Native Insect Pollinators

Northern California Climate Change Vulnerability Assessment Synthesis

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

Peer reviewers for this document included Anonymous (University of California, Davis), Anonymous (U.S. Fish and Wildlife Service), Laurel Goldsmith (U.S. Fish and Wildlife Service), Jesse Irwin (Bureau of Land Management), and Jennifer Wheeler (Bureau of Land Management). Vulnerability scores were provided by Laurel Goldsmith. Upper Lake workshop participants provided additional comments on the climate change vulnerability of this species group.

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Species Group Description

This assessment focuses on native insect pollinators, including bees, butterflies, beetles, flies, wasps, ants, and moths.[1] Native insect pollinators in northern California are incredibly diverse,

[1] Birds and bats are not considered in this assessment, although they also function as pollinators.

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spanning many taxa, and encompassing diverse life histories (National Research Council 2007). For example, they may be resident or migratory, forage generalists or specialists, and nest below, on, or above the ground surface (National Research Council 2007). Native insect pollinators play a critical role in the reproductive fitness of native plant communities, which in turn support diverse wildlife communities and first foods utilized by regional Native American tribes (Black et al. 2009 and citations therein).

Executive Summary

The relative vulnerability of native insect pollinators 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.

Native Insect Pollinators	**Rank**	**Confidence**
Sensitivity	Moderate-High	Low
Future Exposure	Moderate	Moderate
Adaptive Capacity	Moderate	Low
Vulnerability	**Moderate**	**Low**

Sensitivity & Exposure Summary	
	Climate and climate-driven factors: • Air temperature, heat waves, drought, soil moisture, precipitation amount and timing, storms and wind *Disturbance regimes*: • Flooding, disease, wildfire *Non-climate stressors*: • Fire suppression, residential and commercial development, poisons, invasive and problematic species, agriculture, livestock grazing

Native insect pollinators exhibit both direct and indirect sensitivities to climate and climate-driven stressors and disturbance regimes, such as rising temperatures, heat waves, precipitation, storms, and wind. These stressors can alter foraging activity, recruitment success, development rates, and geographic distribution. Native pollinators are also vulnerable to direct mortality from a variety of factors, including fire, flooding, heat waves, disease, pesticides, and livestock grazing. Many climate factors, disturbance regimes, and non-climate stressors can also indirectly influence pollinator abundance, recruitment, and species richness by affecting habitat availability, pollen and nectar resources, and/or nest sites. For example, pollinator habitat can be degraded or lost to wildfire, fire suppression, urban development, and agriculture, and remnant habitat can be further degraded by drought, herbicide use, invasions of non-native pollinators and plants (particularly invasive weeds), and livestock grazing. Cumulatively, habitat loss and degradation results in reduced pollinator abundance, fitness, recruitment, and species richness. Habitat loss and degradation can interact with climatic changes to further restrict pollinator distribution and persistence.

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Adaptive Capacity Summary	
	Factors that enhance adaptive capacity: + Widespread and occupy many different habitat types + Diverse life histories among taxa buffer complete species group losses + Short generation times promote rapid responses to changing conditions + Provide significant pollination services, increasing societal support for management *Factors that undermine adaptive capacity*: – Declining abundance and species richness in response to multiple stressors – Small and/or isolated populations more vulnerable to extirpation and have lower capacity for genetic adaptation to change – Certain life stages (e.g., eggs, larvae) more vulnerable to mortality

While widely distributed across a variety of habitat types in northern California, native pollinator population numbers and species richness are declining in response to multiple stressors, including climate change, land-use conversion, disease, and detrimental human activities. Further type-conversion within native ecosystems will likely interact with climate change to limit dispersal and migration opportunities for insect pollinators. Small and/or isolated pollinator populations are more vulnerable to extirpation from extreme events, and usually have lower potential for genetic adaptation to rapid climate change. However, native pollinators include a variety of taxa with diverse life histories, and some species may exhibit behavioral plasticity in response to changing conditions. Short generation times for larger populations also provide the opportunity for rapid adaptation, which may help buffer this species group from complete losses as climate changes. Several government agencies and advocacy groups support pollinator management and conservation, with efforts focused on protecting federally or state-listed endangered or threatened species as well as maintaining critical pollination services for regional native vegetation and agriculture.

Sensitivity and Exposure

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

Because insect pollinators are ectotherms (i.e., their body temperature is regulated by ambient air temperatures), native pollinator ranges will be directly influenced by air temperature (Deutsch et al. 2008; Sunday et al. 2012). Range shifts driven by rising temperatures have already been documented in several insect species regionally and world-wide (Karban & Strauss 2004; Forister et al. 2010; Kerr et al. 2015). Climate change will also likely shift native insect pollinator distributions by altering vegetation distribution on the landscape (Kerr et al. 2015). Many native habitats that support abundant pollinators (e.g., grasslands, chaparral, oak woodlands) are projected to experience significant changes in landscape distribution in northern California by the end of the century (Lenihan et al. 2008; Thorne et al. 2016).[2]

[2] See individual habitat vulnerability syntheses for more in-depth discussion of potential range shifts.

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However, pollinators will not necessarily be able to colonize new areas as the flora shifts. New colonization areas may be unavailable or inaccessible by factors such as elevation limitations (Mac Nally et al. 2003; Forister et al. 2010; Buckley & Kingsolver 2012) or development that fragments habitat and creates prohibitively large dispersal distances (Casner et al. 2014). The presence of local microrefugia (e.g., due to diverse topography) may allow local pollinator persistence in otherwise unsuitable landscapes, potentially buffering larger-scale habitat shifts (Mac Nally et al. 2003; Schweiger et al. 2008).

Potential Changes in Species Group Distribution
• Highly variable depending on taxa, but distribution changes likely due to physiological impacts of climate change and altered vegetation distribution • Some species may not be able to colonize new areas (e.g., due to elevation restrictions or other geographical barriers) • Microrefugia may buffer larger distribution shifts and/or prevent local extirpation
Source(s): Mac Nally et al. 2003; Schweiger et al. 2008; Sunday et al. 2012; Kerr et al. 2015

The direction and magnitude of distributional shifts are likely to vary by taxa (Sunday et al. 2012). For example, non-migratory European butterfly species have exhibited distinct northward range shifts and overall range expansions as a result of warming temperatures (Parmesan et al. 1999). Comparatively, bumble bees (*Bombus* spp.) across Europe and North America exhibited range contractions from 1975–2010 due to range losses at their southern limits and failure to expand at their northern limits (Kerr et al. 2015). Simultaneously, bumble bee species in the warmest locations in North America have exhibited shifts toward higher elevations as temperatures increase, while species from cooler areas have shifted to lower elevations (Kerr et al. 2015). In general, high elevation habitat changes as a result of climate change and human activities could contribute to a loss of montane generalist bumble bees, particularly if these species do not expand northward (Kerr et al. 2015).

Sensitivity and future exposure to climate and climate-driven factors

Regional experts evaluated native insect pollinators as having high sensitivity to climate and climate-driven factors (moderate confidence in evaluation), with an overall moderate-high future exposure to these factors within the study region (moderate confidence). Key climatic factors that affect native insect pollinators include air temperature, heat waves, drought, soil moisture, precipitation amount and timing, storms, and wind.[3]

<u>Air temperature</u>

Most native insect pollinators are ectotherms, and increasing air temperatures will directly affect their behavior, fecundity, development, and phenology. For example, warmer temperatures will likely alter insect activity levels, thermoregulatory behavior, flight behavior, and nesting site selection (Cane & Neff 2011; Buckley & Kingsolver 2012; Kellermann et al. 2012). In topographically diverse northern California, impacts to pollinators from rising air temperatures will likely vary by elevation (Buckley & Kingsolver 2012). For example, butterfly

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

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flight is restricted by body temperature, so warmer temperatures may increase the frequency or duration of suitable flight periods at historically cooler, higher elevations, while decreasing flight opportunities (by requiring thermoregulatory behavior) at warmer, lower elevations (Buckley & Kingsolver 2012). However, the broad array of microclimates present in the northern California study region (e.g., due to coastal fog, cool air drainages) will likely contribute to very nuanced changes in thermal patterns depending on location, moderating future changes in pollinator behavior (Vuln. Assessment Reviewers, pers. comm., 2018).

Increasing air temperatures will also affect pollinator reproductive capacity and larval development. For example, higher temperatures reduce red flour beetle (*Tribolium castaneum*) fecundity by reducing sperm fertility (Sales et al. 2018) and alter the rate of egg production, deposition (Forister & Shapiro 2003; Stevens & Frey 2010), and viability in some butterfly species (Buckley & Kingsolver 2012). However, heat stress may limit butterfly egg viability at lower elevations (Buckley & Kingsolver 2012). Warmer temperatures are also likely to accelerate larval development rates (Forister & Shapiro 2003; Stevens & Frey 2010), which can lead to earlier emergence, driving shifts in pollinator phenology (Forister & Shapiro 2003). For example, several butterfly species at low elevations in the Central Valley have advanced the timing of first spring flight days by up to 24 days over 31 years in response to increasing winter maximum temperatures and drier winter conditions (Forister & Shapiro 2003). The most significant changes occurred in species that historically emerged earliest in the spring, indicating that these may be more sensitive to changes in seasonal conditions (Forister & Shapiro 2003). Earlier emergence was also more common for species that overwintered as pupae rather than larvae (Forister & Shapiro 2003).

Shifting pollinator phenology as a result of warmer air temperatures creates the potential for phenological mismatch between pollinator flight activity (Forister & Shapiro 2003) and plant flower production (Dukes & Shaw 2007; Inouye 2008; Rundel & Millar 2016). Not all species shift phenologies at the same rate or even the same direction in response to the same climate signals (Memmott et al. 2007). Mismatches may disrupt mutualistic plant-pollinator relationships to the detriment of both groups (Memmott et al. 2007). For example, specialist pollinators that utilize only a few plant species for foraging may experience partial or complete absence of food resources during their flight period, while generalists are more likely to experience a reduction in food source diversity and greater temporal and spatial variation in food supply (Memmott et al. 2007). Reduced food availability is likely to limit pollinator fecundity and survival, which may ultimately reduce pollinator population sizes and/or drive pollinator extinction (Memmott et al. 2007).

Changes in pollinator activity, reproductive capacity, development, and interactions with plant hosts as a result of warmer temperatures are likely to affect overall pollinator population abundance and species richness, although effects may vary within and across years (Casner et al. 2014). For example, butterfly species richness in the Central Valley has been found to be lower when the prior year had high summer minimum and fall maximum temperatures and low rainfall (Casner et al. 2014). Conversely, high minimum spring temperatures increase within-year butterfly species richness, both by accelerating larval development and creating favorable

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early season growing conditions for plant resources (Casner et al. 2014). Increasing pollinator activity, fecundity, abundance, and/or species diversity in areas where insects have previously been temperature-limited may also contribute to range expansions at high elevations and more northern latitudes (Mac Nally et al. 2003; Forister et al. 2010). Alternatively, some native pollinators may be adversely affected by warmer temperatures that exceed their tolerance thresholds (Forister et al. 2010).

Responses to increasing temperatures are likely to vary by taxa and by individual species phenotype. For example, bumble bees (*Bombus* spp.) have larger body sizes relative to most insects and are covered in chitin "hairs", which historically have allowed occupancy and activity in temperate climates, including mountainous areas (Goulson 2010). However, they are largely absent from warmer areas of the world (Goulson 2010), suggesting that warmer temperatures may force range shifts to higher elevations or more northerly latitudes and/or may cause local range contractions (as has already been observed in some areas; Kerr et al. 2015). Additionally, species with cold-adapted phenotypes may be more likely to experience behavioral and reproductive impacts as adaptations that previously allowed them to thrive make them more susceptible to heat stress with rising ambient temperatures (Buckley & Kingsolver 2012).

Regional Air Temperature Trends[4]	

Historical & current trends:
• 0.03°C (0.05°F) decrease to 0.5°C (0.9°F) increase in the average annual temperature between 1900 and 2009 for the Northwestern California, Southern Cascade, and Great Valley ecoregions (Rapacciuolo et al. 2014)
  ○ No seasonal temperature trends are available
	Projected future trends:
	• 2.2–6.1°C (4.0–11.0°F) increase in the average annual temperature 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)
	  ○ 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 Species Group *(see text for citations)*	
• Altered activity levels, thermoregulatory behavior, flight and foraging behavior, reproductive capacity, and nesting site selection, although exact impacts will vary by elevation and degree of current stress experienced in a given location • Phenological shifts due to accelerated larval development, which creates potential for mismatches with plant phenology that can ultimately reduce survival and recruitment	

[4] Trends in climate factors and natural disturbance regimes presented in this and subsequent summary tables are not species group-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.

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Regional Air Temperature Trends[4]
• Potential changes in abundance and species richness due to changes in activity, reproduction, development, and interaction with plant resources • Variable range shifts and expansion/contraction depending on species life history and other factors • Increased vulnerability to heat stress, particularly for cold-adapted species

Heat waves

Heat waves may contribute to pollinator mortality by exceeding lethal temperature thresholds or via prolonged exposure to sub-lethal temperatures (Rasmont & Iserbyt 2012). Heat exposure can also reduce pollinator fecundity. For example, female alpine butterflies in the Rocky Mountains have been found to have shorter life spans and reduced egg production following exposure to extreme heat (Kingsolver & Watt 1983 cited in Buckley & Kingsolver 2012). Additionally, heat waves have been shown to reduce sperm production and viability in red flour beetles, and offspring from heat-exposed sperm were found to have lower reproductive potential and a reduced lifespan (Sales et al. 2018). Mortality and reduced fecundity as a result of heat waves can lead to low pollinator population numbers and/or local extirpations (Rasmont & Iserbyt 2012). Cold-adapted species (e.g., bumble bees) and those whose key foraging or reproductive phases overlap with heat wave periods (e.g., late summer) may be particularly vulnerable to heat-related impacts (Rasmont & Iserbyt 2012).

Heat waves can also impact pollinators by increasing energetic costs associated with thermoregulatory heat avoidance behaviors and/or by decreasing forage resource availability (Rasmont & Iserbyt 2012; Sunday et al. 2014). For example, heat waves may increase local pollinator reliance on and utilization of shady or cool microhabitats (e.g., underground), reducing foraging opportunities and potentially impacting fitness or survival (Rasmont & Iserbyt 2012; Sunday et al. 2014).

Regional Heat Wave Trends	
Historical & current trends: • Increase in the frequency of humid nighttime events over the past several decades (Gershunov & Guirguis 2012) • High interannual and interdecadal variability in heat waves (Gershunov & Guirguis 2012)	*Projected future trends:* • Increased heat waves, with the greatest increase in humid nighttime heat waves and in coastal areas (Gershunov & Guirguis 2012) • 2–6°C (3.6–10.8°F) increase in the temperature of the hottest day of the year by 2100 (Pierce et al. 2018)
Summary of Potential Impacts on Species Group *(see text for citations)*	
• Enhanced mortality and reduced fecundity, leading to population declines and potential local extirpations • Reduced foraging opportunities due to increased need for thermoregulatory behaviors • Reduced forage resource availability	

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Drought and soil moisture

Drought and reductions in soil moisture will likely have negative impacts on many native insect pollinators through bottom-up controls on native plants, including declines in plant resource availability, quality, and diversity (Bell 1998; Carroll et al. 2001; Hatfield & LeBuhn 2007; Stevens & Frey 2010). The recent droughts in California have been correlated with declines in wildflower diversity (Harrison et al. 2015; Copeland et al. 2016). Drought is also associated with reduced flower availability (Stevens & Frey 2010), flower size, and nectar volume (Carroll et al. 2001). This may enhance pollinator reliance on fewer plant species and/or increase competition, and could be particularly problematic for host-specialists (Stevens & Frey 2010). Drought periods can also accelerate seasonal plant senescence timing (Bell 1998), potentially reducing seasonal forage or breeding site availability later in the flight season (Stevens & Frey 2010). Low moisture availability can also cause plant latexes (i.e., the milky fluid in stems of some flowering plants) to become more viscous, potentially reducing larval fitness or survival by making leaf eating more difficult (Bell 1998).

Moisture-driven changes in plant abundance and diversity will likely impact native insect pollinator abundance, survival, distribution, and recruitment. For example, monarch butterfly (*Danaus plexippus*) abundance was positively correlated with moisture availability across the western United States (U.S.) from 1998–2007, likely due to the strong control soil moisture exerts on the abundance and phenology of its host plant milkweed (*Asclepias* spp.; Stevens & Frey 2010). Future declines in moisture availability as a result of drought and increasing climatic water deficit may exacerbate monarch population declines by limiting host plant availability (Stevens & Frey 2010; Jepsen et al. 2015).

It is possible that drought may enhance nesting opportunities for ground-nesting species by creating bare patches in areas that were previously covered with non-native thatch (Vuln. Assessment Reviewer, pers. comm., 2018). However, these benefits would likely be lost if drought also eliminated native foraging plants for these species (Vuln. Assessment Reviewer, pers. comm., 2018).

Regional Drought & Soil Moisture Trends	
Historical & current trends: • Drought years have occurred twice as often over the last two decades compared to the previous century (Diffenbaugh et al. 2015) • 2012–2014 drought set records for lowest precipitation, highest temperatures, and most extreme drought indicators on record (Griffin & Anchukaitis 2014; Diffenbaugh et al. 2015) • No trends available for soil moisture	*Projected future trends:* • Drought years are twice as likely to occur over the next several decades due to increased co-occurrence of dry years with very warm years (Cook et al. 2015) • 80% chance of multi-decadal drought by 2100 under a high-emissions scenario (Cook et al. 2015) • 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)

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Regional Drought & Soil Moisture Trends	
	• Decreased top-level soil moisture is likely even if precipitation increases due to temperature-related changes in evaporative demand (Pierce et al. 2018)
Summary of Potential Impacts on Species Group *(see text for citations)*	
• Reduced forage and host plant availability, quality, and diversity; may enhance competition or increase individual reliance on fewer plant species • Likely reductions in pollinator survival, abundance, recruitment, and distribution in response to declining resource availability • May benefit some ground-nesting species by increasing access to or extent of bare ground	

Precipitation amount and timing

Native insect pollinators exhibit several direct vulnerabilities to changes in precipitation regimes. For example, heavy precipitation has been found to limit bumble bee and butterfly foraging activity, potentially affecting fecundity and survival (Goulson 2010; Jepsen et al. 2015). In butterfly populations of the Coast Range north of San Francisco, wetter springs delayed the onset of butterfly life cycles and butterfly emergence over a 27-year period effectively shortening the foraging season for early-emerging species (Thorne et al. 2006). By contrast, reduced precipitation generally increases flying pollinator activity, fecundity, abundance, and/or species composition, especially at higher elevations (Mac Nally et al. 2003).

Shifts in precipitation may also alter plant phenology (Inouye 2008; Willis et al. 2013; Rundel & Millar 2016), affecting seasonal resource availability and potentially causing phenological mismatches (Memmott et al. 2007). For example, increased late spring or summer precipitation could extend the flowering season and/or delay plant senescence for some species, maintaining forage availability for native pollinators (Thorp et al. 2002) and increasing recruitment success (Stevens & Frey 2010).

Regional Precipitation Trends	

Historical & current trends:
• 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 regions (Rapacciuolo et al. 2014)
	Projected future trends:
	• Changes in mean annual precipitation by 2100 (compared to 1951–1980; (Flint et al. 2013; Flint & Flint 2014):[5]
	  ○ 20% decrease to 27% increase on the North Coast
	  ○ 20% decrease to 28% increase in the Northern Coast Range
	  ○ 17% decrease to 34% increase in the Northern Interior Coast Range

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

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Regional Precipitation Trends	
	○ 16% decrease to 18% increase in the Klamath Mountains ○ 19% decrease to 27% increase in the Southern Cascades ○ 23% decrease to 38% increase in the Great Valley • Seasonal changes are projected to be more significant as the wet season becomes wetter and shorter (i.e., later onset of fall rains and earlier onset of summer drought) and the dry season becomes drier and longer (Pierce et al. 2018; Swain et al. 2018) • Overall, interannual variability is expected to increase (Pierce et al. 2018; Swain et al. 2018)
Summary of Potential Impacts on Species Group *(see text for citations)*	
• Rainy conditions can slow development and delay emergence, potentially reducing foraging windows • Heavy precipitation limits adult foraging activity • Precipitation changes will likely alter plant phenology, which may restrict existing or create new foraging opportunities • Reduced precipitation may promote range expansions to higher elevations	

Storms and wind

Stormy conditions (e.g., heavy precipitation, high winds) and inclement weather (e.g., cloud cover) can reduce pollinator foraging opportunities, fecundity, and survival (Mac Nally et al. 2003; Forister & Shapiro 2003; Goulson 2010). For example, winter storms can kill overwintering monarch butterflies (Jepsen et al. 2015) and storms occurring during initial bumble bee queen foraging and colony-building periods can lead to starvation and mortality of the entire colony (Goulson 2010). Heavy rain associated with storms can also promote flooding, inundating and drowning larvae at or below the ground surface (National Research Council 2007; USFWS 2011, 2015). Storms can also alter habitat suitability for pollinators by causing vegetation damage or mortality (Jepsen et al. 2015).

Regional Storm Trends	
Historical & current trends: • Decline in the frequency of extreme two-day precipitation events between 1950 and 2009, with a slight decrease in the amount of precipitation received during extreme two-day events (Mass et al. 2010) • No trends available for storm-related wind events	*Projected future trends:* • Increase in storm intensity and duration, resulting in greater maximum precipitation rates and volume (Dettinger 2011; Shields & Kiehl 2016; Prein et al. 2017) • Slight to moderate increase in storm frequency (up to 30% increase in atmospheric river days, or ~2.5 days per year; (Dettinger 2011)

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Regional Storm Trends	
	• Projected statewide increases in daily extreme precipitation values of 5–20% by 2100 (Pierce et al. 2018) • No projections available for storm-related wind events
Summary of Potential Impacts on Species Group *(see text for citations)*	
• Enhanced mortality • Reduced foraging opportunities and fecundity • Reduced habitat suitability via vegetation damage	

Sensitivity and future exposure to changes in natural disturbance regimes

Regional experts evaluated native insect pollinators as having moderate-high sensitivity to changes in natural disturbance regimes (low confidence in evaluation), with an overall moderate future exposure to these stressors within the study region (moderate confidence). Key natural disturbance regimes that affect native insect pollinators include flooding, disease, and wildfire.[6]

Flooding

Several native pollinators are vulnerable to mortality from flooding. For example, flooding can inundate nests of ground-nesting taxa (National Research Council 2007). Similarly, species utilizing short-stature plants during larval development periods are vulnerable to flood inundation and mortality, particularly where water pools on non-porous soils (USFWS 2011, 2015).

Flooding is believed to have caused a decline in the Oregon silverspot butterfly (*Speyeria zerene hippolyta*) population adjacent to Lake Earl in Del Norte County in the winter of 2016–2017 (Vuln. Assessment Reviewer, pers. comm., 2018). Previous sequential drought years reduced the abundance of the butterfly's larval host plant (western early blue violet [*Viola adunca*]), and only plants in the wettest areas survived (Vuln. Assessment Reviewer, pers. comm., 2018). Then, an above-average precipitation winter in 2016–2017 caused flooding in the area, and the remnant host plants were inundated, likely drowning larvae (Vuln. Assessment Reviewer, pers. comm., 2018). Previous studies in the Lake Earl area had predicted that the majority of this butterfly's host plants/oviposition sites were at low risk of flooding (Damiani 2011 cited in U.S. Fish and Wildlife Service 2011). However, the loss of *V. adunca* in all but the wettest areas resulted in the remaining plants being concentrated in areas at increased risk of flooding, demonstrating the complexity of predicting impacts of interacting climate stressors on native pollinators (Vuln. Assessment Reviewer, pers. comm., 2018).

[6] All disturbance regimes presented were ranked as having a moderate or higher impact on this species group.

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Regional Flooding Trends	
Historical & current trends: • No trends available for flooding	*Projected future trends:* • More frequent/severe winter flooding due to an increase in extreme precipitation events (Dettinger 2011; AghaKouchak et al. 2018; Swain et al. 2018; Grantham et al. 2018) • State-wide, 200-year floods are expected to increase in frequency by 300–400%, becoming 50-year floods (Swain et al. 2018)
Summary of Potential Impacts on Species Group *(see text for citations)*	
• Mortality from ground-nest flooding or host plant inundation • Unanticipated impacts due to interacting climate stressors (e.g., drought combined with high precipitation enhanced mortality risk for riparian species)	

Disease

Disease contributes to pollinator mortality and fitness declines, depressing regional pollinator populations (Evans et al. 2008; USFWS 2011, 2015). For example, many invertebrates can carry bacteria of the *Wolbachia* genus, which is a maternally heritable intracellular bacteria (Nice et al. 2009). *Wolbachia* bacteria can have a variety of negative effects on insect reproductive biology, including preventing infected males from successfully fertilizing eggs of uninfected females or females carrying a different bacterial strain (Nice et al. 2009). Thus, the presence of the *Wolbachia* bacteria effectively separates a given insect population into two sub-populations that cannot breed, limiting recruitment and exacerbating the risk of inbreeding depression, population declines, and extirpation, particularly for species that already have small populations (Nice et al. 2009; USFWS 2011, 2015).

Exotic pathogens and parasites are particularly problematic for native pollinators, especially bumble bees. Observed declines in wild bumble bees are believed to be caused by the exotic parasite *Nosema bombi* (National Research Council 2007; Evans et al. 2008). Researchers hypothesize that western bumble bee (*Bombus occidentalis*) colonies were exposed to this microsporidian parasite during commercial rearing efforts in Europe (National Research Council 2007; Evans et al. 2008). When these bumble bees were returned to the U.S. for greenhouse use, infected individuals likely spread the disease to wild colonies and other similar bumble bee species by visiting shared floral resources, a process termed “pathogen spillover” (National Research Council 2007; Evans et al. 2008). Wild bumble bees are also vulnerable to other diseases via commercial pathogen spillover, including the protozoan *Crithidia bombi* and the tracheal mite *Locustacarus buchneri* (National Research Council 2007; Evans et al. 2008). Even in the absence of bee mortality, these diseases can impair pollinator fitness. For example, *C. bombi* can reduce queen colony founding success and worker foraging efficiency (Evans et al. 2008 and citations therein).

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Regional Disease Trends	
Historical & current trends: • No trends available for disease	*Projected future trends:* • No projections available for disease
Summary of Potential Impacts on Species Group *(see text for citations)*	
• Contribute to pollinator mortality, impaired fitness (e.g., foraging efficiency), and reduced recruitment, exacerbating the risk of inbreeding depression and population declines	

Wildfire

Native pollinator population numbers may decline in the short-term (i.e., hours to 1–2 months) following fire due to mortality from flame, heat, and smoke exposure (Swengel 2001; Cane & Neff 2011; Love & Cane 2016), although vulnerability varies depending on a species' life history and lifestage (Swengel 2001; Cane & Neff 2011). Species nesting on the ground surface and/or in the fuel layer (e.g., twigs, stems, herbaceous material) are frequently incinerated or exposed to lethal heat during fire (Swengel 2001; Cane & Neff 2011). Species with nests made of mud, resin, or leaf pulp are similarly vulnerable, unless nests are located far from flammable material (Cane & Neff 2011). Species that nest more than 5 cm (2 in) below ground are less susceptible to mortality because soil provides heat insulation (Cane & Neff 2011; Love & Cane 2016). Species that burrow or nest in tree wood may also experience insulation benefits from bark if the tree itself is not completely incinerated (Swengel 2001; Cane & Neff 2011). Similarly, species that nest in tree canopies are less vulnerable as long as flames fail to reach the canopy (Swengel 2001). Species' life histories may also predispose them to wildfire mortality, depending on when the fire occurs. For example, late-summer fires commonly occur when eggs, larvae, and pupae are already in nests (Cane & Neff 2011). Although adults may be able to escape via flight, eggs, larvae, and pupae are generally immobile, increasing mortality risk (Cane & Neff 2011; Love & Cane 2016).

Native pollinators also experience indirect wildfire effects. Wildfires can reduce the availability and proximity of forage sources and nesting sites, as well as reduce groundcover that protects below-ground nests from wind and water erosion (Swengel 2001; Williams et al. 2010; Cane & Neff 2011; Love & Cane 2016). Habitat simplification may contribute to short-term declines in pollinator species richness (Swengel 2001; Lillie 2011). For example, a study conducted three years after the Biscuit Fire in northwestern California found that bee species richness on burned serpentine sites was lower than bee species richness on non-burned serpentine sites, likely due to reduced nest site availability (Lillie 2011). Reductions in forage and nesting site availability can also shift pollinator community composition (Swengel 2001; Lillie 2011; Love & Cane 2016). For example harsh post-fire conditions may inadvertently favor pollinator taxa that thrive in xeric, exposed environments (Swengel 2001). Generalist pollinator species may be better suited to post-fire environments than specialist species due to their ability to utilize a wider range of pollen and nectar sources, which can be beneficial in a resource-poor burned landscape (Love & Cane 2016).

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Intermediate- and longer-term indirect fire effects are more variable, depending on variables such as taxa (Williams et al. 2010) and habitat type (Lillie 2011). Burned landscapes may experience large flowering events the following spring, enhancing herbaceous floral resource availability (Seefeldt et al. 2007; Hanula et al. 2016). Wildfire may also increase future nesting opportunities for cavity-nesting species by damaging trees (Swengel 2001; Cane & Neff 2011), and promote more open, sunny forest conditions, which can provide optimal temperatures for foraging (Hanula et al. 2016). Above-ground nesting bee abundance typically rebounds as vegetation communities recover (Williams et al. 2010), but below-ground nesting bee abundance can decline over time due to herbaceous post-fire growth that reduces the availability of bare soil (Williams et al. 2010). Most native bees have maximum foraging ranges between 150–600 m (492–1,968 ft), with smaller body sizes translating to smaller maximum foraging distances (Black et al. 2009). Thus, recolonization of burned areas is highly dependent on fire size and the distance from the source population to the interior of burned patches, as well as short-term persistence and health of adjacent pollinator source populations in the face of resource scarcity (Swengel 2001).

Altered fire regimes as a result of climate change are likely to be problematic for native pollinators (Swengel 2001). While many pollinator species have evolved with wildfire, more intense fires may increase mortality due to higher soil temperatures prolonging exposure to lethal thresholds (Cane & Neff 2011). Increasing fire frequency and size may extirpate isolated populations of rare species (Hanula et al. 2016), reduce available pollinator source populations, or create prohibitively large foraging and/or migration distances, reducing post-fire recolonization and recovery capacity (Swengel 2001; Love & Cane 2016). Too frequent fire may also permanently reduce or eliminate plant species used for pollinator nesting or foraging, leading to long-term resource scarcity (Swengel 2001). Increasing fire frequency may also lead to conversion of native-dominated shrubland and grassland to non-native annual grassland (Haidinger & Keeley 1993; Lenihan et al. 2008; Cornwell et al. 2012; Keeley & Brennan 2012), which typically has lower floral resource availability for pollinators (Vuln. Assessment Reviewer, pers. comm., 2018).

Regional Wildfire Trends	

Historical & current trends:
• 85% of U.S. Forest Service lands in northern California are burning less frequently compared to pre-1850 fire return intervals, largely due to fire suppression (Safford & Van de Water 2014)
• Fire size and total area burned increased on U.S. Forest Service lands in northwestern California between 1910-2008, with the highest values occurring after 2000 (Miller et al. 2012)
	Projected future trends:
	• State-wide, up to 77% increase in mean annual area burned and 50% increase in the frequency of extremely large fires (>10,000 ha) by 2100 (Westerling 2018)
	  ○ Greatest increases in burned area (up to 400%) occur in montane forested areas in northern California (Westerling et al. 2011; Westerling 2018)
	  ○ Less significant increases or possible decrease along the North Coast (Westerling et al. 2011)

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Regional Wildfire Trends	

• Changes in large fires (over 400 ha) in the inland northern California/Sierra Nevada region since the 1970s (Westerling 2016):
  ○ 184–274% increase in frequency
  ○ 270–492% increase in total area burned
  ○ 215% increase in length of the fire season
• Changes in fire size, area burned, and fire frequency over the past several decades remain well below historical tribally-influenced frequency and extent of burning in California (Stephens et al. 2007)
• No significant trends in the average areal proportion of high-severity fire were documented in northwestern CA from 1984–2008 (Miller et al. 2012; Parks et al. 2015; Law & Waring 2015; Keyser & Westerling 2017)
  ○ The relatively short period of record for fire severity data may obscure long-term trends
  ○ To date, there are no peer-reviewed studies on trends in northern California fire severity that include data from the last ten years
	• Little projected change in fire severity in northwestern California by 2050 in models based solely on historical fire-climate relationships (Parks et al. 2016)
	  ○ However, human activity and fuel buildup from decades of fire suppression have altered historical fire-climate relationships (Taylor et al. 2016; Syphard et al. 2017; Wahl et al. 2019), and projections that incorporate these factors suggest that more significant increases in fire severity and size may occur (Mann et al. 2016; Wahl et al. 2019)
	• The majority of impacts to natural and human ecosystems come from extreme fire events (i.e., fires that have a low probability of occurring in any given place and time), which are likely to increase over the coming century (Westerling 2018)
	  ○ Generally, these patterns are not well-represented in studies that evaluate indices of mean fire size, intensity/severity, etc.

Summary of Potential Impacts on Species Group *(see text for citations)*

- *Immediate*:
  - Enhanced mortality and reduced abundance, particularly for immobile eggs and larvae and for species nesting on the ground surface, in the fuel layer, or in shallow below-ground nests
  - Increased mortality from soil heating with more intense fire and/or prolonged exposure
  - Reduced forage and nest site availability, reducing species richness and altering species composition
  - Higher exposure to extreme conditions and wind/water erosion without surface vegetation
- *Short-term (~2-year)*:
  - Site recolonization, depending on persistence of source populations, capacity to migrate, and migration distance
  - Increased foraging opportunities via floral resource blooms in burned areas
  - Increased cavity-nesting opportunities in fire-damaged trees
  - Open canopies enhance foraging conditions by promoting optimal temperatures
- *Long-term*:
  - Possible extirpation of local pollinator populations frequently impacted by fire
  - Possible loss of key forage and host plants or cause vegetation type conversions, reducing plant resource availability for pollinators
  - Increased fire size, frequency, and severity may prohibit recolonization and recovery

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Dependency on habitat and/or other species

Regional experts evaluated native insect pollinators as having moderate-high dependency on sensitive habitats (low confidence in evaluation) and moderate-high dependency on prey or forage species (low confidence).

Native insect pollinators exhibit two general habitat requirements: native flowers that provide pollen/nectar sources and suitable nesting or egg-laying sites (Black et al. 2009). However, native insect taxa vary widely in their degree of specificity related to these factors. Many native pollinators exhibit generalist foraging behaviors; for example, bumble bees visit numerous flowering plant species to obtain nectar and pollen, and in the Klamath Ranges, they have been documented to visit over 226 plant genera (Thorp et al. 2002). Many butterflies are dependent on specific larval host plants. For example, the native Oregon silverspot butterfly is dependent on western early blue violets (USFWS 2011), the seaside hoary elfin (*Incisalia polia maritima*) is dependent on kinnikinnick (*Arctostaphylos uva-ursi*; Ross et al. 2005), and the Mardon skipper (*Polites mardon*) is dependent on fescue grasses (*Festuca* spp.) (Black & Vaughan 2005). Although native pollinators occur across all habitat types in northern California, some species are restricted to specific habitats while others utilize multiple habitat types (Vuln. Assessment Reviewer, pers. comm., 2018). For example, some bumble bees require below-ground cavities created by rodents for nesting (Evans et al. 2008), while other ground-nesting bees will excavate their own nests (Black et al. 2009).

Specialist pollinator species may be more vulnerable to climate changes than generalist species, particularly if climate change reduces the local availability or quality of plant hosts (Ross et al. 2005; Stevens & Frey 2010; Love & Cane 2016) and/or causes a mismatch in landscape distribution between pollinator and host plant (Schweiger et al. 2008). For example, monarch butterflies are reliant on a single host plant genus for larval recruitment (Bell 1998), increasing their vulnerability to climate-driven reductions in milkweed abundance and/or distribution in addition to the direct impacts of climate change on this species (Stevens & Frey 2010). Milkweed germination, survivorship, growth, and seed production has been found to decline in response to drought and low moisture availability (Bell 1998), indicating that future climatic changes could constrain monarch distribution and recruitment success through host plant declines (Stevens & Frey 2010). Host plant availability, dispersal, and migration in response to climatic change will also affect the ability of insect pollinators to colonize newly climatically suitable areas and/or persist within their current range in physiological refugia (Schweiger et al. 2008).

Sensitivity and current exposure to non-climate stressors

Regional experts evaluated native insect pollinators as having moderate-high sensitivity to non-climate stressors (moderate confidence in evaluation), with an overall moderate current exposure to these stressors within the study region (low confidence). Key non-climate stressors

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that affect native insect pollinators include fire suppression, residential and commercial development, poisons, invasive and problematic species, agriculture, and livestock grazing.[7]

Fire suppression

Fire suppression can alter habitat availability and/or habitat characteristics preferred by native insect pollinators (USFWS 1997, 2015; Evans et al. 2008). For example, fire suppression has increased woody vegetation encroachment (i.e., natural succession) and promoted thatch accumulation in coastal prairie grasslands (Ford & Hayes 2007; Stromberg et al. 2007; BLM 2008, 2013). Both of these outcomes reduce habitat suitability for the Behren's silverspot butterfly (*Speyeria zerene behrensii*), which requires early successional coastal prairies for foraging and oviposition sites (USFWS 1997, 2015). Woody vegetation encroachment as a result of fire suppression has similarly reduced habitat availability and suitability for generalist grassland pollinators, such as the western bumble bee (Evans et al. 2008). Fire suppression generally has widespread negative effects on plant species richness (England 1988a, 1988b; D'Antonio et al. 2006), impacting floral resource availability for pollinators.

Fire suppression can also negatively impact forest pollinators by promoting closed canopy conditions, which can limit optimal pollinator foraging conditions by reducing sun penetration and temperatures (Hanula et al. 2016). Fire suppression also increases understory shrub cover, reducing the abundance and diversity of herbaceous foraging resources available to forest pollinators (Hanula et al. 2016).

Residential and commercial development

Residential and commercial development reduces and fragments pollinator habitat, affecting pollinator abundance, species richness, and migration and dispersal opportunities. Development can directly reduce pollinator habitat by removing critical larval host plants, removing key foraging resources, and/or eliminating breeding, nesting, and overwintering sites, particularly at lower elevations (USFWS 1997; Ross et al. 2005; Evans et al. 2008; Forister et al. 2010; Casner et al. 2014; Jepsen et al. 2015). For example, urban development has contributed to the loss of monarch butterfly overwintering sites in coastal California, including Mendocino County (Jepsen et al. 2015). Urban development has also been linked with the elimination of several Behren's silverspot butterfly populations in northern California via habitat destruction (USFWS 1997). In general, urbanized landscapes do not support the same pollinator species richness of wild landscapes, although home gardens can provide some marginal pollinator habitat (Evans et al. 2008). Urbanized landscapes also limit dispersal opportunities relative to more natural landscapes (e.g., native ecosystems, agricultural fields; Casner et al. 2014). Additionally, increasing urban and suburban development and associated urban horticulture and ornamental plantings may increase exposure to exotic plant species and their associated impacts, particularly for forest pollinators that have previously had lower exposure (Graves & Shapiro 2003).

[7] Non-climate stressors presented are those ranked as having a moderate or higher impact on this species group; additional non-climate stressors that may influence the species group to a lesser degree include roads/highways/trails and recreation.

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Reduced habitat availability and increased habitat fragmentation as a result of development will likely undermine pollinator resilience to climate change (Forister et al. 2010). For example, despite climatic conditions generally becoming more favorable for butterflies at higher elevations in the Sierra Nevada, declines in butterfly species richness at low elevation sites have reduced species richness at higher elevation sites by reducing population sources for seasonal migration and colonization opportunities (Forister et al. 2010). This pattern was primarily evident in well-dispersing species, which is concerning given that well-dispersing, generalist species are typically assumed to be more resilient to climate change than species with more limited dispersal or more specialized habitat preferences (Forister et al. 2010). This study demonstrates how human land uses can interact with climate change to enhance the vulnerability of traditionally "resilient" pollinator species (Forister et al. 2010).

Poisons

Insecticides and herbicides used in agricultural and urban areas can cause direct pollinator mortality (USFWS 1997; Kremen et al. 2002a). For example, western bumble bees are vulnerable to mortality when they ingest insecticides, gather poisoned pollen, or absorb insecticides through their skin (National Research Council 2007; Evans et al. 2008). Similarly, monarch butterfly mortality patterns overlap with regions known to apply pesticides (Jepsen et al. 2015). Larvae may be particularly susceptible to insecticide-related mortality (USFWS 1997). Insecticides can also have sub-lethal effects on pollinators, potentially impacting flight, navigation, and foraging efficiency (Evans et al. 2008). Herbicides (particularly broad-spectrum herbicides) also contribute to pollinator mortality and undermine reproductive success by reducing abundance of pollen and nectar resources (Shepherd et al. 2003), specialist pollinator host plants (Jepsen et al. 2015), and nest sites (Evans et al. 2008). Overall, population-level impacts from pesticides are likely highest when insecticide or herbicide application overlaps with peak pollinator foraging and recruitment periods (Goulson et al. 2008). Social insect species may be more susceptible to pesticide impacts than solitary species because their behavioral patterns (e.g., longer activity periods and broader foraging patterns) increase risk of pesticide exposure (Williams et al. 2010).

Invasive and problematic species

Native bees are sensitive to increased competition with and disease risk from the honey bee (*Apis mellifera*) and non-native bumble bees, both of which were originally introduced for commercial agricultural pollination (National Research Council 2007; Evans et al. 2008). In the study region, honey bees are mostly a problem in agricultural edges where they increase competition for limited floral resources (Vuln. Assessment Reviewer, pers. comm., 2018). However, if honey bees increase in wildlands (e.g., via apiary permits granted by the BLM), they could represent a more widespread threat for native pollinators (Cane & Tepedino 2017). For example, one study found that a 40-hive apiary present on wildlands for three months would result in the collection of the same amount of pollen collected by four million wild bees (Cane & Tepedino 2017). Forage specialist pollinators who utilize the same plants as honey bees, as well as species foraging during the same time as apiary placement (e.g., late summer), would be most at risk from increasing apiary use on wildlands (Cane & Tepedino 2017)

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Exotic pollinators can exacerbate native pollinator population declines by directly competing with native pollinators for floral and pollen resources and nest sites (National Research Council 2007; Cane & Tepedino 2017). For example, the presence of introduced honey bees increases the amount of time spent and distance traveled foraging for western bumble bees, resulting in reduced larval production (Thomson 2004, 2006; Cane & Tepedino 2017). Higher foraging requirements as a result of nectar and pollen scarcity can have other cascading impacts on native bee recruitment, such as increasing the risk of nest predation while adults are foraging (summarized in Cane & Tepedino 2017). Competition impacts may be most severe when floral resources are limited (Evans et al. 2008), which may become more common under climate change (e.g., as a result of drought; Copeland et al. 2016).

Exotic pollinators can also enhance disease and pest risk for native pollinators (Evans et al. 2008). For example, bumble bee rearing and cross-continental transport for commercial greenhouse pollination has contributed to increased disease incidence and mortality among wild native bumble bee species (National Research Council 2007; Evans et al. 2008). Honey bees are also the primary host for the small hive beetle (*Aethina tumida*), which can infest and damage bumble bee colonies (Evans et al. 2008). Exotic pollinators can also enhance non-native vegetation pollination and seedset, contributing to habitat modification (National Research Council 2007).

Native pollinators can experience both positive and negative impacts from the establishment and spread of non-native invasive plants. Some exotic plants may increase available forage (Thorp et al. 2002; Graves & Shapiro 2003; Hanula et al. 2016; Loffland et al. 2017), although specialist species are less likely to be able to take advantage of exotic-mediated increases in resource availability (Graves & Shapiro 2003). For example, yellow starthistle (*Centaurea solstititalis*) is a major nectar source for Central Valley and foothill butterflies (Graves & Shapiro 2003), and as a late season exotic annual, may also prolong nectar and pollen availability for bumble bee foraging and breeding in the late summer (Thorp et al. 2002; Hanula et al. 2016). Thirty-four percent of California's butterfly taxa have been shown to feed or oviposit on exotic plant species, with many more likely using introduced species as nectar sources (Graves & Shapiro 2003). Some butterfly species have extended their flight periods, increased the number of generations per year, or expanded their geographic ranges by foraging on exotic plants (Graves & Shapiro 2003). However, utilizing exotic plant species may expose native pollinators to additional risks. For example, increased non-native floral resources could alter pollinator phenologies (e.g., prolonged nectar availability due to the presence of non-native plants could result in longer flight seasons), potentially causing other portions of the life cycle to be out of sync (Vuln. Assessment Reviewers, pers. comm., 2018). Additionally, reliance on exotic plants treated with herbicide may increase pollinator exposure to these toxins (Vuln. Assessment Reviewers, pers. comm., 2018).

Exotic plants that do not supply forage resources (Graves & Shapiro 2003; Evans et al. 2008) and/or that suppress native floral resources (Menke 1992; Dyer & Rice 1999; Corbin & D'Antonio 2004; Suttle et al. 2007; Molinari & D'Antonio 2014) could negatively impact

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pollinator foraging and survival (Graves & Shapiro 2003; Ross et al. 2005; Evans et al. 2008). For example, barbed goatgrass (*Aegilops triuncialis*) invasion has likely contributed to declines in the abundance and number of native pollinator taxa in inland California grasslands by reducing floral resources (Aigner et al. 2011; Vuln. Assessment Reviewer, pers. comm., 2018). The removal of barbed goatgrass caused a resurgence of native wildflowers and a four-fold increase in the abundance and number of native pollinator taxa relative to control plots (Aigner et al. 2011). Nesting bee density also exhibited an increase in treated plots (although statistically insignificant), likely responding to increases in bare ground when barbed goatgrass was removed (Aigner et al. 2011; Vuln. Assessment Reviewer, pers. comm., 2018).

Exotic vegetation can have many other negative impacts on native pollinators. For example, some exotic plants may promote butterfly oviposition but be toxic to larvae, contributing to recruitment failures and potential population declines (Graves & Shapiro 2003). Exotic plant utilization may also promote genetic homogenization by bringing previously isolated insect sub-populations into contact and increase invasion of non-native insect competitors (Graves & Shapiro 2003). Exposure to exotic plant species varies according to location; for example, alpine species having significantly lower exposure than lower elevation species where exotic invasion is currently much higher (Graves & Shapiro 2003). Pollinators in grassland and shrub communities are increasingly being threatened by pervasive invasion of annual grasses, particularly under more frequent fire regimes (Dukes & Shaw 2007; Keeley et al. 2011; Keeley & Brennan 2012; Eviner 2016).

Agriculture

Agriculture has contributed to the decline of native pollinator populations by increasing habitat loss and fragmentation, decreasing floral resource diversity and abundance, increasing exposure to pesticides and herbicides, and introducing exotic pollinators (Kremen et al. 2002b; National Research Council 2007; Evans et al. 2008; Casner et al. 2014). Following initial landscape conversion for agricultural production, agricultural intensification has further reduced remnant suitability of agricultural landscapes for native pollinators (Kremen et al. 2002b; National Research Council 2007; Evans et al. 2008). For example, monoculture agriculture, the loss of legume crop rotations with the rise of fertilizer amendment, fewer hedgerows, and the reduction of field edges and weeds are particularly problematic for native pollinators (Kremen et al. 2002b; National Research Council 2007; Evans et al. 2008). These practices decrease floral resource abundance and diversity and reduce nesting site availability, contributing to pollinator declines and reductions in species richness (Kremen et al. 2002b; National Research Council 2007; Evans et al. 2008).

Exact impacts vary depending on species and farming practices used (Rundlöf et al. 2008; Williams et al. 2010; Morandin & Kremen 2013). Organic agriculture tends to support more native pollinators than conventional agriculture, likely due to less use of pesticides and herbicides (Rundlöf et al. 2008). Additionally, use of hedgerows in the Central Valley has been found to increase native bee species richness and abundance, particularly for less-common species (Morandin & Kremen 2013). Species can also exhibit variable responses to the same agricultural stressors depending on their life history (Williams et al. 2010). For example, below-

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ground nesting bees are more vulnerable than above-ground nesting bees to direct mortality and loss of nesting sites from agricultural tilling (Williams et al. 2010). Comparatively, above-ground nesting bees are more vulnerable to agricultural intensification, which reduces native vegetation and woody debris required for above-ground nesting sites but typically does not affect overall access to bare soil for below-ground nesting (Williams et al. 2010).

Agriculture can also have positive impacts on pollinators. For example, the agricultural crop alfalfa (*Medicago sativa*) is used by both native bumble bees and butterflies as a pollen and nectar source, potentially helping to maintain forage availability throughout the peak breeding and foraging season (Thorp et al. 2002) and contributing to range expansions and higher population numbers (Graves & Shapiro 2003). However, pollinators utilizing alfalfa and other crops may also experience higher exposure to pesticides or herbicides (Vuln. Assessment Reviewers, pers. comm., 2018).

Livestock grazing

Livestock grazing can have variable impacts on native pollinators depending on intensity, timing and duration (Swengel 2001; DeBano et al. 2016). In general, high-intensity grazing is associated with declines in pollinator abundance and species richness (Swengel 2001; Hatfield & LeBuhn 2007). Herbivory impacts plant abundance and community structure, composition, and diversity (Vulliamy et al. 2006; Black et al. 2011), altering the availability of pollen and nectar resources, larval host plants, and nesting material (USFWS 1997; Vulliamy et al. 2006; Evans et al. 2008; Black et al. 2011). Trampling can cause direct pollinator mortality or reduce the suitability of plant resources and nesting habitat (USFWS 1997; Swengel 2001; Evans et al. 2008). The presence of livestock also alters physical habitat characteristics for ground-nesting bees by impacting soil compaction and stability (Kimoto et al. 2012; Schmalz et al. 2013).

The timing of grazing can also impact pollinator survival and recruitment. For example, grazing during periods of low floral resource availability (e.g., mid- to late-summer) can severely reduce remnant floral resources available to pollinators (Hatfield & LeBuhn 2007; DeBano et al. 2016), and grazing during egg and larval lifestages can cause recruitment failure by increasing direct larval mortality and/or creating more severe microclimates that undermine survival (Swengel 2001). Similarly, the type of grazing animal may also cause differing impacts. For example, a multi-year study in montane meadows of the Sierra Nevada found bumble bee abundance to be affected significantly by sheep (*Ovis aries*) grazing (i.e., no bumble bees present after sheep grazing) but to be less or minimally affected by cattle (*Bos taurus*) grazing, likely because sheep grazing tends to remove all floral resources from the grazed parcel (Hatfield & LeBuhn 2007).

Sensitivity to grazing also depends on individual pollinator species. For example, univoltine species (i.e., produce only a single brood per season) are more sensitive to disruption or population losses from grazing than bivoltine species (i.e., produce two broods per season; Swengel 2001). Some native pollinator species may also prefer conditions created by grazing (e.g., lower thatch, more bare ground; Swengel 2001).

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With careful management, grazing could be used to benefit native pollinators. For example, grazing could be used to reduce the abundance of non-native plant species that compete with native plants required by pollinators (USFWS 1997). However, grazing treatments will need to be very site-specific and adjusted according to pollinator species present (Swengel 2001).

Adaptive Capacity

Native insect pollinators were evaluated by regional experts as having moderate overall adaptive capacity (low confidence in evaluation).

Species group extent, integrity, connectivity, and dispersal ability

Regional experts evaluated native insect pollinators as having a high geographic extent (high confidence in evaluation), low overall health and functional integrity (moderate confidence), and a moderate degree of connectivity between populations (low confidence).

Regional experts evaluated native insect pollinators as having a moderate dispersal ability (low confidence in evaluation). Barriers to dispersal were evaluated as having a moderate-high impact on the species group (low confidence). Land-use conversion, agriculture, invasive and other problematic species, and roads/highways/trails were identified as the primary barriers to dispersal.[8]

While native insect pollinators are widely distributed and occupy diverse habitats across northern California (Black et al. 2009), regional pollinator population numbers, distributions, and species richness are declining in response to multiple stressors, including land-use change, human activities (e.g., pesticide use), disease, and climate change (Thorne et al. 2006; Evans et al. 2008; Forister et al. 2010, 2011; Casner et al. 2014; Jepsen et al. 2015; Kerr et al. 2015). For example, a 27-year study in the Coast Range north of San Francisco documented the loss of at least three butterfly species over the duration of the study (Thorne et al. 2006). Western bumble bee populations have declined significantly in central California since 1998, almost to the point of extirpation (Evans et al. 2008). Habitat loss and fragmentation, as well as detrimental human land-use activities, have contributed to the listing of several native pollinators in northern California under the federal and state Endangered Species Acts (ESA), including the Behren's silverspot butterfly, Oregon silverspot butterfly, and the Lotis blue butterfly (*Plebejus anna lotis*; potentially extinct; USFWS 1997, 2011; Essig Museum of Entomology 2018). Small and/or isolated pollinator populations are more vulnerable to extirpation from stochastic environmental events (e.g., fire, annual weather fluctuations; USFWS 1997; Casner et al. 2014) and a loss of genetic diversity, which reduces resilience to changing environmental conditions (USFWS 1997). Pollinator population trends in northern California mirror global trends of declining pollinator abundance, species richness, and range size that have been linked with elevated human disturbance and habitat loss/fragmentation (National Research Council 2007; Winfree et al. 2009; Kerr et al. 2015).

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

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Land-use conversion has and will continue to reduce native pollinator habitat availability and connectivity, further affecting native insect pollinator diversity, persistence, abundance, and dispersal in the face of climate change (Winfree et al. 2009; Forister et al. 2010). Land use plays a large role in determining landscape permeability for pollinators (Casner et al. 2014). For example, although they are not ideal habitat, agricultural fields and weedy lots can act as butterfly migration corridors (Casner et al. 2014). While initial conversion from native ecosystem to agriculture undoubtedly reduces habitat availability and landscape connectivity for pollinators, modern day transitions from agriculture to urban development result in more significant losses because urban landscapes are less hospitable to pollinators than agricultural landscapes (Casner et al. 2014). Further, land-use change can interact with climate change to restrict habitat connectivity and suitability, as more severe climatic conditions may limit suitability of remnant migration corridors and habitat patches (Casner et al. 2014). Other land uses can also limit habitat connectivity, dispersal, and foraging opportunities. For example, most native pollinators are vulnerable to mortality from vehicle collisions (Evans et al. 2008). In general, increasingly isolated populations, inhospitable landscapes, and longer required dispersal distances limit the potential for pollinator populations to recolonize potentially suitable habitat areas following disturbance (Casner et al. 2014).

Species mobility and life history also affect dispersal and migration opportunities (Hatfield & LeBuhn 2007; Black et al. 2009). Aerial insects likely have higher dispersal capacity than non-flying insects, although life history mediates aerial dispersal capacity (Hatfield & LeBuhn 2007; Black et al. 2009). For example, native bees must remain near their nesting sites during the larval rearing period, which limits seasonal dispersal and foraging opportunities (Black et al. 2009). Most native bees have maximum foraging ranges between 150–600 m (492–1,968 ft), with smaller body sizes translating to smaller maximum foraging distances (Black et al. 2009).

Intraspecific/life history diversity

Regional experts evaluated native insect pollinators as having moderate-high life history diversity (moderate confidence in evaluation), high genetic diversity (high confidence), low-moderate behavioral plasticity (low confidence), and low-moderate phenotypic plasticity (low confidence).

The diverse topography of northern California likely contributes to native pollinator diversity and species richness. For example, studies in mountain ranges within the Great Basin found butterfly species richness to be positively correlated with topographic heterogeneity, likely because diverse topography supported a higher diversity of plant communities as well as a higher abundance of microrefugia from extreme events (Mac Nally et al. 2003). However, pollinator species richness in northern California has declined with increasing human land use and climate change (Forister et al. 2010; Casner et al. 2014).

While high species diversity within the native pollinator guild contributes to high genetic diversity as a whole, declining population sizes and increasingly fragmented native pollinator

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populations may undermine genetic diversity of individual species. For example, western bumble bees are susceptible to inbreeding due to low effective population size (Packer & Owen 2001). Inbred populations typically have smaller colonies (Herrmann et al. 2007), and may be more vulnerable to diseases, pests, and further habitat fragmentation (Evans et al. 2008).

Native pollinators have diverse life histories, with varied patterns of dispersal and migration, timing of emergence and development, sociality (e.g., colony-building or solitary bees), and reproductive strategies (Kremen et al. 2002a; Thorp et al. 2002; Forister & Shapiro 2003). An individual species' life history can greatly impact its resilience to climate change. For example, species that exhibit prolonged dormancy are better able to weather periods of unfavorable conditions, while those without extended dormancy are more susceptible to annual fluctuations in resource availability and mortality from extreme events (Cane & Neff 2011). Similarly, bivoltine and multivoltine species (e.g., many native bees) are less vulnerable to within-season resource scarcity, disturbance, and associated population declines than univoltine species (e.g., western bumble bee; Swengel 2001; Kremen et al. 2002a). This is because having multiple reproductive events, broods, or nests per season allows species to track fluctuating resource availability (Kremen et al. 2002a) and decreases the risk of significant population losses in response to short-term extreme events (Swengel 2001).

Native pollinators exhibit some behavioral diversity. For example, they exhibit thermoregulatory behaviors (Buckley & Kingsolver 2012; Kellermann et al. 2012), such as shifting the location and/or timing of foraging to help moderate their body temperature (Andrew et al. 2013; Sunday et al. 2014). Additionally, it is believed that some species may be able to switch which larval host and/or forage plant species they utilize in response to changing plant phenologies and forage resource availability (Memmott et al. 2007; Schweiger et al. 2008).

Resistance and recovery

Regional experts evaluated native insect pollinators as having low resistance to climate stressors and natural disturbance regimes (moderate confidence in evaluation). Recovery potential was evaluated as low-moderate (low confidence).

Population declines in many pollinator species suggest that this species group's resistance to climate changes and other stressors (e.g., habitat fragmentation, disease) may be low (Evans et al. 2008; Jepsen et al. 2015). For most native insect pollinators, the egg and larval stages of their life cycle are more vulnerable to mortality than the adult stage due to limited mobility (Goulson 2010; Cane & Neff 2011). For example, bee larvae are nearly immobile and largely defenseless, and depend on adult bees to bring them food resources (Goulson 2010). Resistance to climate stressors also depends on individual species' life history and behavioral characteristics; for example, above-ground nesters and ground-nesters with shallow nesting depths are more vulnerable to fire mortality than species that nest in deeper below-ground areas (Swengel 2001; Cane & Neff 2011).

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Native insect pollinators have short generation times (Mac Nally et al. 2003; Goulson 2010). This generally enables them to respond rapidly post-disturbance and/or in response to management applications (e.g., habitat restoration; Mac Nally et al. 2003; Hanula et al. 2016). Short generation times also increase the potential for genetic adaptation to changing climatic conditions (Mac Nally et al. 2003; Angilletta et al. 2007). However, even with fast generation times, evolutionary adaptation may not be able to keep pace with projected rates of climate change, which emphasizes the importance of facilitating range shifts to maintain biodiversity over the long-term (Kellermann et al. 2012). Additionally, certain traits are less adaptable; studies in several ectotherm taxa, including insects, have found that species' thermal maxima are less adaptable over time than thermal minima (Kellermann et al. 2012; Sunday et al. 2014).

Management potential

Public and societal value

Regional experts evaluated native insect pollinators as having moderate public and societal value (low confidence in evaluation).

Some native pollinators are well-known to and valued by the public, such as the monarch butterfly and other "pretty" pollinators (e.g., other butterflies, some bees; Vuln. Assessment Reviewer, pers. comm., 2018). There have been significant public education and outreach efforts surrounding native insect pollinators by regional groups such as the Xerces Society for Invertebrate Conservation (Vuln. Assessment Reviewer, pers. comm., 2018). While this has increased general knowledge of and appreciation for pollinators and the ecosystem services they provide, policy-makers still lack sufficient knowledge to make informed monetary, policy, and management decisions pertaining to this species group (Vuln. Assessment Reviewer, pers. comm., 2018). Additionally, national-level discussions regarding pollinator conservation often fail to adequately address the differences between native pollinators and commercial non-native pollinators (e.g., honey bees), including the threats the latter group present to native insect diversity and persistence (Geldmann & González-Varo 2018).

For species listed as threatened or endangered under federal and state ESAs (e.g., Behren's silverspot butterfly, Oregon silverspot butterfly), there is some regulatory support for management of the individual pollinators and their associated habitats (USFWS 2011, 2015). Other rare species proposed for listing as endangered include the Franklin's bumble bee (*B. franklinii*), which has an extremely narrow distribution (northern California and southern Oregon; The Xerces Society 2018; USFWS 2019), and the western bumble bee, which has experienced severe population declines (Sheffield et al. 2016; The Xerces Society 2018).

Management capacity and ability to alleviate impacts[9]

Regional experts evaluated the potential for reducing climate impacts on native insect pollinators through management as moderate-high (low confidence in evaluation).

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

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Natural habitat protection, habitat restoration, native plant management, and reduction of non-climate stressors (e.g., pesticide use; competition from introduced honeybees; habitat loss to urbanization, agriculture, and invasive plant species) are likely to represent the most significant management opportunities for native pollinators in the face of climate change (Kremen et al. 2002b, 2007; Black et al. 2009, 2011; Jepsen et al. 2015; Hanula et al. 2016; Vuln. Assessment Reviewer, pers. comm., 2018; Wojcik et al. 2018). However, the high diversity of native pollinators in the study region and the wide range habitats they occupy make management generalizations difficult (Rundlöf et al. 2008; Black et al. 2009; DeBano et al. 2016). In general, restoration and enhancement plantings and other management actions should consider pollinator life histories, forage and nesting requirements, and climate projections (Black et al. 2009). For example, native planting efforts can aim for continual flower resource availability for bees from early spring through late summer (Williams et al. 2015; Havens & Vitt 2016), and can consider utilizing drought-tolerant plants that are likely to be more resilient to future water shortages (Vuln. Assessment Reviewer, pers. comm., 2018). Similarly, management practices such as grazing, prescribed burning, and post-fire restoration and habitat management can be tailored to avoid vulnerable life history stages, accommodate foraging and nesting needs, and designed to maintain source populations for later recolonization (Swengel 2001; Black et al. 2009; Hanula et al. 2016; Brown et al. 2017; Loffland et al. 2017; Wojcik et al. 2018). In general, addressing current threats to native pollinator populations through management activities is likely to make them more resilient to emerging threats from climate change (Vuln. Assessment Reviewers, pers. comm., 2018). Additional management opportunities identified by study participants include increasing research and monitoring of phenological shifts to inform future pollinator management strategies, and engaging citizen scientists to increase monitoring capacity (Vuln. Assessment Reviewers, pers. comm., 2018).

Ecosystem services

Native insect pollinators provide a variety of ecosystem services, including:

- Provisioning of food and genetic resources;
- Regulation of pollination;
- Cultural/tribal uses for spiritual/religious purposes, aesthetic values, cultural heritage, and inspiration (Vuln. Assessment Reviewer, pers. comm., 2018).

Native insect pollinators are best known for the pollination services they provide to native vegetation (Kremen et al. 2002a; Evans et al. 2008) and regional agriculture (Kremen et al. 2002a, 2002b; Losey & Vaughan 2006; Black et al. 2009). Across the U.S., native bees provide pollination services valued at $3 billion per year (Losey & Vaughan 2006), and in many cases, native insect pollinators provide superior crop pollination services relative to imported pollinators, such as the honey bee (Kremen et al. 2002a). Native bees are considered the most productive pollinators, providing critical agricultural and native flower pollination (Black et al. 2009 and citations therein). Native flies are critical pollinators in alpine areas (Black et al. 2009 and citations therein).

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Pollination webs are somewhat resilient to the loss of individual species due to their nested and asymmetrical nature, which has evolved over time in response to variable climatic conditions (Memmott et al. 2007). However, climate changes beyond the range of historical variation may cause loss of even generalist plant and pollinator species, leading to pollination web simplification, which in turn, becomes more vulnerable to climatic changes (Memmott et al. 2007). Because populations of individual pollinator species can vary significantly from year to year and pollinator species vary in their preferred plant resources, sustaining pollination ecosystem services will likely rely on maintaining native pollinator biodiversity (Kremen et al. 2002b).

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Defining Terms

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

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

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

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

Vulnerability Assessment Model

The vulnerability assessment model applied in this process was developed by EcoAdapt (EcoAdapt 2014a; EcoAdapt 2014b; Kershner 2014; Hutto et al. 2015; Gregg 2018),[10] and includes evaluations of relative vulnerability by local and regional stakeholders who have detailed knowledge about and/or expertise in the ecology, management, and threats to focal habitats, species groups, individual species, and the ecosystem services that these resources provide. Stakeholders evaluated vulnerability for each resource by discussing and answering a series of questions for sensitivity and adaptive capacity. Exposure was evaluated by EcoAdapt using projected future climate changes from the scientific literature. Each vulnerability component (i.e., sensitivity, adaptive capacity, and exposure) was divided into specific elements. For example, habitats included three elements for assessing sensitivity and six elements for adaptive capacity. Elements for each vulnerability component are described in more detail below.

In-person workshops were held in Eureka, Redding, and Upper Lake between May and October 2017. Participants self-selected habitat and species group/species breakout groups and evaluated the vulnerability of each resource. Participants were first asked to describe the habitat and/or to list the species to be considered in the evaluation of an overarching species group. Due to limitations in workshop time and participant expertise, multiple resources were not assessed during these engagements. Evaluations for remaining habitats, species groups, and species were completed by contacting resource experts.[11]

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

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

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

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

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

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

Vulnerability Assessment Model Elements

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

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

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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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