



Mixed Grasslands

Northern California Climate Change Vulnerability Assessment Synthesis

An Important Note About this Document: *This document represents an initial evaluation of vulnerability for mixed grasslands in northern California based on expert input and existing information. Specifically, the information presented below comprises vulnerability factors selected and scored by regional experts, relevant references from the scientific literature, and peer-review comments and revisions (see end of document for a glossary of terms and brief overview of study methods). The aim of this document is to expand understanding of habitat vulnerability to changing climate conditions, and to provide a foundation for developing appropriate adaptation responses.*

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Habitat Description

Northern California grassland types considered in this assessment include interior grasslands (also known as valley grasslands), coastal prairie grasslands, and serpentine grasslands (found as discrete habitats on ultramafic soils; Bartolome et al. 2007; Ford & Hayes 2007; Keeler-Wolf

et al. 2007; Eskelinen & Harrison 2013; Eviner 2016).¹ Across these grassland habitat types (collectively referred to here as mixed grasslands), vegetation includes perennial and annual native and non-native species, and most native vegetation is drought- and fire-adapted (Bartolome et al. 2007; Ford & Hayes 2007; Eviner 2016). Mixed grassland species also represent key components of other California ecosystems (e.g., oak woodlands, chaparral; Keeler-Wolf et al. 2007; Eviner 2016).

In the study area, interior grassland distribution covers the northern portion of the Central Valley and extends up to 700 m (2,295 ft) in surrounding coastal hills and foothills (Keeler-Wolf et al. 2007). Interior grasslands on non-serpentine soils are typically dominated by non-native annual forbs, including broadleaf filaree (*Erodium botrys*) and redstem filaree (*E. cicutarium*), and non-native annual grasses, including soft brome (*Bromus hordeaceus*), ripgut brome (*B. diandrus*), foxtail brome (*B. madritensis* ssp. *rubens*), wild oats (*Avena fatua*), barbed goat grass (*Aegilops triuncialis*), and medusahead (*Elymus caput-medusae*; Eviner 2016; Vuln. Assessment Reviewers, pers. comm., 2018). Some native perennial grasses persist, including purple needlegrass (*Stipa pulchra*), valley wild rye (*Elymus triticoides*), blue wild rye (*E. glaucus*), and California brome (*B. carinatus*), albeit with limited growth, survival and seed establishment due to competition with non-native species (Eviner 2016).

In the study area, coastal prairie grasslands occur on coastal bluffs and the western slopes of the Coast Range (Ford & Hayes 2007; Eviner 2016), ranging up to 914 m (3,000 ft) in elevation (Vuln. Assessment Workshop, pers. comm., 2017). Relative to interior grasslands, coastal prairie grasslands typically have higher dominance of both native and non-native perennial herbaceous species due to overall wetter conditions as a result of higher precipitation and coastal fog. In the study area, common native species include California brome, blue wild rye, Lemmon's needlegrass (*S. lemmonii*), tufted hairgrass (*Deschampsia caespitosa*), wood rush (*Luzula comosa*), Idaho fescue (*Festuca idahoensis*), meadow barley (*Hordeum brachyantherum*), leafy reed grass (*Calamagrostis foliosa*), and various *Carex* species (Eviner 2016; Vuln. Assessment Reviewers, pers. comm., 2018). Coastal prairie grasslands are susceptible to invasion by exotic species, and to successional encroachment by shrubs and conifers in the absence of disturbance (e.g., burning, grazing; Eviner 2016). Common invasive species in coastal prairie grasslands include silvery hairgrass (*Aira caryophylllea*), pampas grass (*Cortaderia* spp.), orchard grass (*Dactylis glomerata*), Italian rye grass (*Festuca perennis*), common velvet grass (*Holcus lanatus*), dogtail grass (*Cynosurus echinatus*), purple awned wallaby grass (*Rytidosperma penicillatum*), tansy ragwort (*Senecio jacobaea*), and woody invasive species such as Scotch broom (*Cytisus scoparius*) and French broom (*Genista monspessulana*; Vuln. Assessment Reviewers, pers. comm., 2018).

In the study area, serpentine grasslands predominately occur in the Northern Coast Range, the Northern Interior Coast Range, the Klamath Mountains, and the northernmost Sierra Nevada foothills (Harrison & Viers 2007). They typically occur as islands on shallow slopes and alluvial valleys interspersed with other, more dominant woody serpentine communities (e.g.,

¹ Alpine grasslands are addressed separately in their own assessment.

chaparral, conifer woodlands; Harrison & Viers 2007). Serpentine grasslands are characterized by ultramafic soils with low macronutrients and calcium levels, very high magnesium and iron, and generally rocky and dry conditions (Harrison & Viers 2007; Eskelinen & Harrison 2013). Serpentine, as well other grassland types on low fertility soils, harbor high endemism and much remnant grassland native biodiversity, and flora is adapted to the unique soil conditions (Harrison & Viers 2007; Eviner 2016). Most serpentine sites are dominated by native annual forbs, including the genera *Acmispon*, *Calochortus*, *Calycadenia*, *Clarkia*, *Eschscholzia*, *Gilia*, *Hesperolinon*, *Lasthenia*, *Layia*, *Lessingia*, *Linanthus*, *Madia*, *Navarretia*, *Stellaria*, *Streptanthus*, and *Trifolium* (Harrison & Viers 2007; Keeler-Wolf et al. 2007). Serpentine sites may feature some native perennial and annual grasses (e.g., purple needlegrass, foothill needlegrass [*S. lepida*], small fescue [*F. microstachys*]), as well as exotic annual grasses (e.g., soft brome, Italian rye grass, barbed goat grass) that can comprise more than 50% cover, especially by late spring (Harrison & Viers 2007).

Mixed grasslands support a variety of species of cultural importance. Iris (various species) and grass species were historically used for twine and cordage, and grasses were also used for thatch (Ford & Hayes 2007). A variety of forbs are utilized for medicinal purposes, and tribes also harvest bulbs and tubers of Indian potatoes, which include numerous geophyte plant species (e.g., *Brodiaea* spp., *Dichelostemma* spp., *Triteleia* spp., *Calochortus* spp., *Lilium* spp., *Fritillaria* spp., and *Allium* spp.; Lake 2007; Norgaard et al. 2016; Karuk Tribe 2019). Mixed grasslands are also critical foraging areas for Roosevelt elk (*Cervus canadensis roosevelti*), which tribes traditionally used for food, clothing (hides), regalia, and implements (Lake 2007; Norgaard et al. 2016; Karuk Tribe 2019).

Executive Summary

The relative vulnerability of mixed grasslands in northern California was evaluated as moderate by regional experts due to moderate-high sensitivity to climate and non-climate stressors, moderate exposure to projected future climate changes, and moderate adaptive capacity.²

Mixed Grasslands	Rank	Confidence
Sensitivity	Moderate-High	High
Future Exposure	Moderate	Low
Adaptive Capacity	Moderate	High
Vulnerability	Moderate	Moderate

² “Mixed grasslands” refers to interior, coastal prairie, and serpentine grasslands. Vulnerability scores were assessed cumulatively, but differences between the three sub-types are highlighted wherever possible.

Sensitivity & Exposure Summary	<p><u>Climate and climate-driven factors:</u></p> <ul style="list-style-type: none"> • Precipitation amount and timing, soil moisture, climatic water deficit, drought, air temperature <p><u>Disturbance regimes:</u></p> <ul style="list-style-type: none"> • Wildfire <p><u>Non-climate stressors:</u></p> <ul style="list-style-type: none"> • Invasive and problematic species, fire suppression, livestock grazing, roads/highways/trails, pollution <p><u>Other sensitivities:</u></p> <ul style="list-style-type: none"> • Decoupling of plant and pollinator phenologies
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Mixed grasslands are sensitive to climate stressors that alter plant germination, recruitment, and growth, including precipitation amount and timing, soil moisture, drought, and air temperature. Changes in these factors are likely to influence overall grassland distribution, and drive shifts in community composition and functional group dominance, potentially increasing the already extensive dominance of exotic species. Additionally, these factors may cause shifts in plant phenologies, leading to a mismatch with pollinator life cycles and resultant impacts on plant diversity and pollinator populations. Increasing wildfire as a result of climate change will likely expand grassland extent, although areas will primarily be comprised of exotic annual species. Shifts in fire frequency and intensity may also alter exotic species distribution and dominance in remnant native grassland areas, as well as impact soil health. Invasive exotic species dominate most interior grassland areas, and represent a significant stressor for remnant native-dominated grasslands (e.g., coastal prairies, serpentine grasslands). Exotic species outcompete and exclude native species, as well as cause shifts in habitat function (e.g., burn frequency and intensity). Livestock grazing, roads/highways/trails, and nitrogen deposition all have the potential to increase the competitive advantage of exotic species, further impacting mixed grassland composition and function. Additionally, coastal prairies are susceptible to woody vegetation encroachment in the absence of disturbances such as fire.³

Adaptive Capacity Summary	<p><u>Factors that enhance adaptive capacity:</u></p> <ul style="list-style-type: none"> + Extensive habitat distribution within study region and state + High topographical, functional group, and species diversity + Valued by the public for a variety of uses (grazing, recreation, biodiversity, aesthetics) <p><u>Factors that undermine adaptive capacity:</u></p> <ul style="list-style-type: none"> – Increasing habitat fragmentation, loss, and degradation – High prevalence of exotic species; exotic species may be better able to take advantage of climatic shifts than native species – Many perennial species not resilient to exotic invasion and increasing disturbance due to low recruitment and slow growth – Hard to generalize management recommendations across the landscape
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³ Although it is a natural process, woody vegetation encroachment (i.e., natural succession to woody climax communities) is considered a threat in the context of this assessment because it reduces mixed grassland habitat.

Mixed grasslands are widely distributed in northern California, but are experiencing increased habitat fragmentation and loss due to conversion and degradation by human land uses. In addition to human land-use changes, coastal prairies are also declining in geographic extent due to woody vegetation encroachment. Grassland composition has been significantly altered by exotic species. Perennial grassland species are resilient to some climatic shifts, but low recruitment and slow growth limit resilience to and recovery from the extensive invasion of exotic annuals and increasing human disturbance. Annual species' seedbank dynamics (e.g., high seed volume, prolonged dormancy) enhance their ability to recover after disturbance or in response to climatic variability. High topographical, functional group, and species diversity generally increases resilience of this habitat to climatic changes, although some components (e.g., forbs) will likely exhibit overall higher sensitivity due to their life histories, rarity, or location. In the face of climate impacts, grassland management could include targeted and site-specific grazing, prescribed burning (including cultural burning), and restoration plantings.

Sensitivity and Exposure

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

Mixed grassland area, particularly exotic-dominated annual grassland, is projected to increase across California by the end of the century as a result of climate change (Lenihan et al. 2008; Butz et al. 2015; Thorne et al. 2016). Lenihan et al. (2008) suggests that gains may occur in the Interior Northern California Coast Range, and current landscape distribution may change across the study area (Lenihan et al. 2008; Butz et al. 2015; Thorne et al. 2016). Grassland expansion is projected largely due to type conversion of oak woodlands and shrublands from higher fire frequencies and reductions in effective moisture (Haidinger & Keeley 1993; George & Alonso 2008; Holmes et al. 2008; Lenihan et al. 2008; Keeley & Brennan 2012).

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 climatic suitability for grasslands by the end of the century (see table below).⁴ Thorne et al. (2016) modeled climatic exposure and suitability for different grassland macrogroups, including "California annual and perennial grassland"⁵ (hereafter annual and perennial grassland) and

⁴ 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).

⁵ Includes "needlegrass grasslands of the foothills, valleys, and Coast ranges, and the largely non-native annual grasslands and weed patches in the dry, warm summer regions of California". Does not include "cool, moist north coastal terrace prairies, montane meadow/upland grasses, and non-native perennial pasture grasses" (Thorne et al. 2016).

“Western North America temperate grassland and meadows”⁶ (hereafter temperate grassland and meadows). The annual and perennial grassland macrogroup includes native and exotic annual and perennial species (e.g., native perennial needlegrass species, native annual poppies [*Eschscholzia californica*] and goldfields [*Lasthenia* spp.], and exotic wild oat, brome, and annual fescue), and the temperate grassland and meadows macrogroup includes both native and exotic perennials (e.g., native Idaho fescue and blue wild rye, exotic Harding grass [*Phalaris aquatica*] and velvet grass).

In northern California, warmer and wetter future conditions are projected to create more climatically exposed annual and perennial grassland and temperate grassland and meadow area than hotter and drier conditions (Thorne et al. 2016, 2017). Current mapped annual and perennial grassland areas that are likely to become climatically exposed under most future climate scenarios include the majority of the northernmost Sierra Nevada foothills, parts of the Northern California Coast (i.e., the southern segment, and near Cape Mendocino), and the northern portion of the Northern California Interior Coast Range (Thorne et al. 2016). Future spatial differences in climatic exposure and suitability for temperate grassland and meadows are difficult to summarize due to the limited current extent of this habitat type combined with the scale of modeling used in the Thorne et al. (2016) assessment.

Significant current annual and perennial grassland and temperate grassland and meadow area is also projected to remain climatically suitable in northern California by the end of the century, representing potential refugia for these habitat types. Under most future scenarios, the central portion of the Northern California Interior Coast Range represents the primary refugia zone for annual and perennial grassland in northern California, although the southern Cascades could act as refugia under high emissions scenarios, and the Northern California Coast Range could act as refugia in a low emissions hotter/drier scenario (Thorne et al. 2016, 2017). In general, refugia areas for both annual and perennial grassland and temperate grassland and meadows in northern California are likely to be dictated by topography (e.g., north slopes; Butz et al. 2015) and coastal influence (e.g., fog; PRBO Conservation Science 2011).

Potential Changes in Habitat Distribution by 2100

- *California annual and perennial grassland*: 24–60% of the state-wide current vegetation distribution is projected to experience an increase in climatic stress, while 18–48% will remain within climatically suitable areas; 10–52% of the projected climatically suitable habitat will be in newly suitable areas
- *Western North America temperate grassland and meadow*: 2–26% of the state-wide current vegetation distribution is projected to experience an increase in climatic stress, while 43–67% will remain within climatically suitable areas; 0–9% 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

⁶ In the study area, occurs in the “hills and mountains of the north Coast Ranges, Klamath Mountains, lower montane Sierra Nevada, and Modoc Plateau” (Thorne et al. 2016).

Additionally, as the climate changes, areas not currently occupied by annual and perennial grassland and temperate grassland and meadows may become newly climatically suitable, representing migration opportunities. In northern California, the largest increases in newly climatically suitable areas for annual and perennial grassland occur primarily under hotter/drier scenarios and at higher elevations, with newly suitable areas appearing in the southern Cascades, Klamath Mountains, Northern California Coast Range, and in many areas along the Northern California Coast (Thorne et al. 2016, 2017). Temperate grassland and meadows will experience significantly less newly climatically suitable areas than annual and perennial grasslands; newly suitable temperate grassland and meadow areas are predicted to emerge only under lower emissions scenarios, and will occur at slightly higher elevations than current suitable habitat near and slightly north of Cape Mendocino on the Northern California Coast (Thorne et al. 2016, 2017).

Predicting future persistence and distribution for serpentine grasslands is difficult. As soil specialists, endemic serpentine species are geographically tied to appropriate soils, so persistence in existing locations will largely require maintenance of suitable climatic conditions (Damschen et al. 2012). Migration opportunities to new locations will be limited by soil specificity and the natural isolation and fragmentation of unique soil habitats (Damschen et al. 2012). Soil specialists will 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 mixed grasslands as having moderate-high sensitivity to climate and climate-driven factors (high confidence in evaluation), with overall moderate-high future exposure to these factors within the study region (moderate confidence). Key climatic factors that affect mixed grasslands include precipitation amount and timing, soil moisture, climatic water deficit, drought, and air temperature.⁷ Synergistic effects from multiple factors (e.g., shift in evapotranspiration demand due to changes in soil moisture and air temperature) will create more severe impacts than factors considered in isolation (Vuln. Assessment Workshop, pers. comm., 2017).

Precipitation amount and timing

Precipitation amount and timing controls mixed grasslands at multiple levels: the presence and persistence of grassland communities is influenced by the seasonality and amount of precipitation (Eviner 2016), while high inter-annual precipitation variability influences grassland species composition, species abundance, and productivity at a given site (Jackson & Bartolome 2002; Dukes et al. 2005; Dukes & Shaw 2007; Suttle et al. 2007; Chou et al. 2008). Differences in precipitation amount and timing also drive differences in vegetative communities between

⁷ All climate and climate-driven factors presented were ranked as having a moderate or higher impact on this habitat type.

inland and coastal prairie grassland sites, with coastal communities typically having higher dominance of perennial and woody species, and interior grasslands having higher annual plant dominance (Eviner 2016).

Shifts in precipitation amount will likely alter species composition, species abundance, and grassland productivity. In non-serpentine soils, higher precipitation is correlated with enhanced native plant diversity and productivity, but also increased productivity and growth of exotic annual grasses (Suttle et al. 2007; Chou et al. 2008). Lower precipitation in annual grasslands, particularly over consecutive years (i.e., drought), may provide an advantage to native annual forbs over exotic annual grasses (see drought discussion below; LaForgia et al. 2018). In general, serpentine grasslands exhibit less variability in species composition and biomass in response to interannual precipitation variability than non-serpentine grasslands, largely because nutrient limitations exhibit a stronger control on community composition and biomass than precipitation (Fernandez-Going et al. 2012; Eskelinen & Harrison 2013; Fernandez-Going & Harrison 2013).

Precipitation recharges soil moisture, which, in turn, controls the length of the growing season and plant productivity (Eviner 2016). Low rainfall years may prevent deep soil recharge, potentially affecting perennial grass establishment, root development, and persistence, particularly in the presence of exotic annual species which compete for shallow soil water resources (Reever Morgan et al. 2007). Dry conditions across multiple successive years may facilitate shifts from perennial to annual species dominance in remnant perennial grasslands (Reever Morgan et al. 2007; Pfeifer-Meister et al. 2016).

Precipitation timing is believed to have more influence on grassland species composition, diversity, cover, and plant productivity than precipitation volume. In general, annual grassland species are less tolerant of low precipitation and shifts in precipitation timing than perennial species (Bartolome et al. 2014; Eviner 2016). Perennial grasses may be less affected by interannual variability in precipitation timing and volume (Brandt & Seabloom 2011) as long as sufficient moisture percolates into deeper soil horizons. Fall rain events signal the start of grassland growing season, and the timing of these and subsequent rain events in winter and spring influence community composition and cover (e.g., dominance of grasses and forbs) by affecting plant germination (Eviner 2016). For example, periods of low fall, winter, or spring precipitation following germinating fall rains favor native forbs and perennial grasses over annual grasses, because initial rains stimulate annual grass germination and deplete the seed bank, but a lack of subsequent rain limits annual grass seed production and survival in the soil seedbank (Corbin et al. 2005; Keeler-Wolf et al. 2007; Eviner 2014). Reductions in mid-winter precipitation volume have been linked with declining interior grassland plant diversity in northern California, particularly amongst drought-intolerant native forbs with high specific leaf area (Harrison et al. 2015b). Shifts in spring rainfall timing and volume alter seed production timing and the volume of seeds produced, although exact impacts vary by species (Eviner 2016). High late spring rainfall has been found to bolster species richness in lower elevation grasslands in northern California even in the wake of severe multi-year drought conditions,

demonstrating the complex interactions of moisture availability and timing and temperature on grassland function (Copeland et al. 2016).

Altered precipitation timing may also facilitate establishment of new exotics and/or increase cover of existing problematic exotic species. For example, more late spring or early summer precipitation may increase the abundance and productivity of non-native annuals that thrive through late summer, including medusahead and barbed goat grass, and non-native forbs like yellow starthistle (*Centaurea solstitialis*; Eskelinen & Harrison 2013; Eviner 2014). Similarly, increased late spring and early summer rainfall could also enhance abundance of winter annuals by altering community-level interactions (Suttle et al. 2007). Suttle et al. (2007) found that short-term (<2 years) increases in nitrogen-fixing forb productivity due to more late spring precipitation resulted in longer-term (5 year) declines in grassland species richness in northern California. Winter annual grasses experienced a lagged benefit from enhanced nitrogen availability via high forb production and subsequent decomposition. Over time, higher winter annual grass productivity led to significant thatch accumulation and subsequent loss of native forbs, demonstrating that community-level interactions can mask or reverse initial changes (Suttle et al. 2007). Non-serpentine grasslands and grasslands that are already invaded are particularly vulnerable to increased exotic abundance as the timing of precipitation shifts. Comparatively, serpentine grasslands appear more resilient to precipitation-driven changes in exotic species establishment in the absence of nutrient addition (e.g., nitrogen deposition; Eskelinen & Harrison 2013).

Regional Precipitation Trends ⁸	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> • 2.6–9.4 cm (1.0–3.7 in) increase in mean annual precipitation between 1900 and 2009 for the Northwestern California, Southern Cascade, and Great Valley ecoregions (Rapacciuolo et al. 2014) 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> • 23% decrease to 38% increase in mean annual precipitation by 2100 (compared to 1951–1980) for the North Coast, Northern Coast Range, Northern Interior Coast Range, Klamath Mountain, Southern Cascade, and Great Valley ecoregions (Flint et al. 2013; Flint & Flint 2014)⁹ • 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

⁸ 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.

⁹ Projections for changes in annual and seasonal precipitation by ecoregion can be found in the full climate impacts table (<https://bit.ly/2LHgZaG>).

Regional Precipitation Trends ⁸	
	season becomes drier and longer (Pierce et al. 2018; Swain et al. 2018) <ul style="list-style-type: none"> • Overall, interannual variability is expected to increase (Pierce et al. 2018; Swain et al. 2018)
Summary of Potential Impacts on Habitat <i>(see text for citations)</i>	
<ul style="list-style-type: none"> • Changes in grassland distribution • Changes in species composition, functional group dominance, species abundance, and productivity, particularly in non-serpentine grasslands, but impacts are highly variable • Potential for establishment of new exotic species or increased cover of existing exotic species 	

Soil moisture and climatic water deficit

In general, grasslands occupy drier sites than other vegetative communities in the region (Vuln. Assessment Workshop, pers. comm., 2017). Coastal prairie grasslands typically experience less summer moisture stress than interior grasslands due to fog inputs (Eviner 2016), with fog inputs accounting for 28–66% of perennial grass root water uptake during seasonal drought periods (Corbin et al. 2005). Coastal prairie grasslands have evolved to take advantage of this summer water availability; for example, in coastal areas, perennial purple needlegrass has abundant shallow roots to harvest fog water inputs (Corbin et al. 2005). Reduced fog could limit the abundance of native species that rely on fog for a majority of summer water availability (e.g., reedgrass [*Calamagrostis nutkaensis*]), or increase vulnerability to annual exotic species invasion by reducing the competitive advantage of perennial life histories (Corbin et al. 2005). Alternatively, increased summer fog could facilitate the invasion of exotic perennial grasses (e.g., velvet grass; Corbin et al. 2005). Species richness in coastal prairie grasslands also varies with moisture, with wetter sites typically having higher species richness (Ford & Hayes 2007).

Moisture stress in both coastal and inland grasslands is also likely to change as a result of precipitation changes and increasing evaporative stress. Climatic water deficit (CWD) is a “plant-relevant” measurement of moisture stress that takes into account the interaction between water (precipitation) and energy (temperature; Stephenson 1998).¹⁰ Although the direction and amount of change in future precipitation projections for California are highly uncertain, warmer temperatures and associated increases in evaporative demand mean that even areas where precipitation may increase are expected to see a rise in CWD (Thorne et al. 2015; Micheli et al. 2018). Enhanced CWD will likely increase summer drought stress in grassland systems (Stephenson 1998), with subsequent impacts on recruitment, species composition, and landscape distribution (Reever Morgan et al. 2007; Rapacciuolo et al. 2014; Eviner 2016). For example, overall drier conditions contributed to declines in grassland community diversity from 2000–2014, especially in sites that were already water-limited (Harrison et al. 2015b). Grassland community composition also shifted toward species with

¹⁰ 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).

more “stress tolerant” traits (e.g., lower leaf specific area and leaf water content) in response to increasing aridity (Harrison et al. 2015a, 2015b). Serpentine grasslands and grasslands on other infertile soil types exhibited a slower decline in species diversity and less change in community composition compared to higher-productivity grasslands (Harrison et al. 2015a). Researchers link this resilience with the high prevalence of stress-tolerant functional traits in these communities, which have developed over time in response to the harsh growing conditions on these unique soils (Fernandez-Going et al. 2012; Fernandez-Going & Harrison 2013).

Changes in community composition, regardless of cause, will likely drive further changes in water availability by affecting soil porosity, infiltration, and plant uptake. For example, loss of perennial species may limit water penetration via root channels to deeper soil horizons, while simultaneously reducing community-wide utilization of deep soil water resources (Reever Morgan et al. 2007). This creates a positive feedback loop where community changes alter soil moisture, which then further alters community composition. For example, in the Central Valley, conversion of perennial grasslands to annual grasslands reduced community utilization of deep water resources, and the resulting abundant deep water resources facilitated invasion by yellow starthistle, a deep-rooting exotic forb. Once established, yellow starthistle reduces both shallow and deep water resources, perpetuating native species declines due to dry conditions (Reever Morgan et al. 2007 and citations therein).

Changes in moisture availability will also alter the length of grassland growing seasons and the amount of plant growth. A lack of moisture during the growing season (e.g., due to mid-season drought) limits aboveground plant productivity senesce (Eviner 2016). Seasonal soil drying in spring and early summer triggers plant senesce (Eviner 2016). Soil moisture changes and enhanced CWD are likely to exacerbate limitations on plant productivity and alter the timing of plant senescence (Reever Morgan et al. 2007; Eviner 2016).

Regional Climatic Water Deficit (CWD) & Soil Moisture Trends	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> • 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, Southern Cascade, and Great Valley ecoregions (Rapacciuolo et al. 2014) • No trends available for soil moisture 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> • Increases in average annual CWD by 2100 (compared to 1951–1980; Flint et al. 2013; Flint & Flint 2014): <ul style="list-style-type: none"> ○ 9–29% increase on the North Coast ○ 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 ○ 4–19% increase in the Great Valley • Increased CWD and decreased top-level soil moisture is likely even if precipitation increases due to temperature-related changes

Regional Climatic Water Deficit (CWD) & Soil Moisture Trends	
	in evaporative demand (Thorne et al. 2015; Micheli et al. 2018; Pierce et al. 2018)
Summary of Potential Impacts on Habitat <i>(see text for citations)</i>	
<ul style="list-style-type: none"> • Reduced grassland diversity and species richness, particularly in already water-limited sites; serpentine sites will exhibit less response • Species composition may shift toward species with more stress-tolerant traits • Altered soil porosity and infiltration as plant community changes, causing further changes in community composition • Altered growing season length, impacting plant productivity 	

Drought

Summer drought reduces soil moisture and grassland productivity, impacting community composition (Eviner 2016). Grassland vegetation exhibits variable adaptations to seasonal drought depending on life history. Annual species complete growth and reproduction in spring, prior to summer drought, and seeds delay germination until fall rains (Corbin & D’Antonio 2004). Perennial species maintain above- and below-ground tissue year-round; in inland areas, native perennial grasses utilize deeper rooting systems to access below-ground water during seasonal drought, and in coastal prairies, they utilize both deeper rooting systems and shallow roots to access moisture available from fog (Corbin & D’Antonio 2004; Corbin et al. 2005). Perennial plants may be more resilient to short-term drought periods than shallow-rooted non-native annual grasses, both through access to deeper soil water sources and the ability to utilize resource pulses more efficiently (Corbin & D’Antonio 2004; Corbin et al. 2005).

Despite these adaptations, intensifying summer drought periods (e.g., warmer and drier) and/or multi-year droughts are likely to reduce herbaceous cover, grassland productivity, and species richness (Copeland et al. 2016). For example, severe drought in 2013–2014 reduced herbaceous cover and species richness on both serpentine and non-serpentine soils and amongst almost all grassland functional groups, but annual grasses and forbs were particularly susceptible (Copeland et al. 2016). Although more resilient to interannual precipitation variability, serpentine grasslands appeared similarly vulnerable to this extreme drought, indicating that critical stress thresholds were exceeded even for generally more stress-tolerant vegetation (Copeland et al. 2016). However, exotic annual grasses experienced the most significant decline, which could have important implications in serpentine areas harboring the highest remnants of native grassland biodiversity (Copeland et al. 2016). Long-term drought periods generally negatively affect most grassland components by impacting plant physiology. For example, in southern California, drought has been linked with declines in leaf photosynthesis for both perennial and exotic annual species (Potts et al. 2012).

Multi-year or severe droughts may also cause shifts in grassland functional group dominance, although the permanence of these shifts will depend on drought length and drought interaction with other factors such as competition, herbivores, pollinators, and belowground plant life stages (Copeland et al. 2016). Shifts in functional groups may include perennial to annual

species in perennial grasslands (Reever Morgan et al. 2007; Pfeifer-Meister et al. 2016) and exotic annual grasses to native annual forbs in annual grasslands (LaForgia et al. 2018). For example, a study in an annual grassland in the Interior North Coast Range found that native forbs with low-specific leaf area increased in both aboveground cover and seed bank abundance during a severe three-year drought while exotic annual grasses declined (LaForgia et al. 2018). This is likely because native forbs can utilize prolonged seed dormancy to weather extreme drought events and then capitalize on reduced competition from exotic grasses (LaForgia et al. 2018). Exotic annual grasses, in comparison, produce a lot of seed annually, but lower seed longevity reduces resilience to long-term drought events (LaForgia et al. 2018).

Regional Drought Trends	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> • Drought years have occurred twice as often over the last two decades compared to the previous century (Diffenbaugh et al. 2015) • 2012–2014 drought set records for lowest precipitation, highest temperatures, and most extreme drought indicators on record (Griffin & Anchukaitis 2014; Diffenbaugh et al. 2015) 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> • Drought years are twice as likely to occur over the next several decades due to increased co-occurrence of dry years with very warm years (Cook et al. 2015) • 80% chance of multi-decadal drought by 2100 under a high-emissions scenario (Cook et al. 2015) • Severe droughts that now occur once every 20 years will occur once every 10 years by 2100 and once-in-a-century drought will occur once every 20 years (Pierce et al. 2018)
Summary of Potential Impacts on Habitat <i>(see text for citations)</i>	
<ul style="list-style-type: none"> • Reduced productivity, herbaceous cover, and species richness in all grassland types; annual grasses will be most affected • Declines in leaf photosynthesis • Altered functional group dominance, including shifts from perennial species to annual species in perennial grasslands, and shifts from annual grasses to annual forbs in annual grasslands 	

Air temperature

Along with changes to rainfall and soil moisture, increasing air temperatures are likely to alter grassland composition and functional group diversity, including increasing dominance of annual and exotic species. Exotic dominance tends to be higher in warmer grassland areas in California (Sandel & Dangremond 2012). Grassland diversity and species richness decline with warmer temperatures, with marked declines in native forbs and increases in annual species, particularly exotic annual grasses (Pfeifer-Meister et al. 2016). These impacts are likely to be more severe in water-limited sites (Pfeifer-Meister et al. 2016). In a study by Sandel and Dangremond (2012) comparing grass species composition across California, the proportion of perennial species in a given area declined in response to increasing temperatures. Community shifts toward exotic species are also likely because C4 grasses tend to fare better under warmer, drier conditions than C3 grasses (Lenihan et al. 2008), and because fall temperatures influence germination patterns (Eviner 2016). For example, low temperatures have been associated with declines in

germination of some annual grass species (Reynolds et al. 2001), while warmer temperatures have been shown to promote earlier annual grass emergence, with negative impacts on native perennial recruitment (Pfeifer-Meister et al. 2016). Additionally, warmer temperatures will likely favor traits typically associated with exotic grasses (e.g., larger seeds, higher specific leaf area), contributing to shifts in community composition, including the potential for some exotic, non-invasive species to become invasive as the climate warms (Sandel & Dangremond 2012).

Warming temperatures are also likely to alter grassland productivity and phenology. Warm spring temperatures typically increase annual grass growth rates (Dyer & Rice 1999; Reever Morgan et al. 2007). Warmer winter temperatures may also increase growth opportunities in previously cold-limited environments for a variety of functional groups (Bartolome et al. 2007; Eviner 2016). Shifts in phenology are also likely with warmer temperatures, as annual grassland senescence timing, as well as perennial grass summer dormancy (Laude 1953), is influenced by temperature rather than precipitation (Zavaleta et al. 2003; Cleland et al. 2006). Thus, warmer temperatures are likely to accelerate emergence, flowering, and senescence of many species (Dukes & Shaw 2007).

Temperature impacts on grassland composition and productivity are likely to be moderated by precipitation, soil moisture, and other factors. For example, studies in central northern California found community responses to temperature were moderated by precipitation and nitrogen deposition (Zavaleta et al. 2003). Additionally, changes in the relative productivity of annual grasses and forbs in California have been associated with precipitation during warmer periods of the growing season in the fall (October–November) or late spring (March–April; Pitt & Heady 1978). In general, however, warmer temperatures are likely to increase potential evapotranspiration and drive larger water deficits in the absence of significant precipitation increases, thus impacting grassland species composition and productivity (Sandel & Dangremond 2012).

The interplay among temperature, precipitation, and seasonality highly influence grassland community dynamics, and will continue to influence grassland responses to climate change. For example, annual grasses generally have rapid germination and growth rates (Reynolds et al. 2001 and citations therein), especially with warmer temperatures (Dyer & Rice 1999; Reever Morgan et al. 2007). Precipitation during warmer times of year will favor the growth of annual grasses relative to other species (Corbin et al. 2007; Keeler-Wolf et al. 2007; Eviner 2014). Comparatively, a lack of precipitation during warm periods appears to favor other vegetative groups (Corbin et al. 2007; Keeler-Wolf et al. 2007; Eviner 2014). Seasonal and annual climatic variability will play a key role in determining grassland composition and diversity in the future (Reever Morgan et al. 2007; Bartolome et al. 2014).

Regional Air Temperature Trends	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> • 0.03°C (0.05°F) decrease to 0.5°C (0.9°F) increase in the average annual temperature 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> • 2.2–6.1°C (4.0–11.0°F) increase in the average annual temperature by 2100 (compared to

Regional Air Temperature Trends	
between 1900 and 2009 for the Northwestern California, Southern Cascade, and Great Valley ecoregions (Rapacciuolo et al. 2014) <ul style="list-style-type: none"> ○ No seasonal temperature trends available 	1951–1980) for the North Coast, Northern Coast Range, Northern Interior Coast Range, Klamath Mountain, Southern Cascade, and Great Valley ecoregions (Flint et al. 2013; Flint & Flint 2014) <ul style="list-style-type: none"> ○ 1.9–5.8°C (3.4–10.4°F) increase in average winter minimum temperatures ○ 2.0–6.8°C (3.6–12.2°F) increase in average summer maximum temperatures
Summary of Potential Impacts on Habitat <i>(see text for citations)</i>	
<ul style="list-style-type: none"> ● Altered community composition and functional group dominance toward exotic species and annual grasses by causing overall drier conditions ● Shifts in germination patterns ● Increased productivity and growth opportunities during growing season ● Phenology shifts toward earlier emergence, flowering, and senescence 	

Sensitivity and future exposure to changes in natural disturbance regimes

Regional experts evaluated mixed grasslands as having moderate sensitivity to changes in natural disturbance regimes (high confidence in evaluation), with an overall low-moderate future exposure to these stressors within the study region (low confidence). Wildfire is the key natural disturbance regime that affects mixed grasslands.¹¹

Wildfire

Prior to Euro-American settlement, regular burning of mixed grasslands by tribal communities was common, particularly in coastal areas (Underwood et al. 2003; Anderson 2005; Bartolome et al. 2007; Stephens et al. 2007; Keeley et al. 2011; Norgaard et al. 2016; Karuk Tribe 2019). Tribes conducted burns to prevent woody species encroachment, convert shrublands to grasslands, maintain and increase herbaceous plant productivity by reducing competition, and maintain hunting grounds by increasing forage for ungulates (Underwood et al. 2003; Anderson 2005; Bartolome et al. 2007; Stephens et al. 2007; Keeley et al. 2011; Norgaard et al. 2016; Karuk Tribe 2019). However, grassland fire regimes have changed significantly over time, declining in areas managed for fire suppression, grazing, and agriculture (Keeley et al. 2011; Cornwell et al. 2012; Norgaard et al. 2016),¹² and increasing in urban-interface areas (Bartolome et al. 2007). Climate change will likely contribute to further changes in wildfire regimes (Westerling et al. 2011; Mann et al. 2016; Parks et al. 2016; Westerling 2018), and thus,

¹¹ Disturbance regimes presented are those ranked as having a moderate or higher impact on this habitat type; additional changes in disturbance regimes that may influence the habitat to a lesser degree include herbivory (e.g., see Hayes & Holl 2003; Bartolome et al. 2014) and wind. Wind is more likely to impact mixed grasslands when there is insufficient vegetation cover due to drought or overgrazing, and/or where soils are finely textured (Vuln. Assessment Reviewers, pers. comm., 2018).

¹² Refer to the section on non-climate stressors for a more complete discussion of the impacts of fire suppression on mixed grasslands.

grassland community composition and distribution (Eviner 2016). Over the long-term, increased fire frequency is projected to increase grassland extent (primarily exotic-dominated annual grassland) throughout California, including the interior northern California Coast Range, at the expense of woodlands and shrublands (Lenihan et al. 2008).

Many native grassland species are fire-adapted (D’Antonio et al. 2006). Wildfire generally stimulates plant growth and reproduction, elevating biodiversity and spatial heterogeneity (Marty et al. 2005). However, site-level impacts are highly variable depending on burn timing, frequency, intensity, and severity (D’Antonio et al. 2006; Keeley et al. 2011; Livingston & Varner 2016). Impacts also vary depending on the species present, fuel moisture, and precipitation and grazing post-burn (D’Antonio et al. 2006; Keeley et al. 2011; Livingston & Varner 2016). For example, too frequent burning leads to communities of invasive annuals (Cornwell et al. 2012), and reduces soil sulfur and nitrogen (D’Antonio et al. 2006). Additionally, high-severity fires can scorch soils (Norgaard et al. 2016) and can volatilize soil nitrogen, carbon particulate matter, and in some areas, mercury (Vuln. Assessment Reviewer, pers. comm., 2018). Burned areas experiencing soil saturation from subsequent heavy precipitation events may be more susceptible to landslides, reducing grassland area (Vuln. Assessment Reviewers, pers. comm., 2018). In general, short-term negative impacts of wildfire include productivity loss (e.g., significant mortality of established perennial bunchgrasses; Marty et al. 2005) and reduced soil moisture (Henry et al. 2006 cited in Eviner 2016). However, fire char deposition and/or nitrogen pulses post-fire could enhance plant growth, particularly for exotics, via nutrient enrichment (Vuln. Assessment Reviewers, pers. comm., 2018).

Regional Wildfire Trends	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> • 85% of U.S. Forest Service lands in northern California are burning less frequently compared to pre-1850 fire return intervals, largely due to fire suppression (Safford & Van de Water 2014) • Changes in large fires (over 400 ha) in the inland northern California/Sierra Nevada region since the 1970s (Westerling 2016): <ul style="list-style-type: none"> ○ 184–274% increase in frequency ○ 270–492% increase in total area burned ○ 215% increase in length of the fire season • Changes in fire size, area burned, and fire frequency over the past several decades remain well below historical tribally-influenced frequency and extent of burning in California (Stephens et al. 2007) • No significant trends in the average areal proportion of high-severity fire were documented in northwestern CA from 1984– 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> • State-wide, up to 77% increase in mean annual area burned and 50% increase in the frequency of extremely large fires (>10,000 ha) by 2100 (Westerling 2018) <ul style="list-style-type: none"> ○ Greatest increases in burned area (up to 400%) occur in montane forested areas in northern California (Westerling 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) <ul style="list-style-type: none"> ○ However, human activity and fuel buildup from decades of fire suppression have altered historical fire-climate relationships (Taylor et al. 2016; Syphard et al. 2017; Wahl et al. 2019), and projections that incorporate these factors suggest that more

Regional Wildfire Trends	
<p>2008 (Parks et al. 2015; Law & Waring 2015; Keyser & Westerling 2017)</p> <ul style="list-style-type: none"> ○ The relatively short period of record for fire severity data may obscure long-term trends ○ To date, there are no peer-reviewed studies on trends in northern California fire severity that include data from the last ten years 	<p>significant increases in fire severity and size may occur (Mann et al. 2016; Wahl et al. 2019)</p> <ul style="list-style-type: none"> ● The majority of impacts to natural and human ecosystems come from extreme fire events (i.e., fires that have a low probability of occurring in any given place and time), which are likely to increase over the coming century (Westerling 2018) <ul style="list-style-type: none"> ○ Generally, these patterns are not well-represented in studies that evaluate indices of mean fire size, intensity/severity, etc.
Summary of Potential Impacts on Habitat <i>(see text for citations)</i>	
<ul style="list-style-type: none"> ● <i>Immediate:</i> <ul style="list-style-type: none"> ○ Reduces thatch, causing temporary increase in forb abundance and flowering ○ Temporarily reduces soil moisture and plant productivity ○ Reduces woody species encroachment ○ Enhances abundance and flowering of culturally important species (e.g., Indian potatoes) ● <i>Short-term (~2-year):</i> <ul style="list-style-type: none"> ○ Shifts in community composition; exact impacts vary at site level ○ Flowering increases in open grassland/forb habitat, providing pollinator forage ● <i>Long-term:</i> <ul style="list-style-type: none"> ○ Expansion of grasslands with type-conversion from shrublands or woodlands, which may increase culturally important grassland species (e.g., Indian potatoes) ○ Prevents woody species encroachment, maintaining existing open grassland area 	

Sensitivity and current exposure to non-climate stressors

Regional experts evaluated mixed grasslands as having moderate 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 mixed grasslands include invasive and other problematic species, fire suppression, livestock grazing, roads/highways/trails, and pollution.¹³

Invasive and other problematic species

Exotic species displace and outcompete native plant species, limiting establishment, recruitment, and growth of native vegetation by competing for limited resources (e.g., light, soil moisture, space; Menke 1992; Dyer & Rice 1999; Corbin & D’Antonio 2004; Molinari &

¹³ 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 land use conversion to residential/commercial development and cannabis (*Cannabis sativa* or *C. indica*) cultivation. Development and land-use conversion contribute to mixed grassland loss and fragmentation and can alter critical ecological processes (e.g., hydrological flow patterns; Vuln. Assessment Reviewers, pers. comm., 2018).

D'Antonio 2014). Invasive species also alter grassland physical structure (Molinari and D'Antonio 2014), alter fire return intervals and fire intensity and spread (Dukes & Shaw 2007; Livingston & Varner 2016), alter habitat quality and food availability for higher trophic levels (e.g., insects; Suttle et al. 2007), and degrade habitat suitability for wildlife (Ford & Hayes 2007). Increasing dominance of exotic vegetation has contributed to many rare or endangered species listings for native grassland taxa across the state (Ford & Hayes 2007).

Inland grasslands, coastal prairie grasslands, and serpentine grasslands have experienced significant invasion by and dominance of exotic species since the mid 1880s, including annual and perennial grasses and forbs (Bartolome et al. 2007; Ford & Hayes 2007; Harrison & Viers 2007). Mediterranean herbaceous species are the most common invaders (Keeley et al. 2011), and frequent invasive genera include *Avena*, *Bromus*, and *Erodium*, among others (Bartolome et al. 2007; Ford & Hayes 2007; Harrison & Viers 2007; Keeler-Wolf et al. 2007).

Interior grasslands are especially vulnerable to invasion dynamics due to dry, hot summers, which limit plant productivity (Corbin & D'Antonio 2004). As a result, a majority of interior grasslands on non-serpentine soils are dominated by exotic annual species (Eviner 2016). Comparatively, coastal prairie grasslands have experienced a slightly lower extent of invasion, and there is some evidence that in the absence of disturbance, native perennial grasses in coastal prairie grasslands can limit germination and productivity of exotic annual species and maintain dominance due to the more mild conditions associated with a maritime climate (Corbin & D'Antonio 2004). Additionally, serpentine grasslands are more resistant to invasion than non-serpentine grasslands due to nutrient-poor soils; invasion does still occur, particularly in areas experiencing nitrogen deposition or other forms of nutrient enrichment (Harrison & Viers 2007; Eskelinen & Harrison 2013; Eviner 2016). With the continued introduction of new exotic species as a result of human activity, as well as the evolution of serpentine-adapted ecotypes of existing exotic species, serpentine grasslands will likely face increasing exotic pressure in the future (Harrison & Viers 2007).

Invasive exotic species may be better than native species at responding to changes in climate via phenotypic plasticity and/or rapid genetic changes. For example, two invasive grasses (*A. barbata* and *B. madritensis*) were found to flower earlier, in addition to other changes, in response to decreasing precipitation in southern California (Nguyen et al. 2016). Additional disturbances, shifts in climatic conditions, and nitrogen deposition may all lead to substantial changes in exotic plant abundance in the future (Dukes & Shaw 2007; Keeler-Wolf et al. 2007; Eskelinen & Harrison 2013), affecting competitive interactions with native species, the overall persistence and quality of grassland systems, and overall grassland response to climate change (Dukes and Shaw 2007).

Fire suppression¹⁴

Grassland fire regimes have significantly changed over time as human occupation and use of the landscape has changed. For example, fire frequency likely increased following Native American settlement, and then declined with European-American settlement and the rise of agriculture and grazing (D’Antonio et al. 2006; Keeley et al. 2011; Cornwell et al. 2012). Fire suppression management objectives beginning in the 20th century have further reduced fire frequencies across many habitat types, including mixed grasslands (Skinner et al. 2006, 2009; Stuart & Stephens 2006; Cornwell et al. 2012; Steel et al. 2015).

The absence of fire can alter grassland community composition and structure through several mechanisms (see Table 1). Fire exclusion results in the loss of important soil nutrients provided by fire (Anderson 2005) and can limit germination and/or seed production of species that require fire-related heat or chemical cues (Tyler & Borchert 2003; D’Antonio et al. 2006). For example, many legumes require heat to break seed dormancy (D’Antonio et al. 2006), and many geophytes (e.g., *Brodiaea* spp.) flower and release abundant seeds following exposure to fire (Tyler & Borchert 2003). A lack of fire also promotes thatch buildup, which can limit native forb, geophyte, and legume germination through a reduction of light and soil temperature (D’Antonio et al. 2006; Norgaard et al. 2016), ultimately providing a competitive advantage to exotic species (Eviner 2016).

Additionally, the absence of disturbances such as fire allows encroachment of woody species in coastal prairie grasslands (Ford & Hayes 2007; Stromberg et al. 2007; BLM 2008, 2013). Coastal prairie grasslands in the study region have lost roughly half of their distribution in the last 50 years due to natural succession in the absence of disturbance (Vuln. Assessment Reviewers, pers. comm., 2018). Without disturbance, coastal prairies can be replaced by shrub species in 15–25 years (Ford & Hayes 2007). Coastal prairies on fertile soils are more dependent on fire disturbance for habitat persistence than interior arid grasslands and grasslands on infertile soils (Cornwell et al. 2012; Eviner 2016). In these latter habitats, aridity and soil properties, respectively, play the most critical role in limiting woody vegetation establishment (Vuln. Assessment Reviewers, pers. comm., 2018). In general, woody species encroachment in the absence of disturbance reduces habitat extent and undermines the productivity of culturally important grassland plant species (e.g., Indian potatoes; Underwood et al. 2003; Anderson 2005; Lake 2007; Norgaard et al. 2016; Karuk Tribe 2019). Patches of geophytes and other species valued by tribes can also be degraded by fire line construction and/or post-fire treatments that impact establishment (Norgaard et al. 2016).

¹⁴ The authors recognize that the majority of fires in California grasslands are human-caused (i.e., historically by Native American cultural burning, and currently through prescribed fire management or unintentional human ignitions). In this assessment, we use “fire suppression” to generally discuss the impacts of reduced fire frequency in grassland landscapes.

Table 1. Impacts of fire suppression on the resilience of mixed grasslands to climate stressors and climate-driven changes in fire regimes (table adapted from Norgaard et al. 2016).

Prior to Fire	During Fire	After Fire
<ul style="list-style-type: none"> • Woody species encroachment of prairies • Loss of culturally-valued species such as Indian potatoes • Loss of important soil nutrients provided by fire 	<ul style="list-style-type: none"> • Reduced productivity in geophytes and other native species in areas with heavy thatch buildup • Degradation of geophyte habitat patches by fire line construction 	<ul style="list-style-type: none"> • Reduced establishment of geophytes and other culturally-valued species due to post-fire treatments
Source(s): Underwood et al. 2003; Anderson 2005; D’Antonio et al. 2006; Ford & Hayes 2007; Norgaard et al. 2016		

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Livestock grazing

Livestock grazing can have a variable influence on grassland communities depending on timing, frequency, duration, and intensity, as well as the characteristics (e.g., soil structure) of the habitat in question (Bartolome et al. 2014; Eviner 2016). The broad diversity of grassland responses to grazing underscores the importance of maintaining a diverse spatial and temporal grazing matrix (Hayes & Holl 2003), conducting localized analyses of herbivory impacts and trends (Callaway & Davis 1993; Jackson & Bartolome 2002; Bartolome et al. 2014), and developing species- and site-specific management objectives to achieve various management goals (Bartolome et al. 2007; Jackson & Bartolome 2007; Stahlheber & D’Antonio 2013).

Livestock grazing can potentially benefit grasslands. For example, grazing can mitigate coastal prairie loss to shrub and forest encroachment (Ford & Hayes 2007) and help mitigate the high biomass and litter accumulation associated with exotic grass cover (Hayes & Holl 2003 and citations therein; Molinari & D’Antonio 2014), which benefits both native plant species and wildlife (Stahlheber & D’Antonio 2013; Bartolome et al. 2014). Grazing also helps distribute seeds and nutrients across the grassland system, as well as incorporate seeds into the soil (although this may exacerbate exotic species issues; Vuln. Assessment Reviewers, pers. comm., 2018). It should also be noted that wildlife and insect herbivory influence grassland productivity and diversity (Hayes & Holl 2003; Bartolome et al. 2014).

Livestock grazing can also harm grasslands, including causing unintentional increases in exotic species (Underwood et al. 2003; Eviner 2016; Livingston & Varner 2016). A meta-analysis of grazing in California grasslands found that, on average, grazing tends to increase non-native forb cover and non-native grass species richness, while simultaneously elevating native forb species richness and native grass cover (Stahlheber & D’Antonio 2013). Additionally, cattle trampling may compact soil, reducing infiltration and increasing bulk density and runoff (Daniel et al. 2002 cited in Jackson and Bartolome 2007), although compaction tends to be transient, isolated, and seasonally reversed with new root growth (Vuln. Assessment Reviewers, pers. comm., 2018). Livestock travel corridors can also destabilize slopes and remove vegetation that

would stabilize soils along streams, contributing to stream downcutting and headcutting (Belsky et al. 1999).

Roads, highways, and trails

Road, highway, and trail construction destroys and fragments grasslands, and increases opportunities for invasive species to spread (Bartolome et al. 2007). Invasive plant seeds may be carried on vehicles, and invasive generalist species (e.g., exotic grasses) may thrive in roadside environments where light and water runoff is abundant (Coffin 2007). Additionally, road and trail maintenance activities act as a disturbance mechanism, providing invasive colonization opportunities (Stromberg et al. 2007). Roads and highways also increase the likelihood of human wildfire ignitions, and vehicular traffic exhaust may increase nitrogen deposition (Vuln. Assessment Workshop, pers. comm., 2017).

Pollution

Nitrogen deposition reduces native plant diversity (particularly forbs) and increases exotic grass dominance and productivity (Dukes et al. 2005; Dukes & Shaw 2007; Eviner 2016). Nitrogen deposition may also exacerbate the impacts of changing precipitation regimes by enhancing exotic species establishment in both non-serpentine and serpentine areas (Eskelinen & Harrison 2013). Thirty percent of California's grasslands experience nitrogen deposition rates of 5 kg N per hectare per year; deposition rates are higher in some areas of northern California, as much as 16 kg N per hectare per year (Weiss 2006; Dukes & Shaw 2007).

Sensitivity to other critical factors

Regional experts evaluated mixed grasslands as having moderate-high sensitivity to decoupling of plant phenologies with pollinator migration or life cycles (moderate confidence in evaluation).

Although many grassland species are wind- or self-pollinated, some grassland species, particularly forbs, rely on pollinators (Eviner 2016). Shifting plant phenologies as a result of temperature, precipitation, and soil moisture changes creates the opportunity for mismatches with pollinator life histories or migrations, with potential impacts on grassland plant genetic diversity as well as the maintenance of pollinator populations (Eviner 2016).

Adaptive Capacity

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

Habitat extent, integrity, continuity, and permeability

Regional experts evaluated mixed grasslands as having a high geographic extent (high confidence in evaluation), moderate structural and functional integrity (moderate confidence in evaluation), and variable continuity (moderate confidence in evaluation). Native grasslands

were evaluated as having significantly lower habitat continuity than non-native grasslands (Vuln. Assessment Workshop, pers. comm., 2017).

Landscape permeability for mixed grasslands was evaluated as moderate (high confidence). Geologic features (e.g., soil type), roads/highways/trails, agriculture (e.g., grazing, cropland), land-use conversion (e.g., to vineyards), and invasive and problematic species were identified as the primary barriers to habitat continuity and dispersal across the study region.¹⁵ In general, non-native-dominated grasslands have higher landscape permeability and are less affected by the above barriers than native grasslands (Vuln. Assessment Workshop, pers. comm., 2017).

Coastal prairie grasslands have a wide distribution in California, extending from Santa Barbara County to the Oregon border in areas with maritime influence (Ford & Hayes 2007). Coastal prairie has experienced loss and fragmentation as a result of human land uses (Ford & Hayes 2007), as well as woody vegetation encroachment (BLM 2008, 2013). Additionally, coastal prairie composition has been significantly altered over time, and many sites are now dominated by exotic species (Ford & Hayes 2007).

Interior grasslands also have a wide distribution in California. In northern California, they occupy low elevation areas between the Cascade-Sierra Crest and the northern Coast Range, as well as some sites within the Coast Range. Interior grasslands extend south, through the Central Valley, to Baja California, Mexico, with more coastward distributions in central and southern California (Bartolome et al. 2007). Interior grasslands are largely dominated by exotic species, now naturalized (Bartolome et al. 2007). Active management is required to keep non-native annual grasses from completely excluding native species (Bartolome et al. 2007).

Serpentine grasslands also have a relatively wide distribution in California, but do not occupy significant land area (Harrison & Viers 2007). For example, annual grasslands only cover 7% (417 km²) of all serpentine soils in California (Harrison & Viers 2007). In northern California, serpentine grasslands occur primarily in the Northern California Coast Range and the Northern California Interior Coast Range, with lower abundance in the Klamath Mountains and the northernmost Sierra Nevada foothills (Harrison & Viers 2007). They are not common along the North Coast and do not occur in the Central Valley (Harrison & Viers 2007). In general, serpentine grassland distribution, as well as distribution of other special grassland types (e.g., alkali sinks, vernal pools), is patchy and localized, which will undermine migration opportunities in the face of climate change (Damschen et al. 2012). Serpentine habitat integrity is threatened by the existing presence of exotic species, and increasingly by habitat fragmentation and modification as a result of human land-use conversion and associated activities (Harrison & Viers 2007 and citations therein). For example, habitat fragmentation may reduce serpentine resilience to invasion by decreasing patch size (Harrison & Viers 2007 and citations therein).

¹⁵ All barriers presented were ranked as having a moderate or higher impact on this habitat type.

Habitat diversity

Regional experts evaluated mixed grasslands as having high physical and topographical diversity (high confidence in evaluation), high component species diversity (high confidence), and moderate functional diversity (high confidence). In general, non-native grasslands were evaluated to be less diverse than native grasslands (Vuln. Assessment workshop, pers. comm., 2017).

Grasslands have high plant diversity across their distribution as result of large precipitation and temperature gradients, different soil types, and topography (Bartolome et al. 2014; Spiegel et al. 2014), but also feature high temporal and spatial diversity within a given site (Eviner 2016). Functional group diversity is also relatively high, with grasses and forbs alternating in dominance according to precipitation and temperature conditions (Eviner 2016). In general, annual forbs and grasses will be most sensitive to climate impacts due their sensitivity to precipitation and temperature cues (Vuln. Assessment Workshop, pers. comm., 2017).

California's coastal prairie grasslands are quite biodiverse, potentially having highest plant diversity of North American grasslands (Stromberg et al. 2002). Species richness is highest in moister maritime sites (Ford & Hayes 2007). Across its geographic range, coastal prairie grassland has both native perennial species and over 250 forb species, 33% of which are native annual forbs (Stromberg et al. 2002). Additionally, coastal prairie hosts nearly 80 endemic species, and many rare and endangered species (Ford & Hayes 2007). There are three vegetation types of coastal prairie (Ford & Hayes 2007):

- *California oatgrass prairie*: California oatgrass communities are found on moist coastal terraces, and typically feature high abundance of native perennial species. Three sub-types exist: one dominated by California brome and purple needlegrass, a second dominated by tufted hairgrass and coyote brush (*Baccharis pilularis*) and a third defined by hairy oatgrass (*Danthonia pilosa*), various *Carex* species, or exotic perennial species such as Kentucky bluegrass (*Poa pratensis*), common velvet grass, sweet vernal grass (*Anthoxanthum odoratum*), tall fescue (*F. arundinacea*), or Harding grass.
- *Moist native perennial grassland*: occurs on the wettest coastal sites and is dominated by native meadow barley, brownheaded rush (*Juncus phaeocephalus*), various *Carex* species, and exotic curly dock (*Rumex crispus*) and hyssop loosestrife (*Lythrum hyssopifolia*).
- *California annual grassland*: typically has low abundance of native perennial species and a mixture of mostly exotic annual plants, including ripgut brome, wild oat, and several forbs including Italian plumeless thistle (*Carduus pycnocephala*), milk thistle (*Silybum marianum*), and short pod mustard (*Hirschfeldia incana*).

Interior grasslands are also biodiverse, although non-native species comprise a significant portion of species richness. Dominant species are all non-native, and include soft brome (*B. hordeaceus*), *Vulpia* species, ripgut brome, broadleaf filaree (wetter areas), redstem filaree (drier areas), and foxtail brome (dry sites). Local dominants may also include slender wild oat (*A. barbata*), wild oat, and Italian rye grass (*Lolium multiflorum*). Native annual *Trifolium* species

are also common in more mesic sites (Bartolome et al. 2007). In the Central Valley, interior grasslands are often interspersed with vernal pools, which support extremely rare plant assemblages (Bartolome et al. 2007). Vernal pool grassland species (e.g., *Orcuttia* spp.) may be particularly sensitive to climate changes (Vuln. Assessment Workshop, pers. comm., 2017).¹⁶

Species composition of serpentine grasslands is controlled by unique, and often harsh, edaphic (soil) conditions, contributing to high endemism and many serpentine-adapted ecotypes of common grassland species (Harrison & Viers 2007; Eviner 2016). For example, common serpentine endemics in northern California include Tracy's clarkia (*Clarkia gracilis* ssp. *tracyi*), Jepson's navarretia (*Navarretia jepsonii*), and small-flowered calycadenia (*Calycadenia pauciflora*; Fernandez-Going & Harrison 2013). Along with alkali sinks, vernal pools, and mesic refugia areas (e.g., swales, oak understory), these unique, low productivity grassland communities are essentially the only intact remnants of native-dominated grasslands in the state, and collectively the most important repository of native grassland species diversity in California (Vuln. Assessment Reviewers, pers. comm., 2018).

Resistance and recovery

Regional experts evaluated mixed grasslands as having moderate resistance to climate stressors and natural disturbance regimes (high confidence in evaluation). Recovery potential was evaluated as high (high confidence), although rate of recovery depends on the severity of the stressor. In general, non-native grasslands were evaluated to have higher resistance and recovery potential than native grasslands (Vuln. Assessment Workshop, pers. comm., 2017).

Perennial species

Established native perennial grasses are fairly resistant to seasonal drought (Corbin et al. 2005), precipitation fluctuations, and other climatic variation (Eviner 2014). However, native perennials are patchily distributed within a given site (Ford & Hayes 2007), have small seedbanks (LaForgia et al. 2018), low dispersal (Vuln. Assessment Reviewers, pers. comm., 2018), and low recruitment (Pfeifer-Meister et al. 2016). Their seedlings are less resistant to impacts than established plants (Pfeifer-Meister et al. 2016). Slow development and small stature during the first year of growth also make perennial grasses susceptible to invasion and exclusion by exotic annuals (Bartolome et al. 2007), particularly in the presence of disturbance (Corbin & D'Antonio 2004). As a result, perennial species are difficult to establish in areas dominated by annual species (Eviner 2014), and have not naturally recovered dominance in annual-invaded areas (Corbin & D'Antonio 2004).

Annual species

Annual seedbanks can persist for long periods, waiting for optimal environmental conditions (whether climate- or disturbance-related) for germination (Bartolome et al. 2014). These and other annual seedbank dynamics, such as the high abundance of annual species across grassland landscapes and high dispersal capacity (Vuln. Assessment Reviewers, pers. comm., 2018), enhance the ability of annual species to recover after disturbance or in response to

¹⁶ Vernal pools are addressed separately in their own assessment.

climatic variability (Bartolome et al. 2014; Eviner 2014). However, seed longevity varies somewhat between annual functional groups, with native annual forbs exhibiting prolonged seed dormancy relative to exotic annual grasses (LaForgia et al. 2018). Native annual forbs remain dormant until periods of low competition, capitalizing on poor-growth periods (e.g., droughts) experienced by perennial (Eviner 2014) and exotic grasses (LaForgia et al. 2018).

Serpentine grasslands

Serpentine grasslands are more resilient to changing climatic conditions and exotic invasion due to harsher growing conditions, which contribute to more specialized flora (Eskelinen & Harrison 2013). However, component species have low abundance and are often not well-dispersing (Vuln. Assessment Reviewers, pers. comm., 2018), and habitat areas with suitable substrate are small and spatially isolated (Damschen et al. 2012). These constraints can limit the ability of serpentine species to adapt to and track climate changes (Damschen et al. 2012).

Management potential

Public and societal value

Regional experts evaluated mixed grasslands as having low-moderate public and societal value (high confidence in evaluation).

Grasslands are valued for grazing, open space, recreation (e.g., hiking equestrian use), aesthetic views, wildflower displays, wildlife (e.g., deer, elk), hunting, and erosion control (Vuln. Assessment Workshop Participants, pers. comm., 2017). There is societal support for habitat management, including grassland restoration projects, from native grassland and native plant groups, hunting groups, and from the California Mule Deer Foundation, which supports protecting habitat for trophy mule deer, and the Rocky Mountain Elk Foundation, which has donated funds to improve habitat for future elk recolonization efforts (Vuln. Assessment Workshop, pers. comm., 2017).

Management capacity and ability to alleviate impacts¹⁷

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

Some coastal prairie areas are protected by local ordinances (Ford & Hayes 2007). Similarly, some interior grassland area has been protected in order to preserve vernal pools, but the majority is under private ownership (Bartolome et al. 2007). Conservation easements are in place in some areas to facilitate grassland management (Bartolome et al. 2007), but the high extent of private ownership means that the efficacy of conservation and restoration activities

¹⁷ Further information on climate adaptation strategies and actions for northern California can be found on the project page (<https://bit.ly/31AUGs5>).

will be highly influenced by individual land owner values and management practices (Eviner 2016).

High site variability makes management generalizations for grassland systems difficult and requires tailoring of management techniques to meet local goals and needs (Bartolome et al. 2014; Eviner 2014). A variety of management actions, including prescribed burning and livestock grazing, are being utilized and researched to enhance mixed grasslands and associated culturally important species in northern California (see Table 2; Bartolome et al. 2007; Ford & Hayes 2007; Eviner 2016; Norgaard et al. 2016; Livingston & Varner 2016; Karuk Tribe 2019). For enhancing native species, these tools are typically most successful in areas where a substantial component of native species remain (e.g., low productivity sites, swales, oak understory; Vuln. Assessment Workshop, pers. comm., 2018). For example, management targeted at enhancing existing native species populations in coastal prairie grasslands have been somewhat successful (Ford & Hayes 2007), as have native plug planting efforts in the North Coast Range (Vuln. Assessment Reviewers, pers. comm., 2018). Comparatively, restoration efforts in degraded grassland areas with low native species abundance have been less successful (e.g., efforts fail to create self-sustaining populations, or resulting populations lack species richness of intact systems; Ford & Hayes 2007).

Table 2. Effects of prescribed fire on mixed grasslands across time (table adapted from Norgaard et al. 2016). Cultural burning practices, in particular, have played a role in maintaining these habitats on the landscape over very long time scales.

Immediate	2-Year	Long Term
<ul style="list-style-type: none"> • Reduces competition from grasses and shrubs • Enhances geophyte bulblet production 	<ul style="list-style-type: none"> • Releases soil nutrients to enrich grassland soils, benefitting native species 	<ul style="list-style-type: none"> • Controls woody species encroachment, maintaining coastal prairies over long time scales
Source(s): Anderson 1997, 2005; Karuk Tribe 2019		

An emerging threat to remnant native grasslands is the proposal to add compost to rangelands to increase carbon sequestration. Enhancing soil fertility via compost addition is a potential maladaptation because it is likely to enhance exotic grass productivity, likely further undermining native diversity, which is currently highest on low-nutrient sites (Vuln. Assessment Reviewers, pers. comm., 2018).

Ecosystem services

Grasslands provide a variety of ecosystem services, including:

- Provisioning of biodiversity, food, fiber, fuel, genetic resources, natural medicines, ornamental resources, and fresh water;
- Regulation of air quality, climate, carbon storage, flood/erosion control, water purification, and pollination;

- Support of primary production, soil formation/retention, nutrient cycling, and water cycling; and
- Cultural and tribal uses for spiritual/religious purposes, knowledge systems, educational values, aesthetic values, social relations, sense of place, cultural heritage, inspiration, and recreation (Vuln. Assessment Workshop, pers. comm., 2017).

Mixed grassland ecosystems provide a variety of ecosystem services. Grasslands play a significant role in maintaining regional biodiversity, and support numerous endemic, rare, threatened, and endangered species (Eviner 2016). Grasslands are valued for their scenic qualities (e.g., scenic vistas, wildflower displays) and recreational opportunities (Eviner 2016). High infiltration rates in grassland systems help minimize flood risk and improve water quality through pollutant filtration (e.g., pathogens, nutrients, sediment; Eviner 2016). Grassland areas also provide significant livestock forage (although see above issues related to grazing and exotic species), and support many pollinators critical for California agriculture (Eviner 2016). Additionally, mixed grassland systems provide carbon storage and support terrestrial food chains via high primary productivity (Eviner 2016). Grassland areas are also highly valued by regional Native American tribes for food and medicine provisioning (Anderson 2005; Eviner 2016; Norgaard et al. 2016). While grasslands provide many ecosystem services, managing for simultaneous objectives can be challenging; for example, maintaining native biodiversity can conflict with management objectives aimed at enhancing carbon storage and/or enhancing provisioning of livestock forage (Stein et al. 2014; Eviner 2016).

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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),¹⁸ 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.¹⁹

¹⁸ Sensitivity and adaptive capacity elements were informed by Lawler 2010, Glick et al. 2011, and Manomet Center for Conservation Sciences 2012.

¹⁹ 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:

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

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

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

Vulnerability Assessment Model Elements

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

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

Sensitivity & Exposure (Applies to Species Groups and Species)

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

Sensitivity & Exposure (Applies to Species ONLY)

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

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

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

Adaptive Capacity (Applies to Habitats ONLY)

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

Adaptive Capacity (Applies to Species Groups, Species)

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

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