, and is often found on steeper and/or rockier slopes (Pavlik et al. 1991; CNPS 2019). Interior live oak can be found in both tree and shrub forms; co-dominant trees can include California buckeye, Pacific madrone (*Arbutus menziesii*), foothill pine, blue oak, and black oak (CNPS 2019). Interior live oak is frequently associated with blue oak woodlands and montane chaparral habitat types (CNPS 2019).

Oregon white oak savannas and woodlands are distinct and important habitat types in northwestern California, where they frequently occur in areas where competition from other species is minimized, such as on harsh sites (e.g., dry, poorly drained, frequently disturbed) and/or on serpentine soils (Schriver 2015). Oregon white oak can be found in closed-canopy stands and as a component of forested habitats as well as in a bald hills habitat type unique to California (Griffin 1988; Stuart & Stephens 2006; Green & Magnuson 2011; Schriver 2015; CNPS 2019). The latter structure is characterized by a mosaic of oak stands, evergreen hardwood and conifer stands, and grassland (Griffin & Critchfield 1972; Sugihara et al. 1987; Griffin 1988; Green & Magnuson 2011). Oregon white oak is often associated with Jeffrey pine (*P. jeffreyi*), ponderosa pine (*P. ponderosa*), foothill pine, Douglas-fir, canyon live oak, black oak, and California bay (*Umbellularia californica*; CNPS 2018).

Several other oak species are also found in northern California, including black oak, canyon live oak (*Q. chrysolepis*), leather oak (*Q. durata*), huckleberry oak (*Q. vaccinifolia*), scrub oak (*Q. berberidifolia*), sadler oak (*Q. sadleriana*; also called deer oak/sweet acorn oak), and coast live oak (*Q. agrifolia*; primarily distributed south of the study area; Pavlik et al. 1991; Easterday et al. 2016; Calflora 2018). Along with Oregon white oak, several of these are frequently found within serpentine communities (Pavlik et al. 1991; Green & Magnuson 2011). The inner Coast Range in Napa and Lake Counties is considered a historical hotspot of oak species richness, where up to seven oak species can be found within a small area (Easterday et al. 2016). This area comprises much of Berryessa Snow Mountain National Monument, which was protected because of its high biological diversity and significant area of serpentine soils (Easterday et al. 2016). Overall, only 13% of hotspots within the state with high historical oak species richness have conservation protections, though about half of areas with moderate richness (2-5 species) are protected (Easterday et al. 2016).

All oak-dominated habitat types support a broad array of understory species, as well as wildlife, insects, fungi, and lichens (CalPIF 2002; Grivet et al. 2008; Davis et al. 2016b). Species diversity was historically high, but has declined over the past century due to fire suppression, conifer encroachment, and the establishment of invasive species, among other variables (Jimerson & Carothers 2002). Because Oregon white oaks are tolerant of warm, dry conditions (Hahm et al. 2018), frequent disturbances (Agee 1996), and are resistant to the sudden oak death pathogen (Hansen et al. 2005), white oak woodlands may become important refugium for wildlife and other elements of biodiversity under changing climate conditions (Vuln. Assessment Reviewer, pers. comm., 2017).

Resistance and recovery

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

The long lifespan of oak trees may increase their resistance to changing conditions, as even relatively rare survival of seedlings/saplings to adulthood can potentially offset adult mortality (Agee 1996; Davis et al. 2011). Oregon white oak saplings can exist within the understory for many decades, maturing into acorn-producing trees following disturbances that open up the canopy (Swiecki & Bernhardt 1998; Devine et al. 2007b; Gilligan & Muir 2011). Oaks do not form a seedbank because acorns are unable to germinate after more than a year (Swiecki & Bernhardt 1998; Thorne et al. 2016); however, mast seeding strategies (i.e., the production of large volumes of seed every few years) increase the chances of successful oak recruitment (Koenig et al. 1994). Oak species go through rotational mast production (i.e., oak species produce mast crops in different years), suggesting that this process is largely independent of climate controls (Koenig et al. 1994).

Mature oaks are generally more resilient than seedlings and saplings, and can often survive disturbances such as drought and wildfire (Holmes et al. 2008; McLaughlin & Zavaleta 2012; McLaughlin et al. 2014). However, changing climate conditions and ongoing management practices (e.g., fire suppression, grazing) may increase competition from conifers and shrubs and cause further mortality in existing oak populations, limiting forest regeneration and resilience to fire and other disturbances (Sork et al. 2010; Hayes & Donnelly 2014). Oregon white oak woodlands impacted by fire suppression and subsequent encroachment are particularly vulnerable to damage from wildfire and other disturbances (Engber et al. 2011).

Due to low recruitment, slow growth, and limited dispersal in the absence of animal vectors (e.g., jays), blue oak and valley oak may be unable to shift their range rapidly enough to successfully track projected shifts in climate (Kueppers et al. 2005; Loarie et al. 2008; Sork et al. 2010; Barbour & Kueppers 2012; McLaughlin & Zavaleta 2012; Serra-Diaz et al. 2014), especially where fragmented populations are unable to disperse at distances required to maintain healthy gene flow (Grivet et al. 2008). Persistence in moist, shaded microsites may allow oaks to resist increasing drought stress; however, greater drought sensitivity in saplings could constrain the successful regeneration of mature oak woodlands (McLaughlin & Zavaleta 2012; McLaughlin et al. 2014). Grazed areas may also be less able to regenerate following wildfire compared to ungrazed areas (Arévalo et al. 2009; Vuln. Assessment Workshop, pers. comm., 2017).

Management potential

Public and societal value

Regional experts evaluated oak savannas and open woodlands as having moderate-high public and societal value (high confidence in evaluation).

Oak savannas and open woodlands are valued by the public for their aesthetic beauty, recreational opportunities, wildlife habitat, and grazing areas (Huntsinger & Fortmann 1990; Oviedo et al. 2012; Vuln. Assessment Workshop, pers. comm., 2017), and are also of critical importance to northern California tribes (Anderson 1993, 2005; Anderson & Lake 2013; Risling Baldy 2013). However, the abundance and distribution of oak savannas and open woodlands have been greatly reduced by a variety of means since Euro-American colonization in the 1850s, including clearing under public policies until at least the 1970s (Rossi 1980; Holmes 1990; Mensing 2006; Whipple et al. 2011). Over the past 15-20 years, state legislators have passed measures such as the California Oak Woodlands Conservation Act of 2001 (Cal. Assem. Bill No. 242), which recognizes the importance of oak woodlands and provides incentives for their protection, and the Oak Woodlands Conservation Act of 2004 (Cal. Sen. Bill No. 1334), which requires management plans for oak woodlands at the municipal and county levels (Gaman & Firman 2006; Green & Magnuson 2011). Additionally, legislation in 2016 has made it easier for land managers to remove encroaching conifers from Oregon white oak woodlands in northern California without being subject to conifer restocking requirements (Cal. Assem. Bill No. 1958; CAL FIRE 2017). However, replanting is not required when this species is commercially harvested, and the harvest and removal of most other oak species are not regulated at all (Green & Magnuson 2011; CAL FIRE 2017; Vuln. Assessment Reviewer, pers. comm., 2018). Existing regulations and perceptions of risk may also complicate efforts to reintroduce frequent fire into oak savannas and open woodlands (Green & Magnuson 2011).

Existing policies can also compound tribal vulnerabilities to climate change impacts (Voggesser et al. 2013; Norton-Smith et al. 2016). For instance, the historical exclusion of fire from oak woodlands has resulted in loss of access to cultural resources and reduced food security in many tribal communities (Richards & Creasy 1996; Jewell & Vilsack 2014; Norgaard 2014a, 2014b). The loss of acorn groves and orchards that have been managed and utilized as gathering sites for centuries impacts cultural and personal identity, health, and traditional ecological knowledge of tribal members no longer able to practice traditional ways of life (Jewell & Vilsack 2014; Norgaard 2014a; Norgaard et al. 2016).

Management capacity and ability to alleviate impacts[11]

Regional experts evaluated the potential for reducing climate impacts on oak savannas and open woodlands through management as low-moderate (moderate confidence in evaluation). Regional experts identified multiple use conflicts and/or competing interests for oak savannas and open woodlands, including recreation, cultural/tribal access and allowed uses, grazing, and development (Vuln. Assessment Workshop, pers. comm., 2017).

The primary causes of oak savanna and open woodland declines in northern California are anthropogenic, and include clearance for grazing and agriculture, fuelwood harvest, fire suppression activities, and habitat fragmentation (Rossi 1980; Holmes 1990; Mensing 2006). Several studies have documented successful restoration in oak woodlands, especially Oregon white oak woodlands impacted by fire exclusion and resulting conifer encroachment (Agee 1999; Biswell 1999; Devine & Harrington 2006, 2013; Devine et al. 2007a, 2007b; Hankins 2015). This suggests that there is high potential for more effective management to successfully support climate adaptation within oak-dominated ecosystems (Vuln. Assessment Reviewer, pers. comm., 2017).

Restoration of frequent low-intensity fire is widely recognized to be a critical restoration strategy for oak woodlands (Martin & Sapsis 1992; Neary et al. 1999; Hessburg & Agee 2003; Senos et al. 2006; Engber et al. 2011; Hankins 2013, 2015), and could occur through a return to cultural burning practices or the implementation of non-tribal prescribed burns (Agee 1996; Biswell 1999; Shebitz 2005; Skinner et al. 2006; Hankins 2009, 2015; Jewell & Vilsack 2014; Lake

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

& Long 2014; Long et al. 2016; Taylor et al. 2016). Fire is also viewed as an important tool for enhancing rangeland productivity, but the use of fire as a restoration strategy is less widely accepted in foothill oak woodlands compared to Oregon white oak woodlands due to the risks of tree mortality and lower post-fire regeneration rates (Vuln. Assessment Reviewer, pers. comm., 2018). However, oak woodlands were frequently burned by northern California tribes prior to Euro-American settlement and the implementation of fire suppression (Mensing 2006; Anderson 2007; Hankins 2015), and prescribed fire has been shown to enhance native vegetation in foothill oak woodlands (Hankins 2013, 2015).

Although frequent fire is required for ongoing persistence of oak woodlands at the landscape scale (Mensing 2005), more intensive management (e.g., mechanical treatments) may be required to initiate and maintain recovery in Oregon white oak stands compromised by decades of fire suppression (Devine & Harrington 2013). Because low-intensity fire is not hot enough to injure and kill larger conifers, mechanical thinning is required to release suppressed oaks from competition and increase tree vigor where oak canopies have been pierced and overtopped by conifers (Devine & Harrington 2006, 2013). Rapid improvements in growth, stimulation of epicormic sprouting, and increased acorn production are observed following release (Devine & Harrington 2006, 2013). However, the removal of overtopping conifers increases site vulnerability to invasive species by altering the microclimate and condition of the forest floor (Devine et al. 2007b). Post-treatment control of invasive grasses/forbs and conifer seedlings is necessary to allow released oaks to become reestablished and maintain dominance (Agee 1996; Stuart & Stephens 2006; Devine et al. 2007b).

Habitat restoration efforts in degraded oak savannas and open woodlands dominated by blue oak and valley oak have not proven to be as successful as those focused on Oregon white oak woodlands (Vuln. Assessment Reviewer, pers. comm., 2018). Future persistence of habitats dominated by blue oak and valley oak may depend on management actions that assure favorable conditions for germination, growth, and reproduction (Vuln. Assessment Reviewer, pers. comm., 2018). For instance, ungulate and rodent exclusion can significantly reduce acorn and seedling/sapling herbivory, which contribute to low recruitment rates (MacDougall et al. 2010; Davis et al. 2011); exclusion is particularly effective when paired with restoration plantings (Swiecki & Bernhardt 1998; McCreary 2004, 2009). The current regional abundance and distribution of blue oak and valley oak under future climate conditions is not assured (Vuln. Assessment Reviewer, pers. comm., 2018). However, protection of blue oak and valley oak woodlands may allow persistence of these habitats in climatically suitable areas and in microrefugia that alleviate water stress (McLaughlin et al. 2014, 2017), especially in the foothills of the Coast Range and in the inner Coast Range south of Clear Lake (including Berryessa Snow Mountain National Monument) (Kueppers et al. 2005; Barbour & Kueppers 2012; Vuln. Assessment Reviewer, pers. comm., 2018).

Finally, the ecological and cultural potential importance of Oregon white oak woodlands in a changing climate is often underemphasized in northern California (Vuln. Assessment Reviewers, pers. comm., 2018). Oregon white oak is particularly well-suited to warmer, drier climate conditions (Hahm et al. 2018), and habitat shifts and/or range expansion for this species are likely across the region (Yospin et al. 2015; Vuln. Assessment Reviewer, pers. comm., 2018). As sudden oak death continues to spread northwards and inland, potentially causing mortality in black oaks, the remaining white oak woodlands are likely to become more important for birds and other wildlife (Vuln. Assessment Reviewer, pers. comm., 2018). However, the current high rates of conifer encroachment in Oregon white oak woodlands may result in the elimination of large trees in older age classes before climate change begins to more dramatically reshape vegetation patterns more favorable to their establishment (Vuln. Assessment Reviewer, pers. comm., 2018). Further research and greater focus on habitat restoration efforts in Oregon white oak woodlands could facilitate their persistence on the landscape, despite potential shifts in species composition, woodland structure, and habitat distribution (Devine & Harrington 2006; Devine et al. 2007a; Devine & Harrington 2013).

Ecosystem services

Oak savannas and open woodlands provide a variety of ecosystem services, including:

- Provisioning of food, fiber, fuel, genetic resources, natural medicines, habitat for wildlife and plant species, and shade for livestock;
- Regulation of climate/microenvironments (e.g., shade) and flood/erosion control;
- Support of primary production, carbon sequestration, nutrient cycling, and, to a lesser degree, water cycling; and
- Cultural/tribal uses for traditional foods, spiritual/religious purposes, knowledge systems, educational values, aesthetic values, social relations, sense of place, cultural heritage, inspiration, and recreation (Vuln. Assessment Workshop, pers. comm., 2017).

Oak-dominated habitat types in California are among the most important in the state, providing habitat for 330 species of birds, mammals, reptiles, and amphibians that depend on them at some stage in their life cycle (CalPIF 2002).

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

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Northern California Climate Adaptation Project:
Vulnerability Assessment Methods and Application

Defining Terms

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

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

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

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

Vulnerability Assessment Model

The vulnerability assessment model applied in this process was developed by EcoAdapt (EcoAdapt 2014a; EcoAdapt 2014b; Kershner 2014; Hutto et al. 2015; Gregg 2018),[12] and includes evaluations of relative vulnerability by local and regional stakeholders who have detailed knowledge about and/or expertise in the ecology, management, and threats to focal habitats, species groups, individual species, and the ecosystem services that these resources provide. Stakeholders evaluated vulnerability for each resource by discussing and answering a series of questions for sensitivity and adaptive capacity. Exposure was evaluated by EcoAdapt using projected future climate changes from the scientific literature. Each vulnerability component (i.e., sensitivity, adaptive capacity, and exposure) was divided into specific elements. For example, habitats included three elements for assessing sensitivity and six elements for adaptive capacity. Elements for each vulnerability component are described in more detail below.

In-person workshops were held in Eureka, Redding, and Upper Lake between May and October 2017. Participants self-selected habitat and species group/species breakout groups and evaluated the vulnerability of each resource. Participants were first asked to describe the habitat and/or to list the species to be considered in the evaluation of an overarching species group. Due to limitations in workshop time and participant expertise, multiple resources were not assessed during these engagements. Evaluations for remaining habitats, species groups, and species were completed by contacting resource experts.[13]

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

[13] Resources evaluated by experts included: coastal bluff/scrub habitats, coastal conifer hardwood forest, true fir forest, lakes/ponds, freshwater marshes, vernal pools, seeps/springs, native insect pollinators, native ungulates, salamanders, frogs, native mussels, marbled murrelet, and northwestern pond turtle.

Stakeholders assigned one of five rankings (High, Moderate-High, Moderate, Low-Moderate, or Low) for sensitivity and adaptive capacity. EcoAdapt assigned rankings for projected future climate exposure. Rankings for each component were then converted into scores (High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1), and the scores were averaged (mean) to generate an overall score. For example, scores for each element of habitat sensitivity were averaged to generate an overall habitat sensitivity score. Scores for exposure were weighted less than scores for sensitivity and adaptive capacity because the uncertainty about the magnitude and rate of future change is greater. Sensitivity, adaptive capacity, and exposure scores were combined into an overall vulnerability score calculated as:

Vulnerability = [(Climate Exposure*0.5) x Sensitivity] - Adaptive Capacity

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

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

Vulnerability Assessment Model Elements

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

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

Sensitivity & Exposure (Applies to Species Groups and Species)

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

Sensitivity & Exposure (Applies to Species ONLY)

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

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

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

Adaptive Capacity (Applies to Habitats ONLY)

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

Adaptive Capacity (Applies to Species Groups, Species)

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

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