Southern California Conifer Habitats

Climate Change Vulnerability Assessment Synthesis

An Important Note About this Document: *This document represents an initial evaluation of vulnerability for conifer habitats based on expert input and existing information. Specifically, the information presented below comprises habitat expert vulnerability assessment survey results and comments, peer-review comments and revisions, and relevant references from the literature. 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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Executive Summary

In southern California, conifer habitats occur across a wide elevational gradient, ranging from closed-cone pine and cypress stands on coastal bluffs to mixed conifer forests on high-elevation mountaintops. The spatial arrangement of conifer habitats is dynamic, and strongly influenced by climatic gradients, topography, latitude, distance from the coast and/or desert, disturbances, and understory composition (Grulke 2010; McKenzie et al. 2009; Minnich 2007). These habitats are typically dominated by coniferous species, although hardwoods may also be present in some habitat types. For the purposes of this section, the following California Wildlife Habitat Relationships System (CWHRS) groupings will be considered: montane hardwood-conifer (includes bigcone Douglas-fir [*Pseudotsuga macrocarpa*]; Anderson 1988), closed-cone pine-cypress (Jensen 1988), and ponderosa pine (Fitzhugh 1988).[1]

The relative vulnerability of conifer habitats in southern California was evaluated to be moderate[2] by habitat experts due to moderate sensitivity to climate and non-climate stressors, moderate-high exposure to future climate changes, and moderate adaptive capacity.

Sensitivity and Exposure	Climate sensitivities: Drought, precipitation, snowpack depth, soil moisture Disturbance regimes: Wildfire, insects, wind Non-climate sensitivities: Fire suppression practices, land use conversion

Conifer and mixed conifer habitats are sensitive to decreases in water availability, and moisture-stressed trees are more vulnerable to additional stressors, including insect outbreaks, disease, and pollution. Dead or dying trees can exacerbate wildfire by providing additional fuel, and wildfire may accelerate shifts in species composition by extirpating local populations, promoting fire-tolerant species or habitat types, and creating conditions suitable for invasive species. Conifer habitats are also sensitive to non-climate stressors (e.g., air pollution) that can exacerbate the impacts of climate change.

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[1] The vulnerability of subalpine forests and pinyon-juniper woodlands were assessed separately within this report.
[2] Confidence: High

Adaptive Capacity
Habitat extent, integrity, and continuity: Moderate geographic extent, low-moderate integrity (i.e., partially degraded), low continuity
Resistance and recovery: Low-moderate resistance potential, moderate recovery potential
Habitat diversity: Moderate-high overall diversity
Management potential: High societal value, low-moderate management potential

Southern California conifer habitat types vary in extent and continuity, including stands comprised of endemic species with very limited extent (e.g., Torrey pine [*Pinus torreyana*]) and large tracts of forest made up of species that can be found throughout the western U.S. (e.g., ponderosa pine [*P. ponderosa*]). Range shifts in montane forests are limited by elevation and the lack of connectivity between mountaintops, while lower-elevation forests and coastal stands are primarily limited by habitat fragmentation and human activity. The ability of conifer habitats to resist stressors and recover from disturbance varies depending on specific species requirements; however, recovery is often dependent on the rate of forest regeneration, which is generally slow (~40-60 years). Conifer habitats provide food and shelter for many wildlife species, and are highly valued for aesthetic and recreational purposes, as well as for the many ecosystem services that they provide (e.g., carbon sequestration, water supply, air quality, and flood/erosion protection).

Sensitivity

The overall sensitivity of conifer habitats to climate and non-climate stressors was evaluated to be moderate by habitat experts.[3]

Sensitivity to climate and climate-driven changes

Habitat experts evaluated conifer habitats to have moderate sensitivity to climate and climate-driven changes,[4] including: drought, precipitation, snowpack depth, and soil moisture.[5]

Precipitation, soil moisture, and drought

Soil moisture is the limiting factor in most southern California conifer habitats (Adams and Kolb 2005; McKenzie et al. 2001; Minnich 2007), and water stress is affected by a combination of other factors such as increased evapotranspiration rates due to warm temperatures (Williams et al. 2015). Where water stress is very high, the interaction of multiple stressors (e.g., drought, fire, insect outbreaks, and pollution) can contribute to broad-scale mortality events and shifts in species composition and/or distribution (McKenzie et al. 2009; Peterson et al. 1995).

Precipitation amounts in southern California fall on a rough west to east gradient, with coastal forests receiving more precipitation than inland areas (Peterson et al. 1995). Low clouds and fog often surround coastal forests, and these can provide shade, lower evapotranspiration

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[3] Confidence: High

[4] Confidence: High

[5] Factors presented are those ranked highest by habitat experts. A full list of evaluated factors can be found at the end of this document.

rates, and supplement soil moisture with dripping condensation (Fischer et al. 2008). Mycorrhizal communities may also mediate the effects of drought, and Mohan et al. (2014) determined that plants with mycorrhizal relationships were more productive under drought conditions. In addition, the rate of biogeochemical cycling in drought-affected ecosystems was improved with mycorrhizal inoculation (Mohan et al. 2014).

Shifts in the amount and timing of precipitation can affect growth and seedling survival (Chmura et al. 2011; Legras et al. 2010), and this can offset the potential benefit of warming temperatures to forest productivity (Hicke et al. 2002). For instance, bigcone Douglas-fir has reduced growth during drought periods and for several years afterward; in addition, trees with larger drought-related reductions in growth have larger long-term growth reductions (Peterson et al. 1995). Late spring rainfall extends the growing season by pushing off the beginning of summer drought, as long as it occurs after snowmelt occurs and the species' photoperiod requirements have been met (Royce and Barbour 2001). Growth can be further limited by competition for water resources, and dense forests may be more vulnerable to water stress (Bailey and Covington 2002; Grulke 2010; Royce and Barbour 2001; Weatherspoon et al. 1992).

During periods of drought, long-lived species (e.g., Jeffrey pine, bigcone Douglas-fir) have an advantage, as they are able to carry over more resources from year to year, buffering them from some environmental stress (Grulke 2003, Minnich 2007). Short-lived species (e.g., Coulter pine [*P. coulteri*]) are more vulnerable to the effects of drought, which tax reserves over an extended period of time (Minnich 2007). Drought sensitivity also changes along an elevational gradient, with trees at dry low-elevation sites exhibiting higher drought sensitivity than those at wetter high-elevation sites (Adams and Kolb 2005; Goforth 2009).

Ponderosa and Jeffrey pines are both adapted to water stress and periodic drought, although seedlings are more vulnerable to dry conditions within the first several years of growth (Grulke 2010; Legras et al. 2010; Scott et al. 2013). Ponderosa pine has greater stomatal control and water storage compared to species such as Douglas fir (*Pseudotsuga menziesii*), although it is vulnerable to xylem cavitation (Stout and Sala 2003). By contrast, sugar pine has low water use efficiency during drought periods (Grulke 2010) and sugar pine seedlings are less likely to survive drought than ponderosa pine seedlings (Legras et al. 2010). For Jeffrey pine, one statewide study found that branch and needle growth was highest at the study site with the lowest soil moisture (within the San Bernardino National Forest), which suggests that the species tolerates water stress well (Grulke 2010). Grulke (2010) also found that growth at the San Bernardino site was higher in dense stands, possibly because the canopy provided shade and increased humidity near the ground; this finding contrasts with the usual pattern of decreased growth in dense stands within montane conifer and mixed-conifer forests (Bailey and Covington 2002).

Most of the species that make up coastal forests, such as bishop pine (*P. muricata*), are quite sensitive to drought. Patterns of mortality in bishop pine suggest that drought-related mortality limits the species' range more than low recruitment rates, and that range and/or tree density are typically reduced following drought (Fischer et al. 2008). Higher rates of tree mortality occur on the inland and lower-elevation edges of the species' range (Fischer et al. 2008). However, shade, reduced evapotranspiration, and fog drip can reduce drought stress in this species by about 50% (Fischer et al. 2008).

Conifer habitats are particularly sensitive to multi-year drought events, both through direct mortality due to water stress and through drought-related facilitation of bark beetle attacks, pathogen infection, and increased fire injury (McKenzie et al. 2009; Minnich 2007). Between 2002 and 2004, a historic drought in southern California caused the widespread die-off of large-diameter conifers throughout the region (Minnich 2007). At the landscape scale, conifer mortality was highest on well-drained convex surfaces and south-facing slopes, which experience greater moisture stress, as well as in dense stands and/or areas that had not experienced recent fire (Minnich 2007). Prolonged fire return intervals also increased competition for soil moisture with chaparral and other understory shrubs (Minnich 2007). Large-diameter trees of short-lived species, such as Coulter pine, sustained whole-stand mortality from drought-facilitated bark beetle infestation across thousands of hectares in the San Bernardino and Hot Springs Mountains (Minnich 2007).

Snowpack depth and timing of snowmelt and runoff

Snowmelt contributes to soil moisture availability (Sheffield et al. 2004), and the timing of snowmelt is tied to the beginning of yearly conifer growth (Chmura et al. 2011). At higher-elevation sites, a shift towards earlier snowmelt is associated with earlier seasonal growth in conifers (Royce and Barbour 2001). At lower elevations, photoperiod requirements may not allow earlier growth and early snowmelt may extend the dry season by leaving trees dependent on spring rainfall for moisture (Royce and Barbour 2001; Westerling et al. 2006).

Air temperature

The sensitivity of southern California conifer habitats to air temperature is due in large part to increased evaporative demand and associated water stress (Adams and Kolb 2005; Grulke 2010; McKenzie et al. 2001; van Mantgem and Stephenson 2007). However, at higher elevations the balance begins to shift toward forests limited by energy (e.g., light and temperature) rather than water (Adams and Kolb 2005), and these sites have generally responded to warming temperatures over the last century with increased growth (McKenzie et al. 2001). Additional factors such as season, slope, aspect, and other microsite conditions can also alter the influence of temperature on conifers (Fritts 1976). Little research has been done on the impact of heat waves on conifer habitats, but it is likely that similar patterns would exist.

Among southern California conifers, ponderosa pine is particularly well-adapted to heat, though seedlings are susceptible to cold nighttime temperatures (Scott et al. 2013). Ponderosa pine often becomes established in hot, dry conditions on disturbed sites, eventually creating shade in which species less tolerant of heat can become established (Bollenbacher et al. 2014). Jeffrey pine (*P. jeffreyi*) can also tolerate warm temperatures, and Grulke (2010) found that Jeffrey pine needle growth was highest at the southernmost and lowest-elevation site across a latitudinal gradient in California.

Conifers are also sensitive to climate changes that may alter phenology, which are driven in part by increasing air temperatures. For example, warm temperatures would likely lengthen the growing season, increasing the opportunity for tree growth (Royce and Barbour 2001; Chmura et al. 2011). The timing of bud burst could also shift, depending on the magnitude and timing of temperature increases, whether winter temperatures meet the chilling requirement for seeds, and whether the species is early- or late-leafing (Chmura et al. 2011; Morin et al. 2009). When winters are cold and spring temperatures are warm, bud burst could shift towards an earlier date; in the absence of this weather trend, bud burst could be significantly delayed (Chmura et al. 2011; Morin et al. 2009). Trees are also vulnerable to frost damage when spring temperatures fluctuate widely (Chmura et al. 2011).

Increased temperatures indirectly affect many other processes by exacerbating additional stressors such as drought, insect outbreaks, and wildfire, all of which may increase tree mortality (McKenzie et al. 2009). For example, warming temperatures are associated with increased wildfire size and frequency (Westerling et al. 2006), and may change the timing and/or synchronicity of bark beetle life cycles (Bentz et al. 2010; McKenzie et al. 2009).

Low stream flows

Streamflow is strongly influenced by evapotranspiration rates, which vary seasonally in response to factors such as temperature and precipitation (Franco-Vizcaino et al. 2002; Minnich 2007). As the water deficit increases over the course of the summer, the relationship between precipitation and evapotranspiration becomes stronger and progressively less water runs off the forest floor into streams (Franco-Vizcaino et al. 2002). By late summer, most streams dry up in their middle reaches, except where groundwater sources are close to the surface (Stephenson and Calcarone 1999).

Sensitivity to disturbance regimes

Habitat experts evaluated conifer habitats to have moderate-high sensitivity to disturbance regimes,[6] including: wildfire, insects, and wind.[7] Habitat experts also identified disease as an additional factor that may impact conifer habitats. Although some higher-elevation streams can flood during large precipitation events, which in turn can trigger mudslides, debris flows, and erosion (Carpenter et al. 2007), flood damage is a fairly unusual form of disturbance in conifer habitats (A. Anderson, pers. comm., 2015). Overall, the response of conifer habitats to disturbance events varies widely depending on species composition and ecotype (e.g., mesic, xeric; J. Hooper, pers. comm., 2015).

Wildfire

Wildfire is one of the primary drivers of natural succession in conifer habitats, and all habitat types are adapted to some level of fire. However, fire regimes vary widely among habitats, as do individual species' sensitivity to fire (Minnich 2007; Steel et al. 2015; Stephenson and

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[6] Confidence: High

[7] Factors presented are those ranked highest by habitat experts. For a complete list of scoring, see the end of this document.

Calcarone 1999). Changes in historical fire regimes (e.g., altered size, severity, and/or frequency) can have widespread consequences, causing shifts in species composition, stand extirpation, and type conversion (de Gouvenain and Ansary 2006; Enright et al. 2015; Franklin et al. 2006). Since the 1980s, factors such as warming temperatures, drought, and earlier snowmelt have contributed to increases in wildfire size and duration, primarily by increasing the supply of dead fuel and reducing fuel moisture content (Goforth 2009; Keeley and Syphard 2015; Keeley and Zedler 2009; Westerling et al. 2006). Invasive species can also drive altered fire regimes by changing fuel loads, structure, and continuity (Brooks et al. 2004).

Severe fires may extirpate stands at a rate that exceeds their regeneration capacity (Franklin et al. 2006; Minnich 2007); this is often associated with high fire intensity (which may destroy seed banks), low soil moisture, and few remaining unburned patches (Franklin et al. 2006; Goforth 2009). In addition, loss of vegetative cover contributes to high levels of post-fire erosion and associated loss of topsoil, creating conditions that are less suitable for seedling recruitment (Vulnerability Assessment Reviewers, pers. comm., 2015). An increase in fire severity would favor fire-adapted species such as oaks, which resprout vigorously in open spaces (Franklin et al. 2006; Weatherspoon et al. 1992). Shade-tolerant species are usually sensitive to fire (e.g., incense cedar [*Libocedrus decurrens*]), and are likely to decline in areas that experience frequent or severe fires (Weatherspoon et al. 1992).

Closed-cone pine and cypress

Closed-cone pine and cypress stands are typically surrounded by chaparral, and are adapted to relatively high-severity fire (Minnich 2007). Stands may have a heavy shrub understory that can easily catch and carry fire, and trees are generally small with thin and/or flaky bark; all of these factors contribute to high rates of fire-related tree mortality (Minnich 2007). Closed-cone pine and cypress species typically reach maturity within 10-20 years, and they are dependent on the heat of fire to open and disperse seeds from serotinous or semi-serotinous cones (Minnich 2007). In semi-serotinous species, seed may be periodically released without fire, but most species are shade-intolerant, and seedling recruitment is typically low in established stands with no open areas (Minnich 2007). Most regeneration is pulsed and occurs after a fire, when the disturbed area is free of shade, competition is low, and the soil has been mineralized with ash (de Gouvenain and Ansary 2006; Minnich 2007). For instance, natural regeneration of Coulter pine is relatively low, but seedling regeneration since the 2009 Station Fire has been prolific in burned areas, despite sustained drought conditions (J. Hooper, pers. comm., 2015).

The fire regime for closed-cone pine and cypress stands typically reflects that of the surrounding chaparral, which is adapted to regenerate after stand-replacing fires every few decades (Lombardo et al. 2009; Minnich 2007). Closed-cone pine and cypress species are sensitive to fire return intervals, as fires that occur too frequently may threaten stand regeneration by killing trees before they are able to mature and produce a seed crop (Ne'eman et al. 1999; Enright et al. 2015). Under these conditions, habitat conversion to chaparral or oak woodlands could occur (Stephenson and Calcarone 1999). A study on Tecate cypress (*Hesperocyparis forbesii*) found that populations decline when fire occurs at a return interval of less than 40 years, although it is likely that microsite differences play a part in protecting Tecate cypress stands from more frequent chaparral fires (de Gouvenain and Ansary 2006). Sargent cypress (*Hesperocyparis sargentii*) matures quickly and accumulates a substantial seed bank by 20 years of age, making it more resilient to fire; this species would likely be unaffected by frequent fires unless they occurred at intervals of less than 20 years (Ne'eman et al. 1999). Very long return intervals can also negatively affect stand regeneration, as long-term fire exclusion may prevent seed dispersal within the lifetime of the seed bank (Ne'eman et al. 1999).

Bigcone Douglas-fir

Bigcone Douglas-fir stands are typically found near streams in canyons and on steep north-facing slopes, where they are somewhat protected from fire (Stephenson and Calcarone 1999). Relatively infrequent fires characterize the fire regime for bigcone Douglas-fir forest, which is similar to that of the chaparral often surrounding the stands (Lombardo et al. 2009). Bigcone Douglas-fir trees have very thick bark and can re-sprout new branches and needles from their trunk, so adult trees typically survive fire (Lombardo et al. 2009). Frequent fires may negatively impact seedling regeneration, though, because a decline in nurse plants for this species (e.g., Eastwood manzanita [*Arctostaphylos glandulosa*]) prevents the slow-growing trees from reaching maturity (Vulnerability Assessment Reviewers, pers. comm., 2015).

Over the last hundred years, fires in bigcone Douglas-fir stands have become larger and more severe (Lombardo et al. 2009; Steel et al. 2015), and this shift is correlated with a decrease in fire frequency, suggesting that increased fuel availability may be tied to fire severity in this habitat (Steel et al. 2015). Bigcone Douglas-fir habitat is sensitive to stand-replacing fire because it is slow to recover from such fires; however, the rate of stand regeneration varies across ecotypes. For instance, bigcone Douglas-fir stands on mesic sites (associated with California bay [*Umbellularia californica*] or other riparian species) are recovering from the 2009 Station Fire (J. Hooper, pers. comm., 2015). In xeric sites, bigcone Douglas-fir stands (associated with canyon live oak [*Quercus chrysolepis*]) are not regenerating well, although some pure stands in cooler microsites appear to have an advantage (J. Hooper, pers. comm., 2015).

Montane conifer and mixed-conifer forests

The fire regime in montane conifer and mixed-conifer forests is typically characterized by long fire return intervals (Weatherspoon et al. 1992). Historically, fire intensity was mixed, with low-intensity surface fires causing high mortality in seedlings and young trees but leaving older trees unharmed (Bailey and Covington 2002; Bollenbacher et al. 2014; Goforth 2009). Fires such as these kept the understory relatively open, minimizing fuel accumulation that would contribute to higher-intensity fires (Goforth 2009; Weatherspoon et al. 1992). More severe fires did occur (Keeley and Zedler 2009), and have increased over the past several decades (Steel et al. 2015; Westerling et al. 2006).

Less frequent fires in forests dominated by species such as ponderosa and Jeffrey pine have likely contributed to the development of very dense stands (Goforth 2009). At Cuyamaca Mountain, conifer density doubled between 1928 and 1995, and increased density appears to drive larger, more severe fires because of the additional fuel available (Goforth 2009). Shade-tolerant species (e.g., incense cedar) have become more common in these habitat types, and regeneration of shade-intolerant species such as ponderosa pine and black oak (*Quercus velutina*) has decreased, except in areas that have been recently burned or are managed by thinning (Goforth 2009; Stephenson and Calcarone 1999; Weatherspoon et al. 1992).

Ponderosa pine is somewhat resistant to fire because of its thick bark, which typically allows adult trees to survive (Bollenbacher et al. 2014). Among ponderosa seedlings, fire mortality is high; Bailey and Covington (2002) found that trees over 2 m in height and greater than 5 cm in diameter usually survived a surface fire, even though their trunks were scorched. Ponderosa pine is also shade-intolerant, and burning seems to enhance seedling recruitment (Bailey and Covington 2002). In areas where competing tree species are less tolerant of fire, ponderosa pine stands may expand following a burn (Scott et al. 2013).

Wind

Autumn winds, including hot Santa Ana winds that blow inland from the coast, are associated with large, fast-spreading fires in southern California (Keeley 2004). In these cases, wind likely plays a role in determining fire size, rate of spread, and intensity (Minnich 2007). Keeley (2004) found that Santa Ana winds predicted autumn fire better than temperature, precipitation, or drought. Power lines downed by wind can ignite fires; within San Diego County, efforts are underway to enable the creation of a monitoring system that would prompt shutdown when winds exceed a certain threshold (J. Hooper, pers. comm., 2015).

Wind may contribute to desiccation, especially on exposed high-elevation sites and granite outcrops (Fischer et al. 2008; Minnich 2007); very strong wind can even damage or uproot mature conifers (M. Wells, pers. comm., 2015). In coastal forests, strong winds can cause closed-cone pine and cypress trees to become stunted, especially where soils are shallow or infertile (Jensen 1988). Coastal wind plays a role in the capture of condensed fog, and may push fog and stratus clouds inland for up to 20 km (Fischer et al. 2008; Minnich 2007). However, winds blowing inland from highly developed coastal areas can also carry air pollutants across the state (Fenn et al. 2010; Peterson et al. 1995).

Insects

Widespread mortality events have occurred over the last several decades due to bark beetle outbreaks (e.g., *Dendroctonus* spp. and *Ips* spp.; Bentz et al. 2010). The severity and large scale of these outbreaks are likely due, in part, to other factors that increase a tree's vulnerability to attack, such as drought and air pollution (Allen and Breshears 1998; Breshears et al. 2005; van Mantgem and Stephenson 2007). These factors weaken the ability of the tree to defend itself against attack, and decrease the number of beetles required to cause an outbreak (Herms and Mattson 1992; Bentz et al. 2010).

Insect population success is directly linked to seasonal temperature (Régnière et al. 2012), which can alter developmental stages, synchronicity, and mortality of the bark beetles that affect southern California conifer species (Bentz et al. 2010). In response to warming temperatures, forest pests may be capable of rapid genetic or phenotypic adaptation and range expansion (Régnière et al. 2012; Bentz et al. 2010). Conversely, cold temperatures can inhibit reproduction and cause mortality (Bentz et al. 2010; M. Wells, pers. comm., 2015).

Ponderosa pine, Jeffrey pine, and Coulter pine are particularly vulnerable to beetle attack from multiple insect species (Minnich 2007). Oaks, which make up a large component of mixed-conifer forests, are also vulnerable, and the goldspotted oak borer (*Agrilus auroguttatus*) was recently discovered on Palomar Mountain (A. Anderson and D. Faulkner, pers. comm., 2015).

Recovery from an insect outbreak can be very slow; regenerating ponderosa pine can regain pre-attack basal area within 10-80 years, but in the absence of adequate seedling recruitment, return to pre-attack basal area could take up to 200 years (Briggs et al. 2015). Dead and dying trees may provide additional fuel for wildfire (McKenzie et al. 2009; Briggs et al. 2015), and in ponderosa pine stands surface fuels remain high for 10-60 years after attack (Briggs et al. 2015). In the Sierra Nevada, mortality from Jeffrey pine beetles (*Dendroctonus jeffreyi*) was highest in dry microsites, suggesting that variation in microsite characteristics may lower the sensitivity of trees to beetle outbreaks (Grulke 2010).

Disease

Conifer habitats are vulnerable to a number of diseases, including annosus root disease (*Heterobisidion annosum*), which has caused mortality in Jeffrey pine and mixed conifer forests in the San Bernardino and San Gabriel Mountains, and armillaria root disease (*Armillaria spp.*), which is transferred from dead or dying oaks to young conifer trees nearby (Minnich 2007; Stephenson and Calcarone 1999). Sugar pine (*P. lambertiana*) can be affected by white pine blister rust, which causes heavy damage and tree mortality (Stephenson and Calcarone 1999); blister rust has already caused considerable damage in central coastal California (M. Wells, pers. comm., 2015). Finally, pitch canker fungus (*Fusarium subglutinans*), a pathogen introduced from the southeast U.S., affects closed-cone pines, including knobcone pine (*P. attenuata*), bishop pine, and Monterey pine (*P. radiata*; Stephenson and Calcarone 1999).

Changes in climate conditions and additional stressors (e.g., temperature, soil moisture and drought, wildfire and/or fire exclusion, insect outbreaks, pollution) have the potential to affect interactions between disease organisms and host trees (Chmura et al. 2011; Grulke 2010; Sturrock et al. 2011). Grulke (2010) demonstrated that the incidence of disease may be higher in dense stands, finding that, in the eastern Sierra Nevada, the frequency of needle scale was two and a half times greater in dense stands of Jeffrey pine than in thinned stands. Grulke (2010) also found that the frequency of needle scale was over four times higher on dry sites.

Parasitic dwarf mistletoes (*Arceuthobium* spp.) use water and nutrients produced by the host tree and are common in southern California conifer forests, affecting Jeffrey pine, ponderosa pine, knobcone pine, and Coulter pine (Minnich 2007; Stephenson and Calcarone 1999). True mistletoe (*Phoradendron* spp.) affects conifer habitats in the region as well, impacting incense cedar and hardwoods. However, true mistletoes are able to photosynthesize, and take fewer resources from the host tree (Stephenson and Calcarone 1999). Both dwarf and true mistletoes provide food for wildlife, and dead branches provide habitat for species such as cavity-nesters (Stephenson and Calcarone 1999).

Sensitivity and current exposure to non-climate stressors

Habitat experts evaluated conifer habitats to have moderate sensitivity to non-climate stressors[8] and to have an overall low-moderate exposure to these stressors within the study region.[9] Key non-climate stressors identified by habitat experts for conifer habitats include: fire suppression and land-use conversion.[10] The scientific literature also suggests that invasive species (Brooks et al. 2004; Stephenson and Calcarone 1999), pollution (Fenn et al. 2008, 2010; Grulke 2003), and grazing (Minnich 2007; Stephenson and Calcarone 1999) act as additional non-climate stressors for conifer forests. Southern California forests have historically been exploited for timber and mining exploration, but these factors were limited by the relative isolation of conifer habitats and by the discovery of gold in the late 1800s and early 1900s, which decreased use of the forest for timber or fuel (Minnich 2007).

Fire suppression

Anthropogenic activity altered historical fire regimes beginning in the 1800s, and fire suppression practices were adopted in the early 1900s (Weatherspoon et al. 1992). In some cases, these practices include the direct removal of conifer communities adjacent to the urban interface in order to create fuelbreaks, affecting primarily low-elevation drainages (J. Hooper, pers. comm., 2015). While not all researchers agree on the effect of anthropogenic activity on fire regimes (Keeley and Zedler 2009), many studies conclude that fire exclusion has affected fire size and intensity (McKenzie et al. 2009; Sakulich and Taylor 2007; Steel et al. 2015; Stephenson and Calcarone 1999), and has caused changes in forest structure and composition (McKenzie et al. 2009; Sakulich and Taylor 2007; Scott et al. 2013; Stephenson and Calcarone 1999). Habitats with short fire return intervals have likely been most affected because the number of altered cycles is higher than in habitats with longer fire intervals (Weatherspoon et al. 1992).

Within many conifer forests, fire exclusion has led to increased fuel-loading and stand densification (Bailey and Covington 2002; Minnich 2007), which may contribute to the larger, longer, and more intense fires observed in the late 1900s and early 2000s (Minnich 2007). For instance, fire exclusion in ponderosa pine stands has led to very dense stands with no openings (Bollenbacher et al. 2014), resulting in low seedling recruitment (Bailey and Covington 2002) and increased numbers of shade-tolerant, fire-sensitive species (Weatherspoon et al. 1992). In closed-cone pine and cypress forests, fire exclusion may lead to a lack of forest regeneration, as serotinous cones are not able to disperse seeds in the absence of fire (Ne’eman et al. 1999).

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[8] Confidence: High

[9] Confidence: High

[10] Factors presented are those ranked highest by habitat experts. For a complete list of scoring, see the end of this document.

Land-use conversion

Many high-elevation conifer habitats are relatively protected from development by their lack of accessibility and/or by inclusion within state parks and national forests. However, lower-elevation habitats are impacted more heavily by increasing urban development and agriculture, as well as the infrastructure associated with these activities (e.g., roads and highways, water delivery systems; Stephenson and Calcarone 1999). For instance, transportation corridors fragment conifer communities, inhibiting dispersal and the natural movement of water and nutrients towards downslope areas (J. Hooper, pers. comm., 2015). Roads provide access to forests, and are linked to increased timber harvest, recreational use, invasive species, and fire ignitions (Fites-Kaufman et al. 2007; Stephenson and Calcarone 1999; Syphard and Keeley 2015).

Invasive species

Invasive and non-native species compete for resources (e.g., space, water, nutrients) with native species (Stephenson and Calcarone 1999), and invasive vegetation can inhibit natural habitat regeneration that would otherwise occur in the absence of competition for scarce resources (J. Hooper, pers. comm., 2015). The pressures associated with invasive species can also reduce the size of native populations and increase the risk of extirpation or extinction (A. Anderson, pers. comm., 2015; Stephenson and Calcarone 2015). Non-native predators, such as the seven-spot ladybird beetle, can reach high numbers, consuming native prey and outcompeting native predators (A. Anderson, pers. comm., 2015).

Mediterranean grasses make up the largest group of invasive plants, and readily spread in disturbed areas, including recreation sites, transportation corridors, and recently burned areas (Stephenson and Calcarone 1999). However, grasses can also invade undisturbed areas (Keeley and Brennan 2012; Stephenson and Calcarone 1999), aided in some cases by nitrogen deposition, which spurs growth (Fenn et al. 2010). Invasive and non-native herbaceous vegetation can also contribute to altered fire regimes (Brooks et al. 2004). An increase in Mediterranean grasses can create greater fuel loads and increase fuel continuity, potentially leading to more frequent and/or more severe fires (Brooks et al. 2004). Fire ignitions occur more frequently in grass than in most other habitat types (Syphard and Keeley 2015).

Pollution

Conifer habitats are vulnerable to air pollution, and the primary pollutants of concern are nitrogen and ozone (Fenn et al. 2008; Grulke 2003). Sulfur deposition is much less widespread; however, it can also occur and causes soil acidification (Fenn et al. 2010). Deposition of pollutants occurs on a rough west to east gradient in southern California, originating in highly developed coastal areas and drifting east (Fenn et al. 2010; Peterson et al. 1995). Because oxidants and nitrates are transported by wind, trees at lower elevations and on windward slopes or exposed ridges are at the highest risk of being affected by air pollution, especially those near population centers (Minnich 2007).

Ozone can affect conifers by lowering their rate of photosynthesis and altering resource acquisition and allocation (Grulke 2003). Ozone injury is visible first as chlorotic mottling on older needles, then as necrosis and needle abscission (Miller et al. 1983). Peterson et al. (1995) found that bigcone Douglas-fir in high ozone areas retained fewer needles and had larger decreases in growth compared to trees in low or moderate ozone areas. In a study comparing 11 native conifer species, ponderosa pine and Jeffrey pine were most sensitive to ozone, followed in order from most to least sensitive by Coulter pine, incense cedar, bigcone Douglas-fir, and sugar pine (Miller et al. 1983). In sugar pine and incense cedar, two of the most tolerant species, a large proportion of seedlings had only slight injuries after ozone fumigation treatment (62% and 47%, respectively; Miller et al. 1983).

Although low levels of nitrogen input can benefit herbaceous native and invasive plant productivity (Allen et al. 2009), ongoing or high levels of nitrogen deposition can lead to changes in carbohydrate reserves, root biomass, litter accumulation, and the community composition of grasses, forbs, and lichens (Allen et al. 2009; Eatough Jones et al. 2004; Fenn et al. 2008; Fenn et al. 2010; Grulke et al. 2009). Within mixed conifer forests, nitrogen deposition loads in excess of 3.1-5.2 kg per ha/year causes the concentration of nitrogen to increase in lichen tissue, triggering changes in community composition (Fenn et al. 2008; Fenn et al. 2010). Critical loads in excess of 17 kg per ha/year cause nitrates to leach into streams and reduce fine root biomass of conifers by up to 26%. Within the state of California, 28.7% of mixed conifer forests are estimated to receive over 3-8 kg per ha/year, above the level at which negative effects on the forest have been reported (Fenn et al. 2010).

Air pollution is frequently part of stress complexes that affect conifer habitats, and can result in increased tree mortality, stand densification, and increased risk of disease, insect attack, drought stress, and wildfire (Eatough Jones et al. 2004; Grulke et al. 2009; Grulke 2010; McKenzie et al. 2009). For instance, Grulke (2010) found that sites with high nitrogen and ozone deposition had higher levels of needle scale and Jeffrey pine beetle-related mortality. Physiological stress caused by pollution decreases the tree's water uptake efficiency and increases needle senescence, with the overall effect of reducing the tree's capacity to tolerate drought (Grulke 2003) and resist beetle attack (Eatough Jones et al. 2004; Grulke et al. 2009).

Grazing

Historically, livestock grazing was relatively limited, and most exploitative forest uses ended with the establishment of the region's national forests in 1910 (Minnich 2007). Grazing still occurs within some meadows surrounded by conifer forest, and in nearby open areas (Stephenson and Calcarone 1999); these include Crouch Valley in the Laguna Mountains and the Upper French and Dyche Valleys on Palomar Mountain (A. Anderson, pers. comm., 2015). Grazing that is not managed for ecosystem health increases drought stress on vegetation, reduces populations of native plants and wildlife, spreads invasive species, and increases erosion during storms (A. Anderson, pers. comm., 2015; Stephenson and Calcarone 1999). However, well-managed grazing may benefit the system by maintaining areas of the forest in early successional stages. Most dams and water diversions in grazing areas are cattle ponds, and these do not seem to have a negative effect on the habitat; in fact, they may enhance water availability to wildlife during drought. (A. Anderson, pers. comm., 2015).

Future Climate Exposure

Habitat experts evaluated conifer habitats to have moderate-high exposure to future climate and climate-driven changes,[11] and key climate variables to consider include: changes in precipitation, decreased soil moisture, increased drought, increased wildfire, earlier snowmelt/runoff, and increased extreme heat events (Table 1).[12] For a detailed overview of how these factors are projected to change in the future, please see the Southern California Climate Overview (http://ecoadapt.org/programs/adaptation-consultations/socal).

Temperature refugia for conifer habitats may be located on north- and east-facing slopes, near seeps and springs, and in swales and canyons (M. Wells, pers. comm., 2015; Flint et al. 2013); for instance, canyons on steep, eroded slopes can act as refugia for bigcone Douglas-fir (N. Sugihara, pers. comm., 2015; Minnich 2007). Rocky sites may also be able to serve as fire refugia for conifer species (e.g., bigcone Douglas-fir, Jeffrey pine), as they contain little available fuel (N. Sugihara, pers. comm., 2015). A study of New Mexico ponderosa pine forests also suggests that fire refugia may be found on sites that are at a higher elevation relative to the surrounding area, and/or on gentle southeastern slopes (Haire, Coop, and Miller, unpublished data). Near the San Andreas Fault, highly variable terrain may offer microsites with protection from hot or dry conditions elsewhere (J. Hooper, pers., comm., 2015).

Table 1. Anticipated conifer forest response to climate and climate-driven changes.

Climate and climate-driven changes	Anticipated conifer habitat response
Precipitation, soil moisture, and drought *Variable annual precipitation volume and timing; decreased soil moisture; longer, more severe droughts with drought years twice as likely to occur*	• Decreased growth and seedling survival • Increased competition for water resources, especially in dense stands • Reduced forest extent and/or shift in species composition • Widespread mortality of conifers, especially when other stressors are present (e.g., drought, pollution)
Air temperature and extreme heat events *+2.5 to +9°C by 2100; heat waves will occur more frequently, last longer, and feature hotter temperatures*	• Longer growing season • Increased growth in high-elevation sites typically limited by temperature • Increased evapotranspiration, resulting in lower soil moisture • Changes in phenology (e.g., timing of bud burst)
Snowpack depth and timing of snowmelt/runoff *Up to 70% reduction in snowpack by 2100; snowmelt occurring 1-3 weeks earlier; increased runoff and earlier peak flows*	• Reduced soil moisture, resulting in longer summer drought periods • Increased dependency on spring precipitation events • Earlier start to growing season where snowmelt typically limits length of season • Increased flooding and erosion during peak runoff events

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[11] Confidence: High

[12] Factors presented are those ranked highest by habitat experts. For a complete list of scoring, see the end of this document.

Streamflow *Lower and longer duration of summer low flows*	• Reduced soil moisture and increased summer water deficit • Increased drying of middle stream reaches, with some perennial streams becoming ephemeral
Wildfire *Increased fire size, frequency, and severity*	• Increased stand density where fires are less frequent • Shift in species composition of conifer habitats toward fire-tolerant species (e.g., oaks) • Stand extirpation where fire severity and/or frequency does not allow regeneration, or where fire regime requirements are not met (e.g., cypress stands) • Reduced forest extent and possible type conversion
Insects *Range expansion, increased number of generation, and/or changes in the timing of development*	• Widespread mortality, especially where additional factors increase vulnerability to attack (e.g., drought, pollution) • Increased susceptibility to other stressors during recovery period after attack • Increased availability of dead wood as fuel

Under both wetter and drier future scenarios, climatic water deficit is expected to increase, with the mountain watersheds of southwestern California (e.g., the northern San Bernardino Mountains and the Big Bear Lake region) likely to experience the greatest changes in the region (Thorne et al. 2015). Within San Diego County, Palomar Mountain has the wettest conifer forest sites, with a large area of meadows and valleys that retain groundwater (A. Anderson, pers. com., 2015). Within the Palomar Mountain region, the central Upper French Valley, western and central Mendenhall Valley, north-facing slopes above central Mendenhall Valley, and Jeff Valley will likely remain wet into the future (A. Anderson, pers. comm., 2015).

Conifer habitats are expected to shift towards higher elevations, with an overall loss of habitat in the southwestern United States. (Friggens et al. 2012; Kelly and Goulden 2008). Between 1977 and 2007, the mean elevation of Jeffrey pine shifted upward by 28 m; this change has been attributed to climate rather than simple response to stressors like drought or insect outbreaks (Kelly and Goulden 2008). It is likely that changes in species composition will account for the majority of future habitat shifts (Kelly and Goulden 2008), and mixed conifer forest may gradually displace pure conifer forests due to a projected increase in the productivity of hardwoods (Lenihan et al. 2008). Species range shifts are likely to occur primarily through mortality in lower range sites during dry years, and species already at the southern edge of their range are particularly likely to experience range contraction (Kelly and Goulden 2008).

Adaptive Capacity

The overall adaptive capacity of conifer habitats was evaluated to be moderate by habitat experts.[13]

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[13] Confidence: High

Habitat extent, integrity, continuity, and landscape permeability

Habitat experts evaluated conifer habitats to have a moderate geographic extent (i.e., habitat occurs across the study region),[14] low-moderate integrity (i.e., habitat is partially degraded),[15] and low continuity (i.e., habitat is isolated and/or quite fragmented).[16] Habitat experts identified land-use conversion and geologic features as key barriers to habitat continuity and dispersal.[17]

Many types of conifer habitats occur across multiple ecosystems (e.g., coastal, montane) in southern California, and forest extent, integrity, and continuity vary widely within these habitat types (N. Sugihara, pers. comm., 2015). In some cases, geographic extent is limited by very small species ranges (such as the endemic Cuyamaca cypress [*Hesperocyparis stephensonii*]), but many conifer species are widespread throughout the western United States (e.g., ponderosa pine; J. Hooper, pers. comm., 2015). In montane forests, habitat extent and future shifts are restricted both by elevation and by the lack of connectivity between mountaintops, while lower-elevation and coastal forests are more likely to be limited by development, agriculture, and transportation infrastructure (Vulnerability Assessment Reviewers, pers. comm., 2015). Some habitat types are limited by site requirements; for instance, bigcone Douglas-fir forests may be especially fragmented due to their reliance on steep, eroded slopes to facilitate establishment (Minnich 2007).

Resistance and recovery

Habitat experts evaluated conifer habitats to have low-moderate resistance to climate stressors and maladaptive human responses,[18] and moderate recovery potential.[19] The ability of conifer habitat types to resist and recover from stressors varies, with some habitat types faring much better than others (N. Sugihara, pers. comm., 2015). However, in most conifer habitats full recovery from a disturbance event can take 40-60 years (M. Wells, pers. comm., 2015).

Recovery is largely dependent on the rate of forest regeneration, which depends on seed availability, germination rate, growing conditions, and survival of seedlings and saplings. Sufficient seed for ponderosa pine regeneration may occur only 1-2 times per decade, and factors such as competition for resources severely limit seedling survival (Bailey and Covington 2002). However, under normal conditions ponderosa pine regeneration rates are high enough to sustain multi-aged stands, assuming two good seed years and two fires per decade (Bailey and Covington 2002).

Phenotypic plasticity is also important factor when considering resistance and recovery in long-lived tree species (Grulke 2010). In a study comparing four conifer species across the state of

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[14] Confidence: High

[15] Confidence: Moderate

[16] Confidence: High

[17] Barriers presented are those ranked most critical by habitat experts (not all habitat experts agreed on these landscape barriers). A full list of evaluated barriers can be found at the end of this document.

[18] Confidence: High

[19] Confidence: Moderate

California and into the northern Baja peninsula, Jeffrey pine and white fir trees displayed the greatest amount of ecophysiological variability, while sugar pine consistently displayed the least variation. Inconsistent drought stress may contribute to the maintenance of phenotypic plasticity, especially in long-lived species (Grulke 2010). However, widespread mortality events, such as drought and beetle outbreaks in Jeffrey and ponderosa pine, have likely decreased population variability, affecting the ability of these species to adapt to future change (Grulke 2010).

Finally, inter-species associations with underground fungal and bacterial communities are an important aspect of adaptation in conifer habitats (J. Hooper, pers. comm., 2015). Post-fire pioneer shrubs often establish fungal communities that help to stabilize soil and acquire nutrients following a wildfire (Claridge et al. 2009), and they can help trees withstand drought and other stressors (Hernández-Rodríguez et al. 2013).

Habitat diversity

Habitat experts evaluated conifer habitats to have moderate-high physical and topographical diversity,[20] moderate-high component species diversity,[21] and moderate-high functional group diversity.[22] As a whole, southern California conifer habitats are diverse and host many endemic tree species, including bigcone Douglas-fir, Cuyamaca cypress, and Tecate cypress (Stephenson and Calcarone 1999). Broadly defined, conifer habitats are not dependent on any single species; however, specific types of conifer habitat, such as Torrey pine forest, may have one or more foundational species that may be vulnerable to climate and non-climate stressors (N. Sugihara, pers. comm., 2015).

In species with very small and/or isolated populations, genetic diversity is likely to be low and multiple stressors (or a single large-scale event) could threaten survival of the stand and/or species. For example, Torrey pine occurs in only two populations, one in San Diego and the other on Santa Rosa Island (Ledig and Conkle 1983; M. Wells, pers. comm., 2015). These populations exhibit extremely low genetic diversity. Chromosomal loci are homozygous within each population, and the difference between loci in the two populations is only 8.5% (Ledig and Conkle 1983). Although both populations are on protected land, the species itself does not have a Federal protected status (M. Wells, pers. comm., 2015). Recently, the San Diego population has been facing the additional stress of attacks by the five-spined engraver beetle (*Ips paraconfusus*; M. Wells, pers. comm., 2015).

Conifer habitats support wildlife, including many species that are vulnerable to climate change. Some of the vulnerable wildlife species that depend on conifer habitats include the Laguna Mountains skipper (*Pyrgus ruralis lagunae*), Hilda greenish blue (*Plebejus saepiolus hilda*), banana slugs (*Ariolimax* spp.), mountain yellow-legged frog (*Rana muscosa*), San Bernardino flying squirrel (*Glaucomys sabrinus californicus*), southern California steelhead (*Oncorhynchus*

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[20] Confidence: High
[21] Confidence: High
[22] Confidence: High

mykiss), spotted owl (*Strix occidentalis*), and California condor (*Gymnogyps californianus*; McDonnell et al. 2009; U.S. Fish and Wildlife Service 2015). Overall, insects, amphibians, and reptiles are particularly susceptible to climate change because they are exothermic and many have small isolated populations (A. Anderson, pers. comm., 2015). Likewise, many rare plant and wildlife species are restricted to mountaintops in areas such as the San Jacinto, Palomar, Cuyamaca, and Laguna Mountains, and the potential for range shifts in these species is limited by a lack of higher-elevation habitat (A. Anderson, pers. comm., 2015).

Montane hardwood-conifer

Montane hardwood-conifer forests include a mixture of at least one-third conifer trees and one-third hardwood trees (Anderson 1988). These forests are characterized by a bi-layered canopy structure in which conifers constitute the upper canopy (≤ 65 m) and broad-leaved trees make up the lower canopy (10-30 m) with relatively little understory (Anderson 1988). Although species composition is highly variable in this widespread forest type, common associates for montane hardwood-conifer forests in southern California include ponderosa pine, Jeffrey pine, sugar pine, incense cedar, bigcone Douglas-fir, canyon live oak, coast live oak (*Q. agrifolia*), black oak, big leaf maple (*Acer macrophyllum*), Pacific madrone (*Arbutus menziesii*), tanoak (*Notholithocarpus densiflorus*), and coast redwood (*Sequoia sempervirens*; Anderson 1988).

Groves of bigcone Douglas-fir, a species endemic to southern California, are included within the montane hardwood-conifer forest type (Anderson 1988). These are often found in deep, convex canyons and steep north-facing slopes (Minnich 2007). Stands are usually surrounded by chaparral at lower elevations, and by conifer or mixed-conifer forests at higher elevations (Minnich 2007). Within these groves, bigcone Douglas-fir trees form the upper canopy (30-50 m), and canyon live oak makes up the lower canopy (10-25 m); other hardwood tree associates can include coast live oak and California bay (Minnich 2007). Occasionally a sparse understory of chaparral is present, consisting of *Ceanothus* spp., *Cercocarpus* spp., California buckthorn (*Frangula californica*), and poison oak (*Toxicodendron diversilobum*; Minnich 2007).

Ponderosa pine

Ponderosa pine forests are characterized by a canopy area of at least 50% ponderosa pine, and the structure can vary from open, patchy stands to extremely closed stands (Fitzhugh 1988). Understory grasses and shrubs may be present or absent; typical shrub coverage ranges from 10-30% and grass coverage ranges from 5-10% (Barbour 1986 cited in Fitzhugh 1988). Tree composition can range from pure ponderosa pine stands to a mixture of other tree associates up to 50% which can include: incense cedar, Coulter pine, Jeffrey pine, sugar pine, bigcone Douglas-fir, canyon live oak, California black oak, Pacific madrone, and tanoak (Fitzhugh 1988). When shrubs are present, typical associates include manzanita, *Ceanothus* spp., mountain-misery (*Chamaeboatia foliolosa*), hairy yerba-santa (*Eriodictyon trichocalyx*), yellowleaf silktassel (*Garrya flavescens*), bitter cherry (*Prunus emarginata*), California buckthorn, and poison oak (Fitzhugh 1988). Grasses and forbs may include *Bromus* spp., *Carex* spp., smallflower melic grass (*Melica imperfecta*), *Pao* spp., bottlebrush squirreltail (*Elymus elymoides*), *Galium* spp., *Pteridium* spp., bush morning-glory (*Ipomoea leptophylla*), Child's blue-eyed mary (*Collinsia childii*), shrubby eriastrum (*Eriastrum densifolium*), splendid gilia (*Saltugilia* *splendens*), *Iris* spp., whisker-brush (*Leptosiphon ciliatus*), *Lupinus* spp., purple nightshade (*Solanum xanti*), *Streptanthus* spp., and goosefoot violet (*Viola purpurea*; Fitzhugh 1988).

Closed-cone pine and cypress

Closed-cone pine and cypress habitat composition is highly variable. Stands are typically dominated by either closed-cone pine or cypress species, or occasionally a combination of both (Jensen 1988). These habitats often are characterized by a well-developed shrub layer of chaparral, usually chamise (*Adenostoma fasciculatum*) and manzanita (*Arctostaphylos* spp.), and the trees typically form dense, even-aged stands that result from stand-replacing fires (M. Borchert, pers. comm., 2015; Jensen 1988). Closed-cone forests include knobcone pine, bishop pine, Torrey pine, lodgepole pine (*P. contorta*), and Coulter pine (Jensen 1988). In cypress-dominated habitats, tree species include Tecate cypress, Cuyamaca cypress, and Sargent cypress (Jensen 1988).

Management potential

Habitat experts evaluated conifer habitats to be of high societal value.[23] Conifer habitats are a relatively rare and easily recognizable habitat in southern California, and are valued for providing water, grazing, and recreational opportunities (Vulnerability Assessment Reviewers, pers. comm., 2015); most recreational activity concentrated within state parks and national forests (M. Wells, pers. comm., 2015). Conifer habitats also provide a variety of ecosystem services, including: carbon sequestration, flood and erosion protection, biodiversity, water supply/quality/sediment transport, recreation, grazing, air quality, nitrogen retention, public health, timber, fire regime controls, and quality of life (Vulnerability Assessment Reviewers, pers. comm., 2015).

Habitat experts identified that there is low-moderate potential for managing or alleviating climate impacts for conifer habitats,[24] partly because they do not have much room to shift toward higher elevations. Habitat experts identified the following actions as potential management options for conifer habitats, but commented that actions taken will depend largely on direction and rate of future climate changes:

- Conduct a regional survey of the burned areas outside of state parks to quantify the impact of fire and assess whether reforestation would be practical or beneficial (M. Wells, pers. comm., 2015).
- Reforest sites that have been damaged by wildfire to support regional biodiversity and aid in carbon sequestration (Scott et al. 2013; M. Wells, pers. comm., 2015). Restoration efforts could focus on areas damaged between 2002-2009, during which over 50% of the conifer forest in San Diego County burned (M. Wells, pers. comm., 2015).

Recommended Citation

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[23] Confidence: High

[24] Confidence: High

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