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Natural Resource Stewardship and Science

Effects of Climate Change on Vegetation

Oregon Caves National Monument and Preserve

Natural Resource Report NPS/ORCA/NRR—2020/2155

ON THIS PAGE
Mixed conifer forest at Oregon Caves National Monument and Preserve
Photo by Jessica Halofsky

ON THE COVER
View of the Illinois Valley from the Cliff Nature Trail vista at Oregon Caves National Monument and Preserve
Photo by Jessica Halofsky

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Executive Summary

Climate change is likely to have far-reaching impacts on vegetation throughout the Pacific Northwest. Vegetation assemblages are likely to be affected directly through shifts in growth, mortality, and regeneration and indirectly through changes in hydrologic systems, snowpack, spread of nonnative species, and shifts in other ecosystem processes.

This assessment of climate change effects on vegetation in Oregon Caves National Monument and Preserve is completed within the context of historical management actions and disturbances. The vegetation assemblages of southwest Oregon and northern California have evolved over time, shaped by past changes in climate and the biophysical environment. While pine, oak, and chaparral species were common during the warm post-glacial period of the early Holocene, Douglas-fir and hemlock dominated during the cooler, wetter conditions of the late Holocene.

The mixed-severity fire regime and irregular fire return interval of southwest Oregon have significantly influenced vegetation patterns, diversity, and fuel conditions across the landscape. There is strong observational evidence that historical climate was the main driver of regional fire regimes, with major fire years coinciding with summer drought years. Over the past 100 years, climate and fire observations from the Pacific Northwest indicate that warm and dry conditions typically result in larger fires and greater area burned. There have been a number of mixed-severity fires in southwestern Oregon over the past several decades. For example, the 2002 Biscuit Fire was the largest recorded forest fire in the state, with 500,000 acres (200,000 ha) burned; in 2017, over 100,000 acres (40,000 ha) of this fire area was reburned in the Chetco Bar fire.

Increasing air temperatures, decreasing snowpack, and drier conditions will affect fire frequency, extent, and severity in southwest Oregon. Climate-driven shifts in insect outbreaks will also affect future fire regimes. Modeling studies agree that area burned is projected to increase in the western United States as the climate warms. For example, with a mean temperature increase of 3.6 °F (2 °C), the area burned by wildfire in most western states is projected to increase by a factor of 1.4–5.

Warming temperatures are expected to affect insect species directly by altering insect species distribution and population dynamics, such as enhanced winter survival and shortened generation times, and indirectly via shifts in host tree distribution and natural defenses of trees, and interactions with competitors. As temperatures warm and forest drought stress occurs more regularly, insect-mediated disturbances are expected to become more frequent and more intense throughout the western United States. Native forest insects that currently are major disturbance agents in southwest Oregon include the bark beetle species, fir engraver, and western pine beetle. Warming temperatures may also facilitate the northward migration of introduced and native insects from regions south of southwest Oregon.

Fungal pathogens and other disease outbreaks are projected to change in frequency and location as pathogen suitability increases or as forests become more vulnerable to infection. However, it is difficult to predict how climate change will affect specific pathogens because responses are likely to be species-specific. The growth and spread of Swiss needle cast and white pine blister rust is influenced directly by climate and local site conditions and will likely be affected by climate change. Laminated root rot and dwarf mistletoe appear to be indirectly associated with local climate and are less likely to be affected by climate change. The most significant implication of climate change for forest disease outbreaks will likely be increased host vulnerability driven by drought stress and other environmental factors.

Warming temperatures are expected to increase summer drought stress in southwest Oregon. Water deficit stresses forest ecosystems and can increase the frequency, extent, and severity of forest disturbances, specifically wildfire and insect outbreaks. Long-term drought that overlaps with post-fire regeneration years or repeated burns may lead to shifts in vegetation assemblages. For example, conifer recruitment post-disturbance in the Pacific Northwest can take several years, potentially enabling shrub or grass establishment during drought periods. However, conifers could likely establish over shrubs or grasses during cooler, wetter periods.

A dynamic global vegetation model was used to simulate potential changes in southwest Oregon vegetation groups under different climate scenarios. This model integrates information on plant physiology, range data, biogeochemistry, and biophysics with climate model projections to simulate changes in vegetation and ecosystem function through time. Results of this modeling effort suggest that the climate will no longer be suitable for some of the current vegetation types in southwest Oregon and that changes in species composition and abundance are likely. Specifically, the model projects the loss of climatically suitable habitat for high-elevation subalpine forest, an expansion of subtropical mixed forest in the western portion of southwest Oregon, and an expansion of the warm mixed-forest type. Another landscape change model was used to evaluate the potential effects of climate change on fire in southwest Oregon. Under all future climate scenarios assessed, the model projects a decrease in the amount of forested area; a decrease in area dominated by high-elevation mixed conifers; and an increase in the cover of shrubs, chaparral, and hardwoods.

In summary, climate change is expected to have myriad effects on vegetation assemblages in Oregon Caves National Monument and Preserve. This assessment provides detailed description of expected impacts on focal plant communities, including subalpine, riparian and wetland, and montane shrubland communities. This assessment also identifies 24 adaptation strategies and 71 adaptation tactics that are expected to help vegetation communities in southwest Oregon become more resilient to the effects of climate change or transition ecosystems to new conditions. Together, this information offers a rigorous foundation to inform future efforts to reduce climate risks to the unique flora and fauna of Oregon Caves National Monument and Preserve.

1. Introduction

The geologic, topographic, and climatic diversity of Southwest Oregon fosters the highest vegetative diversity in the Pacific Northwest region, with 26 conifer species that occur in this region (Oregon Flora Project 2018). Southwest Oregon encompasses a portion of the Klamath-Siskiyou Ecoregion, one of seven International Union for Conservation of Nature areas of global botanical significance in North America (Wagner 1997). Diverse floras from several U.S. floristic provinces are found in the region, which is characterized by complex environmental and geomorphological gradients, including an east-west transition from continental to coastal climates and a north-south transition from temperate to Mediterranean climates. These complex gradients have allowed for persistence of localized climatic conditions, or climate refugia, amid broader climatic changes in the past (e.g., glaciations and volcanic events) (Waring 1969, Whittaker 1960).

In order to assess the potential effects of climate change on vegetation at Oregon Caves National Monument and Preserve (ORCA) (Figure 1), it is helpful to understand interactions between climate and this vegetative diversity and complexity in the broader region of southwest Oregon (Halofsky et al. in press). Multiple approaches can be used to assess and project the effects of climate change on ORCA vegetation communities, from expert opinion to simulation models. See Section 2.1 for terminology related to vulnerability assessments, methods used to assess vulnerability, and background on management response options.

This report describes a vegetation-focused climate change vulnerability assessment developed for ORCA. The following sections provide details on expected effects of climate change on vegetation and disturbance in the southwest Oregon region, and an assessment of expected effects of climate change on vegetation and disturbance specific to ORCA. The report concludes with a section on climate change adaptation and a final discussion section synthesizes findings for the entire report.

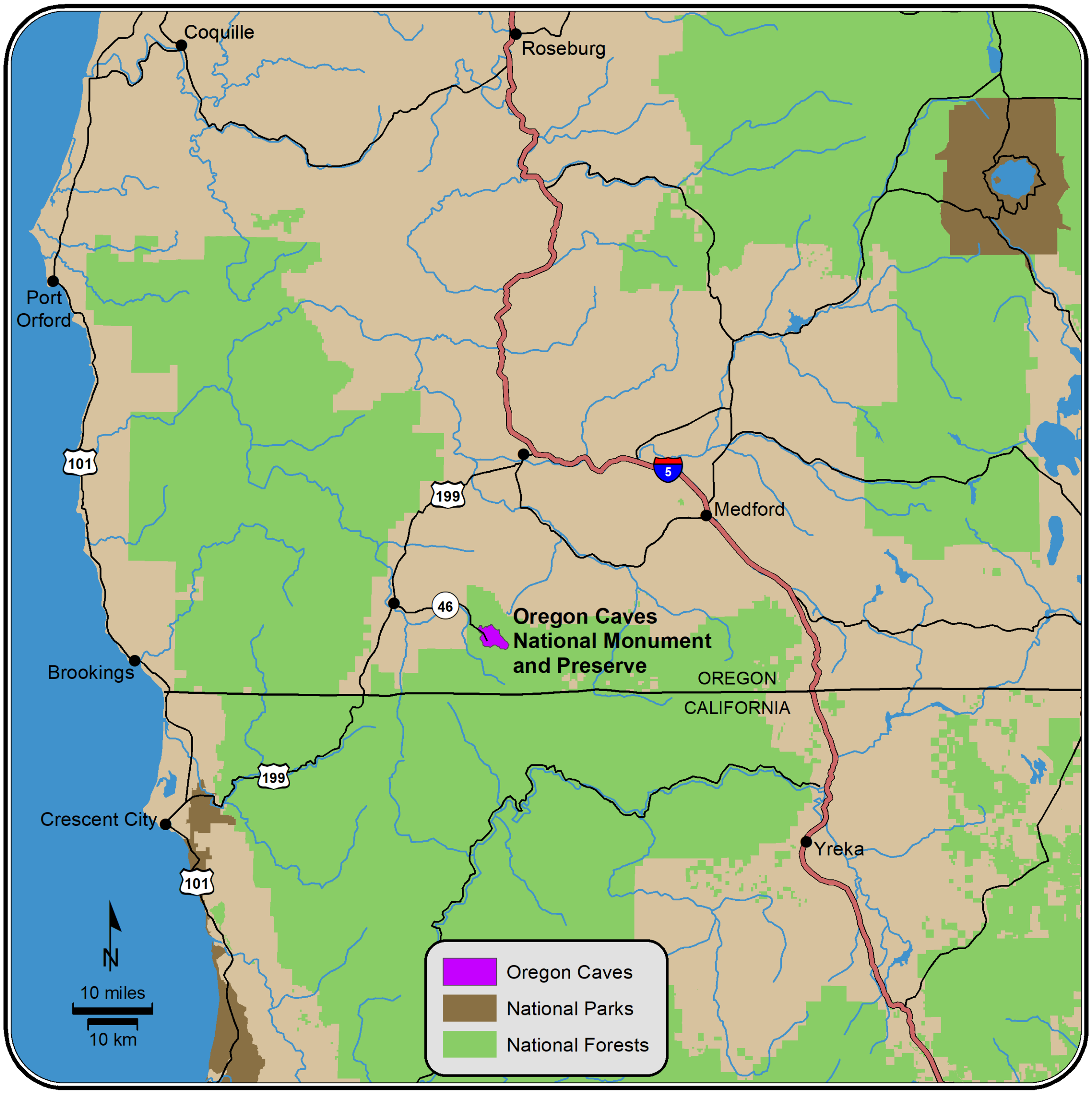

Figure 1. Location of Oregon Caves National Monument and Preserve in southwestern Oregon (map by R. Norheim).

2. Effects of Climate Change on Vegetation in Southwest Oregon

Climate change will likely alter ecosystem processes and vegetation structure and composition in future decades (Peterson et al. 2014). Projected rapid changes in climate have the potential to alter vegetation assemblages in the Pacific Northwest, directly through shifts in vegetation growth, mortality, and regeneration, and indirectly through changes in disturbance regimes and interactions with other ecosystem processes, such as hydrology, snow dynamics, and establishment of nonnative plant species (Hansen and Phillips 2015, Littell et al. 2010, Notaro et al. 2007). If these effects occur over large spatial and temporal scales, they will alter how vegetation is managed by public land agencies.

Some species may be in danger of rapid decreases in abundance, while other may experience range expansion. New vegetation communities might form, while historical vegetation complexes may simply shift to other areas of the landscape or become rare. Juxtaposed with climate change concerns are the consequences of other land management policies and past activities, such as fire exclusion, fuel treatments, and grazing. A thorough assessment of the responses of vegetation to projected climate change is needed, along with an evaluation of the vulnerability of valued species, assemblages, and vegetation-related resources that may be altered by climate change. An assessment of climate change effects on vegetation must be done in the context of historical management actions and disturbances.

2.1. Terminology and methods related to climate change vulnerability assessments and adaptation.

2.1.1. Climate change vulnerability terms

Adaptive capacity is the ability of a plant, species, or system to adjust to climate change (including climate variability and extremes) to moderate potential damages, to take advantage of opportunities, or to cope with the consequences.

Exposure is the nature and degree to which a system is exposed to significant climate variations.

Resilience is the degree to which ecosystems can recover from one or more disturbances without a major shift in composition or function.

Resistance is the ability of an organism, population, community, or ecosystem to withstand perturbations without significant loss of structure or function (i.e., remain unchanged). From a management perspective, resistance includes both (1) the degree to which change can be resisted, and (2) the manipulation of the physical environment to counteract and resist physical and biological change (e.g., burning, harvest treatments) (Millar et al. 2007).

Sensitivity is the degree to which a system is affected, either adversely or beneficially, by climate variability or change. The effect may be direct, such as decreases in tree growth with increased moisture deficits, or indirect, such as damages caused by an increase in the frequency of fire.

Stressors are any physical, chemical, or biological entity that can induce an adverse ecosystem response. Stressors can arise from physical and biological alterations of natural disturbances, increased unmanaged demand for ecosystem services (e.g., recreation), alterations of the surrounding landscape, chemical alterations in regional air quality, or from past management actions (Joyce et al. 2008).

Vulnerability is the degree to which a system is susceptible to, and unable to cope with, the adverse effects of climate change, including associated climate variability and extremes. Vulnerability is a function of the character, magnitude, and rate of climate change and variation to which a system is exposed, its sensitivity, and its adaptive capacity.

2.1.2. Methods for evaluating vegetation vulnerability to climate change

Expert opinion involves having scientific experts qualitatively assess what will happen to vegetation under various climate change scenarios. The majority of papers about climate change impacts on vegetation used in this report were written by experts who have evaluated future climate projections and used their experience to deduce how vegetation will respond to different climates.

Field assessment includes field data collection or remote sensing imagery to monitor vegetation change as the climate warms. Field sampling involves establishing plots in networks across the landscape, detecting change between plot measurements, and correlating these changes to climate data. Demographic studies track individuals over time, rather than use periodic plot-scale inventories, to fully understand the role of climate relative to other factors like competition, variation in physiology and function, and vulnerability to insects and pathogens. Demographic data are rare (Iverson and McKenzie 2013), the U.S. Forest Service Forest Inventory and Analysis has shown promise in quantifying some long-term trends in tree species (e.g., Monleon and Lintz 2015).

Global climate models (GCMs) are an overarching source of uncertainty in projecting climate change effects (Hayhoe et al. 2018). GCMs are used to represent the processes that control climate, but each GCM represents those processes differently. However, all GCMs project a warmer climate in future decades (but with different magnitude and rate of warming) (Roe and Baker 2007). For this reason, multiple GCMs are often used in vulnerability assessments to explore a range of potential future conditions. GCMs generally project changes in mean temperature and precipitation but are not designed to project the frequency and magnitude of future climate extremes, which are expected to drive most ecosystem responses (Easterling et al. 2000).

Mechanistic (process-based) simulation modeling is used to assess climate-mediated vegetation responses (Gustafson 2013, Iverson and McKenzie 2013). Future projections of climate are input into ecological models to simulate climate change effects. Existing models simulate ecological change at broad (e.g., global, regional) and fine (e.g., point, stand, ecosystem) scales. A mechanistic simulation approach that emphasizes physical drivers of vegetation dynamics related to climate is preferable in order to realistically model species composition changes (Gustafson 2013). Ecosystem models that simulate disturbances, vegetation, and climate, and their interactions across multiple scales are especially useful (Purves and Pacala 2008).

Statistical analysis can be used create empirical models that project climate change response. Most studies that project the habitat, range, or occupational shifts of tree species from climate warming use species distribution models (SDMs) (also called bioclimatic envelope models, niche models, and species envelope models) to project future geographical ranges (Hansen and Phillips 2015, Iverson and Prasad 2002). These models link current climate with the current distribution of a species, then compute future species distribution using projected future climate data as inputs. However, SDMs are inherently flawed for projecting future species distributions, because they assume steady-state conditions and fail to consider ecological processes or competition between species.

Uncertainty in projecting climate change effects can result from a lack of information or from disagreement about existing data and inferences. Uncertainty can be represented by quantitative measures, such as a range of values, or by qualitative statements, such as assessment from multiple scientific sources. Uncertainty differs from variability, which is the actual range of a value or variable. *Uncertainty* is also inherent in projecting the effects of climate change on biophysical components of ecosystems, including vegetation (Araújo et al. 2005). It is difficult to accurately quantify mechanistic relationships between climate and plant species, so simulation models often lack important direct effects as well as the indirect effects of disturbance (e.g., wildfire, insects) and other factors (Notaro et al. 2007). It is also difficult to determine how plant reproduction, growth, and mortality will respond to changing climate (Ibanez et al. 2007, Keane 2001). Uncertainties associated with modeling increase as projections are made at finer spatial scales and further into the future (Xu et al. 2009).

2.1.3. Management response options

Adaptation is an adjustment in natural or human systems in response to actual or expected climatic stimuli or their effects, which hopefully moderates harm or exploits beneficial opportunities (Spittlehouse and Stewart 2004). Adaptation is often referred to as preparedness and is based on scientifically supported strategic and tactical activities that support sustainable resource management. Adaptation addresses specific aspects of the sensitivity of resources to an altered climate.

Adaptation strategy describes a general, overarching adaptation approach that encompasses a range of potential activities that can be used to address a specific vulnerability (e.g., increase tree vigor, increase resilience to high-intensity wildfire) (Halofsky et al. 2011b).

Adaption tactic describes a specific action, typically described in management and planning documents, that can be implemented on the ground (e.g., reduce stem density and surface fuels, increase culvert size on roads) (Joyce et al. 2008, Millar et al. 2007).

Ecological restoration is the practice of reestablishing or augmenting historical biophysical conditions and functions, thus increasing the likelihood that specific flora and fauna can be maintained (Egan and Howell 2001). However, this objective can be difficult to manage because (1) little is known about historical conditions, (2) some key species may already be lost, (3) some efforts may be prohibitively expensive, and (4) future climates may produce novel biophysical conditions and habitats.

Functional restoration, or the restoration of abiotic and biotic processes in degraded ecosystems, may be a more realistic approach with climate change. Functional restoration focuses on underlying processes that may be degraded, regardless of the structural condition of the ecosystem, with an emphasis on interactions between structure and function rather than on a historical reference condition. Functional restoration may increase resilience to climate change and shifts in disturbance regimes by promoting conditions that support native species (over invasive species). In a warmer climate, a functionally restored system may look different than the historical reference condition in terms of structure and composition, forcing reconsideration of traditional approaches that focus on species conservation.

2.2. Paleoecology of Southwest Oregon and Northern California

Plant pollen and charcoal from wildfires deposited in lake sediments provide records of past vegetation composition and abundance and fire at local to regional scales over hundreds to thousands of years. Charcoal records can be used to identify individual fire events and to estimate fire frequency (Itter et al. 2017). In combination with sediment pollen records, charcoal records help to determine how vegetation and fire regimes shifted with changes in climate in the past.

Vegetation composition and fire regimes in southwest Oregon and northern California have varied significantly with climate in the past (Briles 2017; Briles et al. 2005, 2008; Colombaroli and Gavin 2010). Paleoecological studies also indicate that species respond individualistically to climate change; the vegetation communities that occurred thousands of years ago are different from those that currently occur, and it is possible that novel communities will develop in response to future climatic changes. However, in southwest Oregon, most of the plant communities of the past have modern analogues, suggesting that even with climatic change, species were able to find niches nearby and persist (Briles 2017, Briles et al. 2011, Daniels et al. 2005).

During wetter and cooler periods in the past, fires were less frequent in southwest Oregon and Northern California, and vegetation was dominated by species favored by wetter and cooler conditions, such as western hemlock (*Tsuga heterophylla*), mountain hemlock (*T. mertensiana*), and true firs (Briles 2017, Briles et al. 2008, Mohr et al. 2000). For example, in the late Holocene, cool wet conditions favored closed forests of true fir, Douglas-fir (*Pseudotsuga menziesii*), and hemlock on most soils (Briles et al. 2008). In contrast, during warmer and drier periods in the past, fire frequency increased in southwest Oregon and northern California, and species such as Douglas-fir, pines, oaks, and chaparral species were more abundant (Briles 2017, Briles et al. 2005, Colombaroli and Gavin 2010, Daniels et al. 2005).

The early Holocene (ca. 10,500 to 5,000 years BP) was the warmest post-glacial period in the Pacific Northwest (Whitlock 1992). During that time, summers were warmer and drier relative to recent historical conditions, with more intense droughts (Briles et al. 2005, Whitlock 1992), causing higher fire frequency in some locations (Walsh et al. 2015, Whitlock 1992). Fire activity during the early Holocene (Briles 2017, Briles et al. 2005, Colombaroli and Gavin 2010), led to expansion of oak, incense cedar (*Calocedrus decurrens*), lodgepole pine (*Pinus contorta* var. *latifolia*), and Jeffrey pine (*Pinus jeffreyi*) (Daniels et al. 2005). Maritime forest in the Oregon Coast Range shifted to open

Douglas-fir forest, with red alder (*Alnus rubra*) in areas of frequent disturbance and oak on the driest sites. Mean fire interval was approximately 110 years (Long et al. 1998).

During warm periods at higher elevations, subalpine parklands were replaced by moist forest. During the early Holocene, subalpine parklands in the Siskiyou Mountains were replaced by a closed forest of western white pine (*Pinus monticola*) or sugar pine (*Pinus lambertiana*), species in the Cupressaceae family (most likely Port Orford cedar [*Chamaecyparis lawsoniana*]), fir, and Douglas-fir (Briles et al. 2008). Vegetation at Twin Lake (1,200 m elevation) in the southern Siskiyou Mountains shifted from subalpine forest dominated by mountain hemlock to a forest of pine and cedar, with Douglas-fir and tanoak (*Lithocarpus densiflorus*) as minor components (Wanket 2002). Fire activity also increased during this time.

2.3. Observational Studies

Warming climate in recent decades and land management effects have been associated with shifts in herbaceous communities on 185 sites in the Siskiyou Mountains between 1950 and 2008 (Damschen et al. 2010, Harrison et al. 2010). These results suggest that species with narrow ecological ranges (habitat specialists) may be at risk (Damschen et al. 2010). Harrison et al. (2010) evaluated compositional changes in herbaceous communities in upper montane primary (never logged) forest, lower montane primary forest, and lower montane secondary (previously logged) forest. They found modest changes in herbaceous communities in the higher-elevation forests, and significant changes in the lower-elevation forests, regardless of management history. Composition changes in lower montane forests, including a reduction in specific leaf area and a reduction in cover of more northerly species, were consistent with a shift to a drier climate. At higher elevations, forest canopy cover increased, possibly because of longer snow-free growing seasons (Harrison et al. 2010). Tree species composition did not significantly shift over the time period (Damschen et al. 2010).

Monleon and Lintz (2016) examined the presence and absence of 46 tree species on Forest Inventory and Analysis plots across Washington, Oregon, and California. They found that the mean temperature of the range of a species' seedlings was significantly different from the mean temperature for the range of mature adults for 20 species. Sixteen of these 20 species had seedling site temperatures below temperatures for mature adults. The authors inferred that despite land management practices and fire suppression, many species are shifting toward cooler environments. These apparent shifters include western hemlock, tanoak, western redcedar (*Thuja plicata*), and Oregon white oak (*Quercus garryana*). Four species (sugar pine, Jeffrey pine, incense cedar, white fir [(*Abies concolor*]) appear to be responding in the opposite direction, with seedlings moving toward warmer environments.

3. Effects of Climate Change on Disturbance

3.1. Wildland Fire

3.1.1. Historical fire patterns

Southwest Oregon is largely characterized by a mixed-severity fire regime, with historical fire return intervals of less than 35 years over much of the region. However, some portions of southwest Oregon are characterized by a mixed-severity fire regime with 35- to 200-year fire return intervals. Higher-elevation and wetter forests in the eastern (Cascade Range) portions of the assessment area and wetter western hemlock forests near the coast are characterized by a high-severity (stand-replacement) fire regime with >200-year fire return intervals.

Fire has helped to shape the complex vegetation patterns in southwest Oregon, and in turn, the diversity of vegetation and fuel conditions contribute to complex burn patterns. Mixed-severity fires are characterized by mixing, at relatively fine scales, of patches of vegetation burned to varied levels of severity (Halofsky et al. 2011a). Mixed-severity fires and irregular fire return intervals in southwest Oregon lead to highly variable patch age. Varied fire effects result in (and result from) fine-scale variation in patch age and vegetation composition, which provides habitat for a variety of species in relatively close proximity and likely promotes resilience to fire (Halofsky et al. 2011a).

Fire-scar studies provide strong evidence that climate was historically a primary determinant of fire regimes in southwest Oregon and across the Pacific Northwest. Years with increased fire frequency and area burned were generally associated with warmer and drier spring and summer conditions in the Pacific Northwest (Hessl et al. 2004, Heyerdahl et al. 2008, Taylor et al. 2008, Wright and Agee 2004). However, summer drought during the year of the fire seems to have the strongest association with major fire years at the site and regional scales (Hessl et al. 2004). Summer drought conditions are important in the Pacific Northwest, where most precipitation falls in winter, and the fire season occurs primarily in late summer (August–September) when drought reduces fuel moisture (Hessl et al. 2004).

Modern climate and fire records indicate that, over the last century in the Pacific Northwest, warm and dry conditions in any given year (primarily in summer, but also in winter and spring) generally led to larger fires and greater area burned (Abatzoglou and Kolden 2013; Cansler and McKenzie 2014; Dennison et al. 2014; Kitzberger et al. 2017; Littell et al. 2009, 2010; McKenzie et al. 2004; Reilly et al. 2017; Stavros et al. 2014; Trouet et al. 2006, 2009; Westerling 2016). In the 20th century, wildfire area burned in the Pacific Northwest was positively related to low precipitation, drought, and temperature (Abatzoglou and Kolden 2013, Littell et al. 2009). In southwest Oregon, area burned is also positively associated with drought (Trouet et al. 2009).

Annual area burned has increased only slightly during the 1985 to 2010 period in the Pacific Northwest, and the proportion of fires that burned at high severity has not increased, across the entire region or for any particular vegetation zone (Reilly et al. 2017). However, as total annual area burned has increased, so has the area that burns at high severity, both in total area and in average patch size (Reilly et al. 2017). Several analyses have shown a positive correlation between annual area burned and area burned severely (in large patches) in the Pacific Northwest (Cansler and McKenzie 2014, Dillon et al. 2011, Reilly et al. 2017).

Over the last several decades, a number of mixed-severity fires have occurred in southwest Oregon, including the 500,000-acre (200,000-ha) Biscuit Fire of 2002, the largest recorded forest fire for the state of Oregon. During the summer of 2017, the 190,000-acre (77,000-ha) Chetco Bar fire burned over 100,000 acres (40,000 ha) of the Biscuit Fire area. The 43,000-acre (17,500-ha) Umpqua North Complex fire also took place during the summer of 2017.

The topography and vegetation of southwest Oregon are complex, so generalizations about the effects of fire exclusion on forests in the region are tenuous (Perry et al. 2011). However, fire exclusion has likely increased forest density and favored shade-tolerant and fire-intolerant species such as white fir in some locations. The effects of fire exclusion, combined with the effects of timber harvest in the region (that created areas of dense, young trees), have likely increased the risk of large, high-severity fires (Myer 2013, Perry et al. 2011). Currently, dense forest cover is found across much of the landscape, creating continuous fuels that can carry high-severity crown fire.

In a comparison of current forest conditions to the historical range of variation for southwest Oregon, Haugo et al. (2015) suggested there is a significant need for thinning and/or prescribed burning to restore forests characterized by low- and mixed-severity fire to historical conditions. A recent terrestrial condition assessment for national forest lands rates wildfire hazard as very high in southwest Oregon (Cleland et al. 2017).

3.1.2. Future fire projections

Continued warming will affect fire frequency, extent, and severity in southwest Oregon. Increased temperatures, decreased snowpack, and earlier snowmelt will probably lead to longer fire seasons, lower fuel moisture, higher likelihood of large fires, and greater area burned by wildfire (Littell et al. 2010, McKenzie et al. 2004, Stavros et al. 2014, Westerling et al. 2016). Climatic changes and associated stressors and disturbances (e.g., drought, pathogens, insect outbreaks) will interact with vegetation conditions, as affected by historical land uses such as tree harvest and fire suppression, to affect fire regimes and ecosystem conditions in the future (Keeley and Syphard 2016).

Numerous studies have developed empirical models to project future area burned or fire potential at both global (Krawchuk et al. 2009, Moritz et al. 2012) and regional scales (e.g., western U.S.) (McKenzie et al. 2004, Kitzberger et al. 2017, Littell et al. 2010, Yue et al. 2013). All of these studies, including the global-scale studies, agree that fire potential and/or area burned will increase in the western United States in a warmer climate. McKenzie et al. (2004) projected that area burned by wildfire will increase by a factor of 1.4–5 for most western states, including Oregon, with a mean temperature increase of 3.6 °F (2 °C). Similarly, Kitzberger et al. (2017) projected increases in annual area burned of five times the median in 2010–2039 compared to 1961–2004 for the 11 states in the western U.S. Empirical models developed for the Pacific Northwest suggested that fire area burned will increase by 300 to 500% with a 2.2 °F (1.2 °C) increase in southwest Oregon (Mote et al. 2014). The probability of very large fires is also expected to increase in the Pacific Northwest (Barbero et al. 2015, Davis et al. 2017).

Mechanistic model simulations for the Pacific Northwest suggest that both fire frequency and area burned will increase in the future (Rogers et al. 2011, Sheehan et al. 2015). Fire severity may also increase, depending partly on forest composition, structure, and productivity over time. In relatively productive areas, fire severity can be greater in reburns than in comparable single burns once the interval between fires exceeds 10–12 years (Thompson et al. 2007). However, in southwest Oregon, short-interval (i.e., 15 years between fires), high-severity (Silver Fire-Biscuit Fire) reburn areas were not qualitatively different than areas that burned once at a longer interval (>100 years between fires).

3.2. Insects

Insect species have relatively short life cycles, high reproductive capacity, and high degree of mobility, so physiological responses to warming temperatures can produce large and rapid effects on species population dynamics (Fettig et al. 2013, Stange and Ayres 2010). Population status, host condition, and weather influence insect life history and potential impact. Epidemics of forest insects and tree diseases exceed (in area) other sources of disturbance to North American forests (Hicke et al. 2016, Weed et al. 2013), and both have significant interactions with climate.

Indirect effects to insects can occur via climate-driven shifts in host tree distribution and defense, as well through interactions among disturbance agents and their own enemies, competitors, and mutualists (Weed et al. 2013). Although insect interactions with physical environmental factors such as temperature, atmospheric gases, and precipitation predominate among climate modelling efforts, the indirect biotic effects of climate on insect disturbance agents and their associates is less well understood.

Practical theory holds that temperature affects insect survival nonlinearly and exponentially, in that insect metabolism rate increases when it is warmer (Ayres and Lombardero 2018). This means that warmer temperatures increase insect consumption, growth, movement, and dispersal, as well as phenology and species interactions. Enhanced winter survival and shortened generation times due to warming may facilitate larger populations of insects, particularly those with multiple generations per year. Those species with necessary sequences of coordinated life cycle events, such as the adaptive seasonality displayed by mountain pine beetle and other bark beetles, have experienced increased population success and recent range expansion due to warmer climatic conditions lifting life cycle constraints (Fettig et al. 2013).

Temperature and precipitation effects on insect hosts (i.e., food), either host plants or other insects, may indirectly affect insect-climate interactions. The relative success of many herbivorous insect species is closely tied to host plant vigor, which can be influenced by changes in climatic conditions. Trees under stress are commonly attacked by aggressive bark beetles and wood borers and may become more vulnerable to root diseases and other disturbance agents. Wildfire and drought, are well known sources of host tree stress that can facilitate insect population increases and associated impacts. In particular, hotter droughts and extreme drought events have induced and been followed by large-scale tree mortality involving insects in western North America where sufficient expanses of susceptible hosts exist (Hicke et al. 2016, Millar and Stephenson 2015, Young et al. 2017).

An increase in the frequency and severity of insect-mediated disturbances is expected throughout the western United States as a result of increased temperatures and more frequent and intense drought stress, although this understanding is derived from a limited number of species in conifer forests (Kolb et al. 2016). In general, warmer temperatures are expected to alter insect population dynamics directly through effects on survival, generation time, fecundity, and dispersal. Insect populations limited by cold during the growing seasons are anticipated to benefit from climate change through more rapid life cycle completion and increased survival. Insect mortality may decrease with warmer winter temperatures, thereby leading to higher elevation and poleward range expansions (Stange and Ayres 2010).

Increased drought severity and frequency are likely to make forests more vulnerable to both direct (reduced growth and mortality) and indirect (insect outbreaks, pathogens, and wildfire) effects (Dale et al. 2001, Kolb et al. 2016, Weed et al. 2013). If extended in time and/or with reduced precipitation, the characteristic summer drought period in southwest Oregon may result in additional tree stress from warming, moisture deficit, and/or wildfire that would favor opportunistic insects. Secondary insect species, including bark beetles in the genera *Ips* and *Scolytus*, may become more significant disturbance agents with an increase in the spatial extent of tree stress. Extreme drought has facilitated short-term, large increases in Douglas-fir mortality due primarily to a wood borer, *Phaenops drummondi*, on lower-elevation, dry sites in southwest Oregon; this is likely to repeat with an increase in drought magnitude and frequency.

Because insects typically migrate much faster than trees, temperate tree species are likely to encounter nonnative insect herbivores that previously were restricted to subtropical forests (Dale et al. 2001). In addition, increased temperatures under a changing climate may allow northward migration of introduced and native insects from areas south of southwest Oregon. Of special concern are ambrosia beetles (Curculionidae: Scolytinae) and their associated fungi, which have large host ranges, are easily transported deep within untreated wood, and affect a broad range of tree species (Ploetz et al. 2013).

Native forest insects that currently are major disturbance agents in southwest Oregon include bark beetle species, fir engraver in true firs, western pine beetle in ponderosa pine (*Pinus ponderosa*), and mountain pine beetle in sugar pine, lodgepole pine, western white pine, and ponderosa pine. Only the firs and localized areas of lodgepole pine occur across a sufficient expanse with a high percentage of hosts that widespread bark beetle epidemics are possible. Although tree mortality from Douglas-fir beetles can be locally important, epidemics have been infrequent in the absence of a large-scale windthrow event. There are no known effects in southwest Oregon from western spruce budworm (*Archips fumiferana*) and Douglas-fir tussock moth (*Orgyia pseudotsugata*), the two major insect defoliators in western North America, though both are present in southwest Oregon.

3.3. Fungal Pathogens and Other Disease Organisms

The dynamics of host-pathogen interactions are influenced by climate, but although a change in climate may affect the behavior and distribution of forest tree diseases (Kolb et al. 2016), it is difficult to project the effects of climate change on specific pathogens (Agne et al. 2018, Kliejunas 2011, Sturrock et al. 2011). The influence of pathogens on the status and function of forests may change coincident with changes in species composition and climate. Any change in climatic conditions that results in an environment more suitable to a pathogen, or host susceptibility to that pathogen, could potentially increase the incidence of disease.

Forest diseases, such as Swiss needle cast (caused by *Phaeocryptopus gaeumannii*) and white pine blister rust (caused by *Cronartium ribicola*), require specific environmental conditions for growth, sporulation, and spread. These pathogens are influenced directly by climate and local site conditions and may be affected by climate change (Sturrock et al. 2011). Diseases such as laminated root rot (caused by *Phellinus sulphurascens*) and dwarf mistletoe (caused by *Arceuthobium* spp.) appear to be indirectly associated with local climate and are less likely to be affected by climate change. The most substantial effect of climate change related to forest diseases will likely be changes in the interactions of pathogenic fungi with trees stressed by drought and other environmental factors. For example, an indirect effect of climate on a pathogen occurs when drought-stressed trees become increasingly infected by Armillaria root disease (Kolb et al. 2016). The effects of climate change on mutualistic microfungi on leaves which can cause entry of pathogenic fungi or render leaves more palatable to insect herbivores is unknown.

Although it is uncertain how specific pathogens will respond to climate change, some general inferences include (Kliejunas 2011):

- Climate change will affect the epidemiology (spread) of plant diseases. Prediction of disease outbreaks could become difficult with rapid climate change or extreme weather events.
- Many pathogens are limited by winter temperature, and higher winter temperature may increase the overwintering survival of pathogens and disease severity.
- The most substantial effect of climate change on plant diseases may be altered interactions between biotic diseases and abiotic stressors such as drought.
- Climate change may facilitate establishment of new nonnative pathogens and thus new epidemics.

Examining host-pathogen interactions can help illustrate what can occur when a pathogen is exposed to a novel environment or host. As the climate warms in southwest Oregon, the pressure of foliage diseases on forest trees will also change. *Phaeocryptopus gaeumannii* (cause of Swiss needle cast in coastal Douglas-fir forests) is known to increase in abundance under warmer winter temperatures (Lee et al. 2016, Manter et al. 2005, Stone et al. 2008). Foliage and canker diseases of Pacific madrone (*Arbutus menziesii*) in southwest Oregon are often associated with moisture stress (Shaw and Bennett 2008). *Phytophthora ramorum*, an introduced oomycete (water mold) that causes sudden oak death and Ramorum leaf and shoot blights on the southwest Oregon coast is strongly influenced by climate. Increased precipitation during spring would favor increased damage by the pathogen and expansion into new locations. (Davidson et al. 2002, Kliejunas 2011).

3.4. Drought

Warmer temperatures will increase evapotranspiration in southwest Oregon, increasing summer water deficit and drought severity (Littell et al. 2013, 2016). Water deficit directly contributes to stresses in forest ecosystems by intensifying negative water balances (Littell et al. 2008, Milne et al. 2002, Restaino et al. 2016, Stephenson 1998). Although water deficit is rarely directly fatal to trees, it is a predisposing factor that can exacerbate the stress complex that includes both biotic and abiotic stressors (McKenzie et al. 2009). Water deficit also indirectly increases the frequency, extent, and severity of disturbances, especially wildfire and insect outbreaks (Logan and Powell 2009, McKenzie et al. 2004, Trouet et al. 2009). These indirect disturbances alter forest ecosystem structure and function, at least temporarily, much faster than do chronic effects of water deficit (e.g., Loehman et al. 2017).

Increasing summer drought stress in southwest Oregon will likely decrease the growth of many tree species (Restaino et al. 2016). Drought may also affect forest regeneration and development, particularly when drought conditions follow fire events (Tepley et al. 2017). The lower the soil moisture, the higher the propagule pressure (smaller high-severity patches with more live seed trees) needed to achieve a given level of regeneration. Therefore, at the levels of water deficit projected for the future, even small high-severity patches are at risk for low post-fire regeneration.

Individual drought years are not likely to alter post-fire successional pathways, especially if wet years occur between dry years (Tepley et al. 2017). However, long-term drought that coincides with critical post-fire regeneration years or repeated fire may lead to altered vegetation states (e.g., where shrub or grass species become dominant for a protracted period, or species that are minor but drought tolerant at the seedling stage become dominant). On some sites, recruitment of conifers following a disturbance can require years to decades in the Pacific Northwest (Little et al. 1994, Shatford et al. 2007, Tepley et al. 2013). Thus, shrubs or grasses may dominate during drought periods, but conifers could establish and overtop shrubs and grasses during wetter and cooler periods (Donato et al. 2016, Dugan and Baker 2015).

4. Vegetation Modelling

4.1. MC2

The MAPSS-CENTURY 2 (MC2) dynamic global vegetation model (DGVM) (Bachelet et al., 2001, Daly et al. 2000) was used to simulate potential changes in broad vegetation groups in southwest Oregon under 28 future climate scenarios (using the NASA NEX-DCP30 climate dataset under Representative Concentration Pathway [RCP] 8.5) (Halofsky et al. in press). DGVMs such as MC2 integrate knowledge of plant physiology, biogeography (the geographic distribution of plants), biogeochemistry, and biophysics with climate model projections to simulate changes in vegetation structure and composition (biogeography) as well as ecosystem function (biogeochemistry) through time (Prentice et al. 1989, 2007). MC2 integrates biogeography, biogeochemistry, and fire into a single modeling environment. For details of modeling methods, see Halofsky et al. (in press).

It is difficult to interpret MC2 model output for 28 different future climate scenarios. Thus, we examined where there was agreement in vegetation projections among those different climate scenarios. Thus, of the 28, we selected five focal GCMs (under RCP 8.5, a “business as usual” or “worst case" scenario) to illustrate a range of potential futures. The selected models are among the better-performing models for the Pacific Northwest, as ranked by Rupp et al. (2013): “mean” CESM1(CAM5) (CESM1 hereafter); “hot-wet” CanESM2; “hot” BNU-ESM; “hot-dry” MIROC-ESM-CHEM (MIROC hereafter); and “warm” MRI-CGCM3 (MRI hereafter).

Across 28 future climate scenarios, MC2 consistently projected vegetation type changes at their elevation extremes (i.e., the High Cascades and low interior valleys) and along the coast (Halofsky et al. in press) (Figures 2 and 3). Agreement was also high for shifts in biomes at the lowest elevations. These include shifts between forest and woodland, and between woodland and shrubland. Agreement that vegetation type and biome changes will occur increased between midcentury and the end of the century.

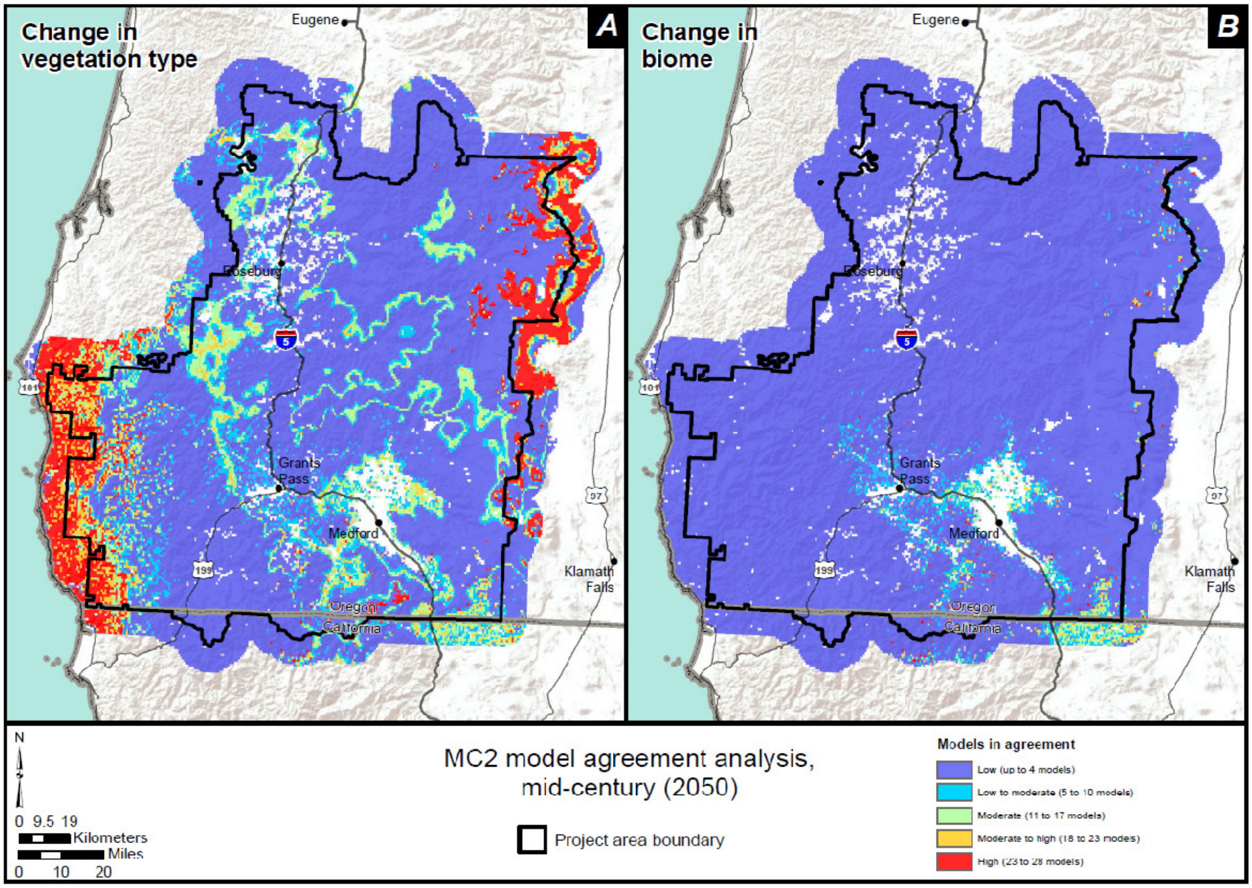

Figure 2. MC2 model agreement (among 28 global climate models run under representative concentration pathway 8.5) at mid-century (2050) for simulated change in vegetation type (A) and simulated change in biome (e.g., forest to woodland or shrubland to grassland) (B). From Halofsky et al. in press.

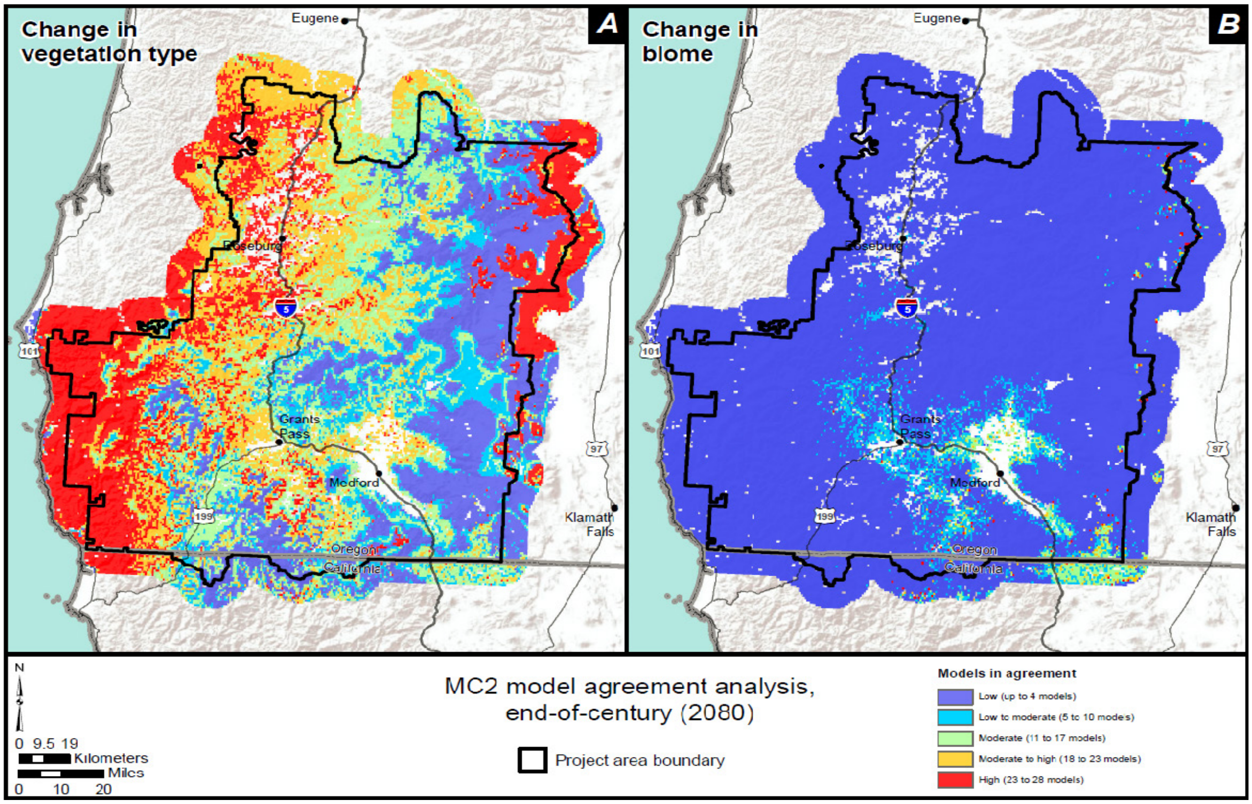

Figure 3. MC2 model agreement (among 28 global climate models run under representative concentration pathway 8.5) at the end of the century (2080) for simulated change in vegetation type (A) and simulated change in biome (e.g., forest to woodland or shrubland to grassland) (B). From Halofsky et al. in press.

Modal (most often occurring) vegetation types for the historical period and middle and end of the 21st century were projected for five different future climate scenarios. Changes in MC2 vegetation type indicate that the climate will no longer be suitable for some current vegetation types and that changes in species composition and abundance are likely (Figure 4; output are shown for only the CESM1(CAM5) GCM ["average/best scenario"] for RCP 8.5). However, changes in species composition and abundance will likely be gradual because of the high tolerance of mature trees to climatic variation. Disturbance by fire will likely be the main trigger for compositional change.

MC2 consistently projected the loss of climatically suitable habitat for high-elevation subalpine forest, which converted to moist coniferous forest. MC2 projected an expansion of subtropical mixed forest in the western portion of southwest Oregon, with greater eastward expansion between roughly 2050 and 2100. The range expansion of the subtropical forest type was at the expense of the currently dominant moist coniferous forest. MC2 also projected expansion of the warm mixed forest type. Under all but the hottest and driest future climate scenario, the area of dry coniferous forest decreased somewhat.

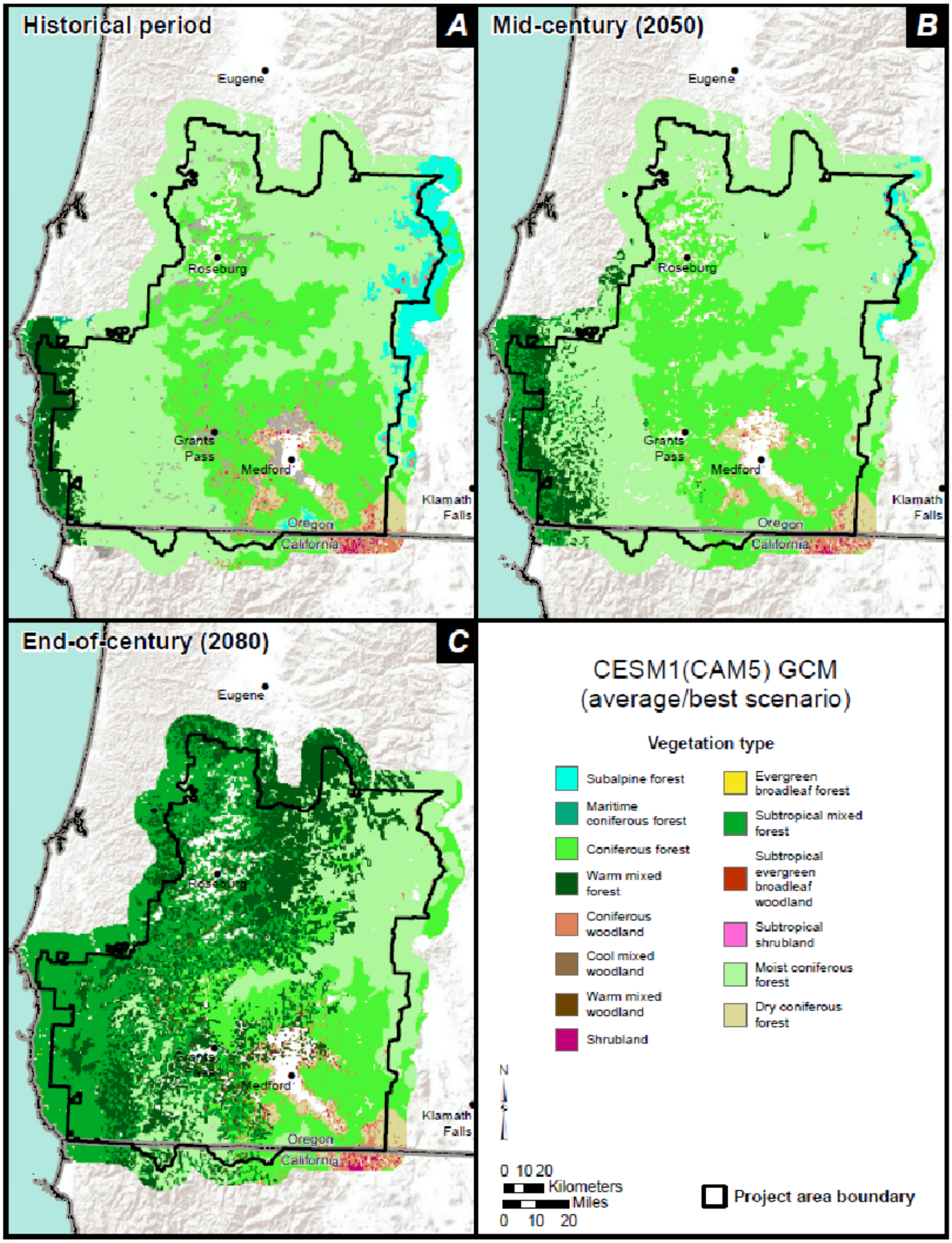

Figure 4. Modal vegetation types for the Southwest Oregon Adaptation Partnership assessment area for the historical period, mid-century and end of century, as simulated by MC2 under the CESM1(CAM5) global climate model (GCM) scenario for RCP 8.5. This model is a highly ranked model for the Pacific Northwest (Rupp et al. 2013), with projected changes in temperature and precipitation similar to the ensemble mean ("near mean"). From Halofsky et al. in press.

Projected woodland area varied with future climate scenario. Under warmer climate scenarios, some woodlands converted to more productive forest types (or novel subtropical evergreen broadleaf or warm mixed woodland types). Shrublands remained a minor component of the landscape across climate scenarios. However, precipitation differences among the models drove whether or not shrublands were projected to increase or decrease; hotter models showed increases in shrublands, whereas wetter models showed decreases.

MC2 simulated decreased mean fire return interval for mid- and late-century compared to the 1970–1999 under all five focal GCMs (under RCP 8.5) (Figure 5). Fire severity (mass of live carbon killed by fire) was generally projected to increase for 2050 and 2100 compared to the historical time period. Under the hot and wet CanESM2 scenario, there were decreases in fire severity for the subalpine forest and woodland types. Changes in mean fire return interval and fire severity can be explained by seasonal changes in temperature and precipitation projected for each of the GCMs that drive fuel moisture, plant productivity, and aboveground biomass, and by the model's complex interplay between these factors.

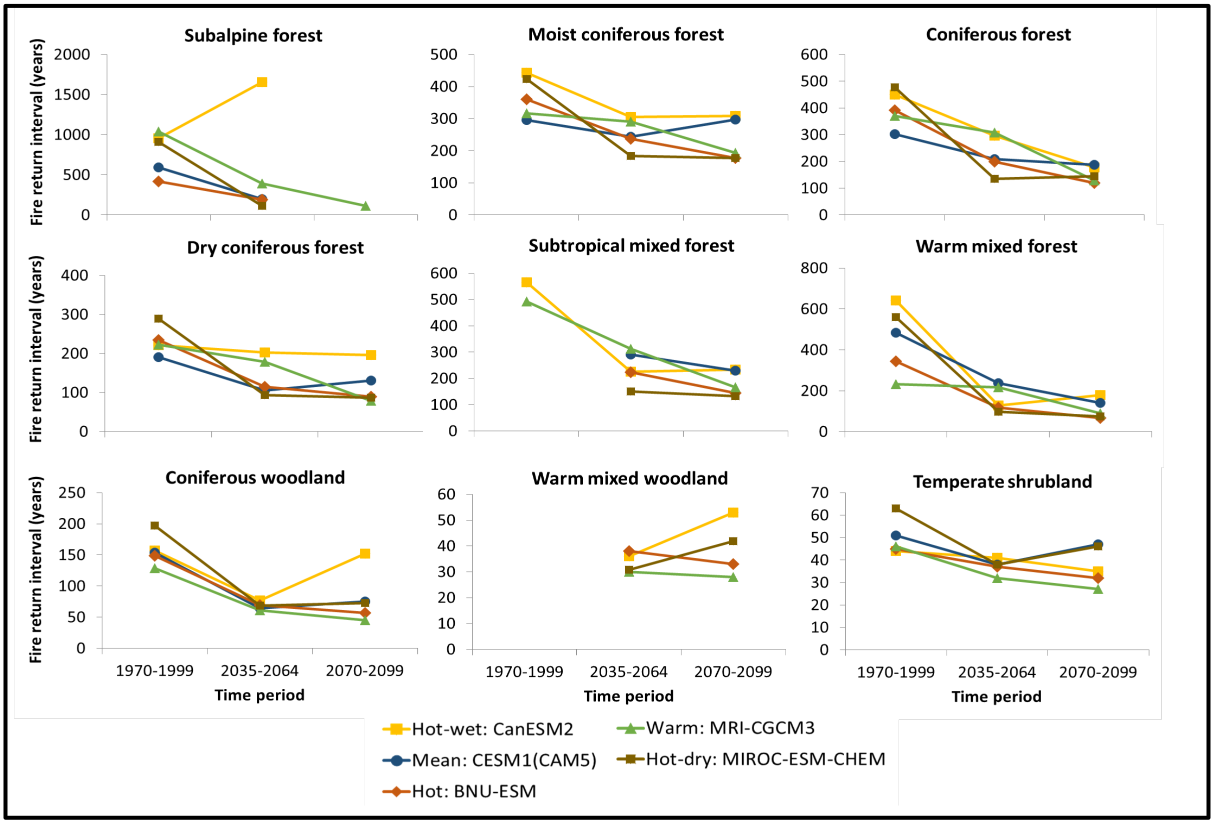

Figure 5. Projected mean fire return interval (MFRI) in years for the historical (1970–1999), mid-century (2035–2064), and end of century (2070–2099) time periods for relevant MC2 vegetation types and global climate model scenarios. A decrease in fire return interval represents an increase in fire frequency. Note differences in scale for the y-axes. Historical data are unavailable for warm mixed woodlands and for two global climate model scenarios for subtropical mixed forest. No warm mixed woodlands occurred in the CESM1 scenario. Data are unavailable for subalpine forest for the end of the century because this type is greatly reduced in area. From Halofsky et al. in press.

The MC2 output should be interpreted only across broad spatial scales. MC2 does not include dispersal processes, genetic adaptation, biotic interactions, or phenotypic plasticity. Although it does incorporate fire, disturbance processes such as insect and disease are not included. MC2 also does not model the complexities of summer drought and climatic water deficit, so potential increases in productivity projected by the model may not be realized.

4.2. LANDIS-II

Another process-based model, LANDIS-II, was used to examine potential effects of climate change on vegetation and fire in southwest Oregon. LANDIS-II (Scheller et al. 2007) simulates growth, mortality, and regeneration at the species level by incorporating biophysical (e.g., climate, disturbance) and ecological processes (e.g., species interactions, dispersal) in a grid-based framework, with processes occurring both within and between grid cells. For details of modeling methods, see Halofsky et al. (in press).

Under all future climate change scenarios, LANDIS-II projected (1) a decrease in the amount of forested area, (2) a decrease in area dominated by high-elevation mixed conifers, and (3) an increase in the cover of shrubs, chaparral, and hardwoods (Figure 6). By 2100, the high-elevation mixed conifer type (including white fir, Shasta red fir [*Abies magnifica*], and western white pine) was almost completely replaced by the Klamath mixed conifer type (including Douglas-fir, sugar pine, ponderosa pine, and incense cedar). Total forest area was reduced by as much as 14% in some future scenarios, concurrent with expansion of shrub, chaparral, and hardwood cover.

Projections for individual species showed reductions in Shasta red fir, western white pine, and white fir. Species that persisted under the different climate scenarios include Pacific madrone, incense cedar, chinquapin (*Chrysolepis chrysophylla*), tanoak, sugar pine, ponderosa pine, Douglas-fir, canyon live oak (*Quercus chrysolepis*), Oregon white oak, and California black oak (*Quercus kelloggii*). Over the course of the simulations, total aboveground biomass substantially increased in the western portion of southwest Oregon, where precipitation less limiting. LANDIS-II projected larger fire sizes and shorter fire return intervals throughout southwest Oregon, especially in the drier eastern portion of the landscape (Figure 7).

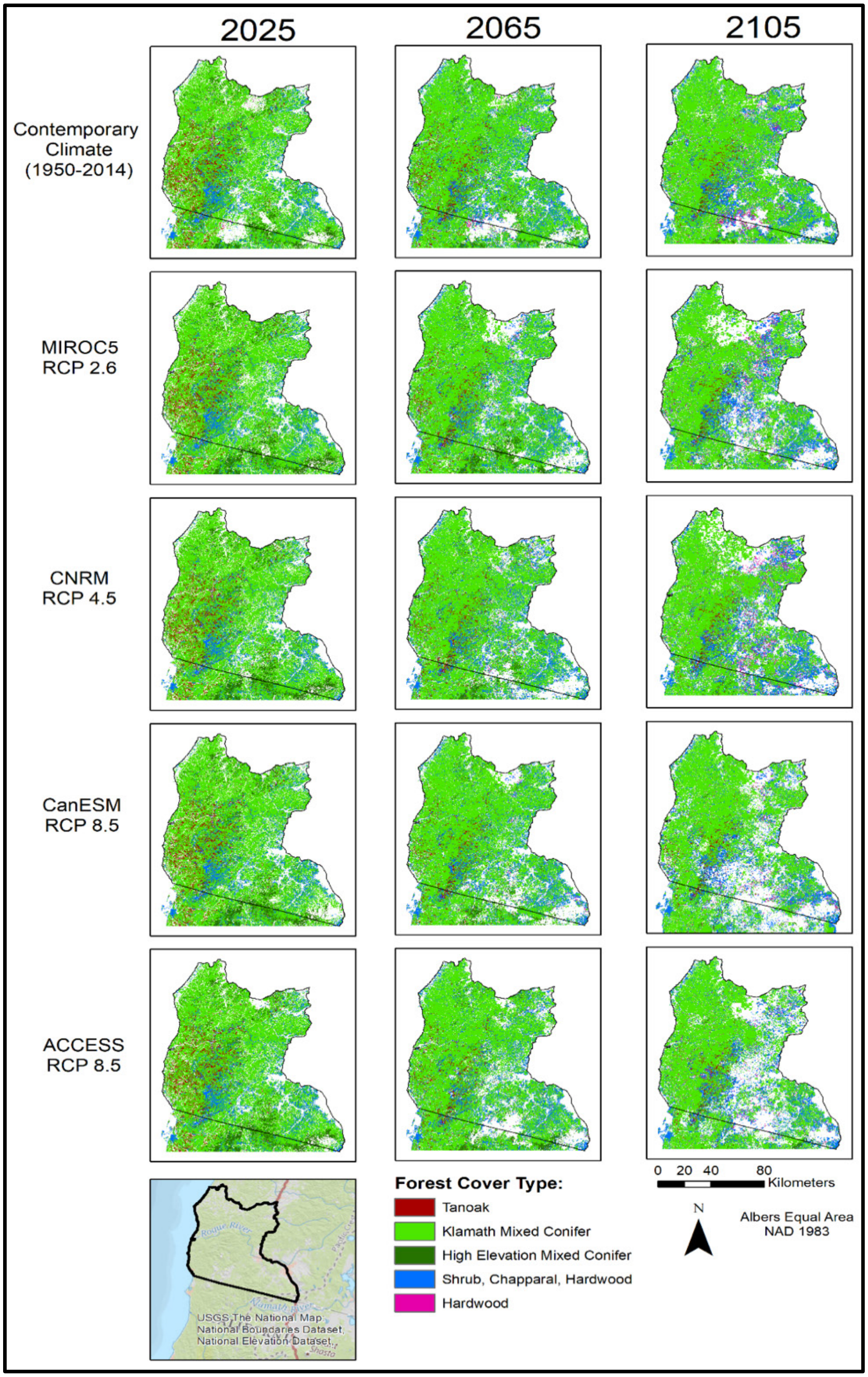

Figure 6. Projected cover of different vegetation types for future time periods under contemporary (1950–2014) and four future climate scenarios, as simulated by the LANDIS-II model for the southwest portion of the Southwest Oregon Adaptation Partnership area. From Halofsky et al. in press.

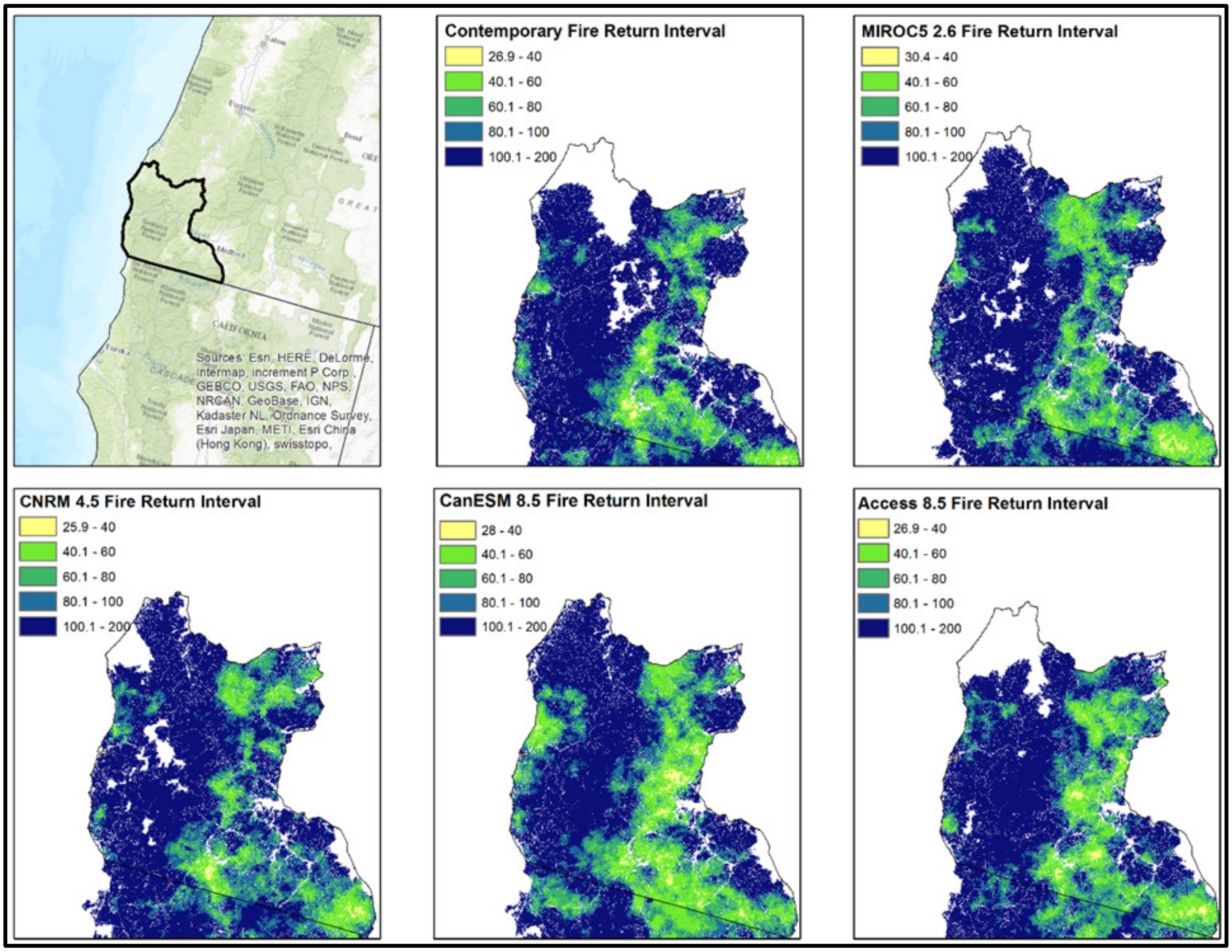

Figure 7. Projected fire return interval under contemporary (1950–2014) and four future climate scenarios over an 85-year period, as simulated by the LANDIS-II model for the southwest portion of the Southwest Oregon Adaptation Partnership area. From Halofsky et al. in press.

5. Assessment of the Effects of Climate Change on Vegetation at Oregon Caves

In Oregon Caves National Monument and Preserve (ORCA), important climatic thresholds, where major shifts in vegetative composition occur, are found at about 4,000 feet (1,220 m) and at about 5,600 feet (1,700 m) (Figures 8 and 9). Lower montane forests, or mixed evergreen forests, found at about 3,000 to 4,000 feet (900 to 1200 m), are multi-storied forests of tall conifer and broad-leaved trees (Whittaker 1960). These forests are dominated by Douglas-fir, white fir, tanoak, and Pacific madrone (Odion et al. 2013) and are also characterized as the Dry Oak-Conifer Zone.

Mid-montane forests of Douglas-fir and white fir are found at middle elevations 4,000 to 5,600 feet (1,200 m to 1,700 m) (Odion et al. 2013). Compared to the lower mountains and valleys, the mid-elevation forests, also characterized as the Cool Mixed Conifer-Oceanspray (*Holodiscus discolor*) Vegetation Zone, are cooler, wetter, have a shorter growing season, and are in a temporary winter snow zone. However, landscapes in this climatic zone have a richer flora and higher tree growth rates than in lower elevation forests. Mid-montane forests lack broad-leaved trees. Port Orford cedar can also be found in these forests, mostly on north-facing slopes and in ravines large enough to provide groundwater to Port Orford cedar roots.

Elevation zones above 5,600 feet (1,700 m) are associated with cold-tolerant ecosystems, a short growing season, and a distinct snow zone. The highest elevations have Shasta red fir at the upper limits of forests and tree islands of red fir and mountain hemlock in the subalpine.

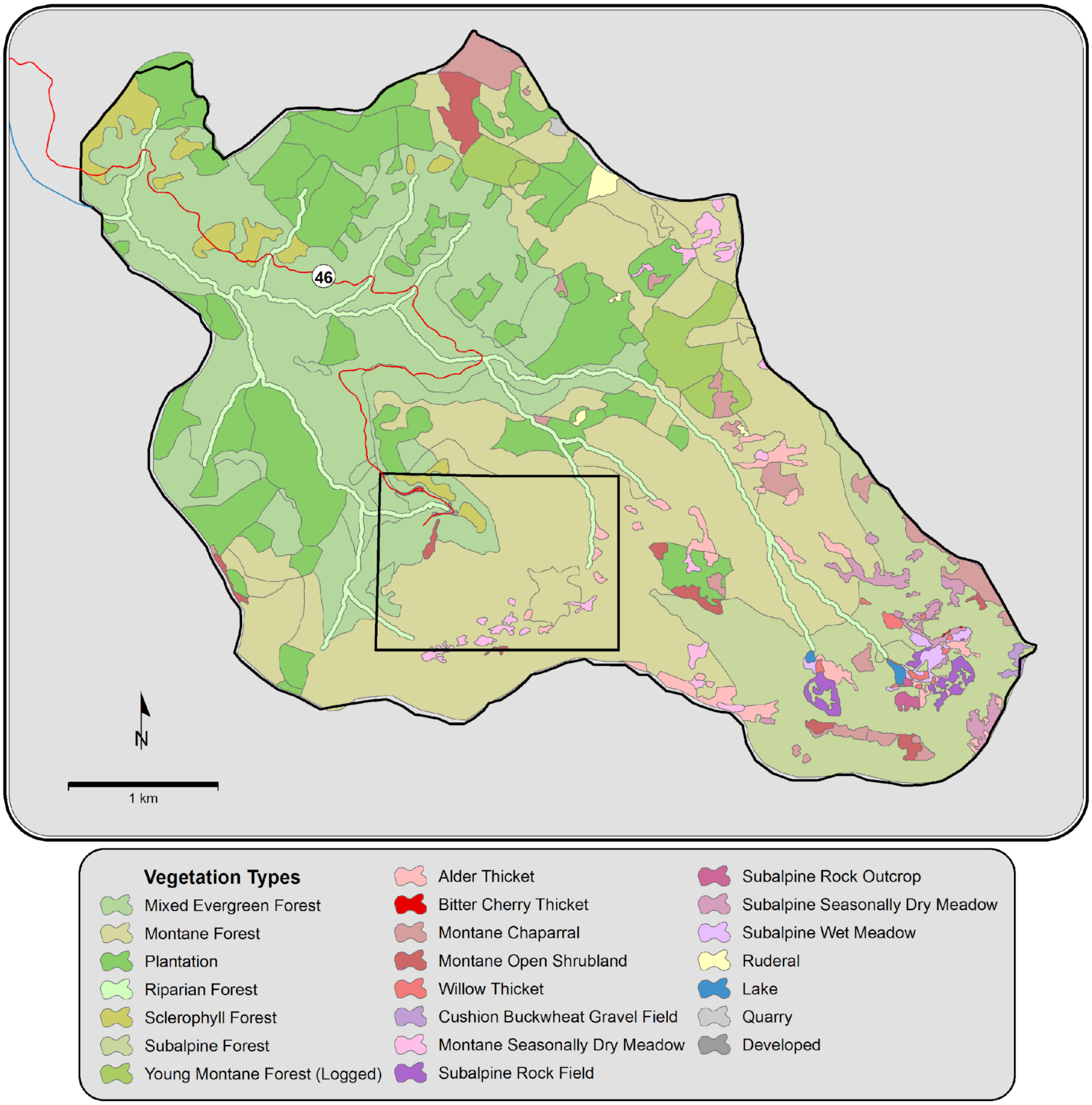

Figure 8. Vegetation types in Oregon Caves National Monument and Preserve. Mapping methods are detailed in Odion et al. 2013 (map by R. Norheim).

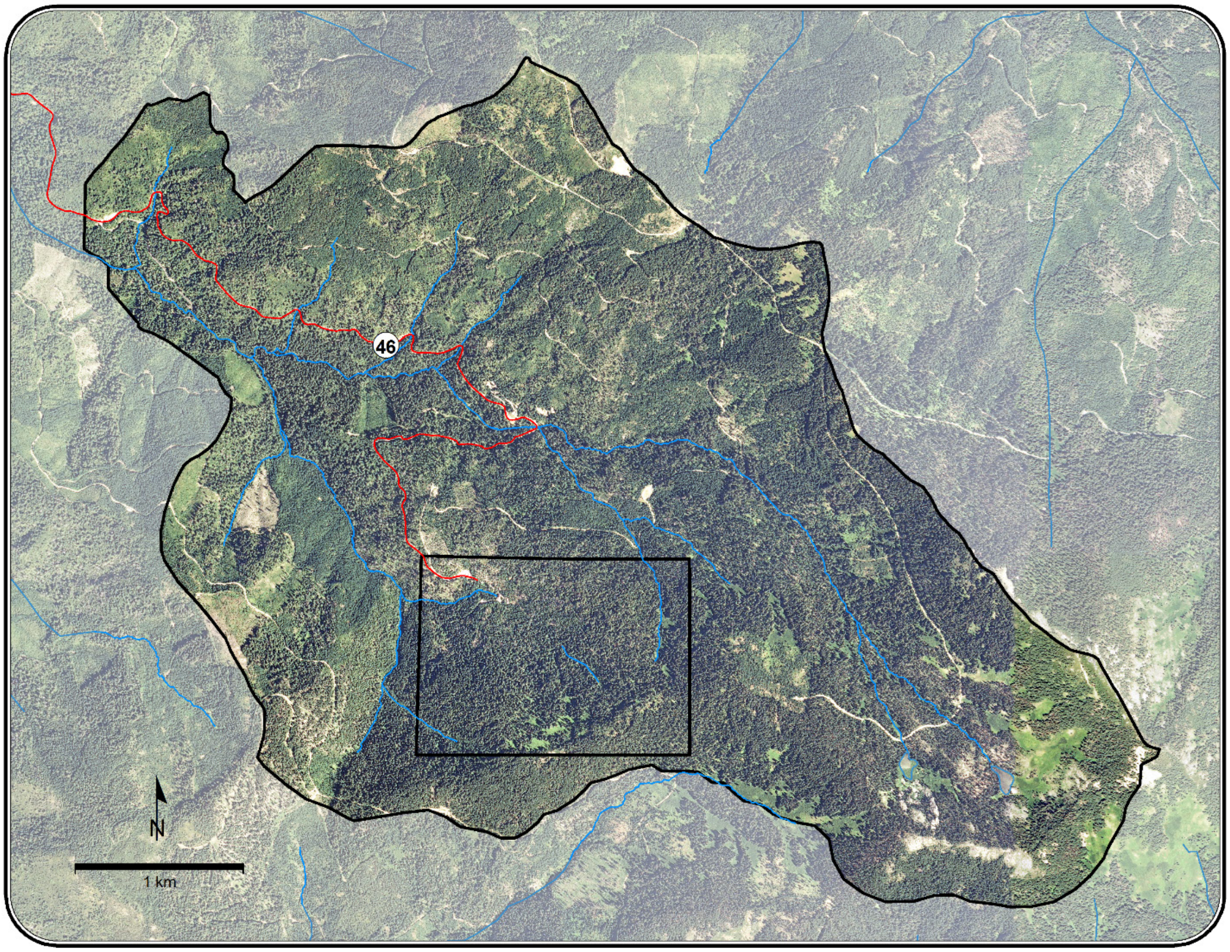

Figure 9. Aerial photograph of Oregon Caves National Monument (inner rectangular boundary) and Preserve (outer black boundary) (2011 image, National Agriculture Imagery Program). Past timber harvest is still visible in the Preserve, particularly in the northern and southwestern portions (map by R. Norheim).

Various other vegetation assemblages are found adjacent to and mixed with the dominant forest vegetation types. Here we assess the effects of climate change on the following focal plant communities that were designated by ORCA natural resource staff:

Subalpine plant communities:

- Subalpine wet meadow
- Subalpine seasonally dry meadow
- Subalpine rock field/outcrop
- Subalpine forest
- Cushion buckwheat gravel field

Riparian and wetland plant communities:

- Alder thicket
- Willow thicket
- Bitter cherry thicket

Montane shrubland communities:

- Montane chaparral
- Montane open shrubland

5.1. Subalpine Plant Communities

Local responses to climate change in the subalpine zone will vary by topography at both large and small spatial scales (Malanson et al. 2007). In addition, dominant species in the subalpine zone may experience increased competition from lower elevation species (Walther et al. 2005). If wildfire becomes more common across southwest Oregon as expected, fire occurrence may exceed the historical range of variation (see above), perhaps resulting in younger age cohorts of trees and early-successional forest structure in the long term (Kerns et al. 2017). Altered phenology of flowering bloom may affect interactions with pollinators and ultimately seed production (Dunne et al. 2003).

5.1.1. Subalpine wet meadow

This community comprises dense swards of emergent wetland vegetation occupying the shallow wet zone along the edge of Bigelow Lakes as well as non-flooded, but perennially wet or saturated areas around the basin floor upslope from Bigelow Lakes (Odion et al. 2013) (Figure 10). Soil texture is muck. Subalpine wet meadows are typically in areas where snow deposition is high and snow melt occurs in late spring or summer. Vegetation consists of a broad range of graminoid, herb, and forb species, often with shrubs around the perimeter. Meadow vegetation is diverse, varying as a function of seasonal soil moisture, topographic position, and soils, with sedges and rushes dominant adjacent to and in standing water.

Figure 10. Subalpine wet meadow in Oregon Caves National Monument and Preserve (from Odion et al. 2013).

Reduced snowpack is expected to have significant effects on wet meadows in terms of both direct water input from snowmelt and subsurface flow from adjacent slopes. Less water during the summer would alter local hydrology (Clifton et al. 2017), potentially reducing the duration and depth of standing water, increasing water temperature, and decreasing the duration of soil saturation. This could over time affect local distribution and abundance of plant species, with a potential transition to species that are currently more common in seasonally dry meadows (Dwire and Mellmann-Brown 2017). Loss of standing water could also be detrimental to aquatic fauna, especially amphibians.

Wet meadows are characterized by species tolerant of continual inundation and saturated soils. Increased frequency and magnitude of droughts would reduce the competitiveness of wet meadow species. In addition, suitable wet meadow habitat may become less common across the landscape, and only those species with propagules transported long distances by wind or animals would be able to find "safe sites" for regeneration. Shrinking wet meadow habitat could reduce the genetic diversity of some plant populations.

5.1.2. Subalpine seasonally dry meadow

This community is composed of dense, low-growing herbaceous vegetation dominated by perennial graminoids and forbs, located in the bottom of Bigelow Basin and gradual slopes and drier slopes dominated by tree and shrub vegetation or rock (Odion et al. 2013) (Figure 11). Soils are saturated during the winter months and following snow melt, but soil moisture decreases greatly during the dry summer. Vegetation is dominated by graminoids and a few herb and forb species but without sedges, often with shrubs and some trees around the perimeter. Seasonally dry meadows are adjacent to wet meadows and shrublands.

Figure 11. Seasonally dry meadow in Oregon Caves National Monument and Preserve (from Odion et al. 2013).

Reduced snowpack is expected to have significant effects on seasonally dry meadows in terms of both direct water input from snowmelt and subsurface flow from adjacent slopes. Less water during the summer would alter local hydrology (Clifton et al. 2017), potentially reducing the duration of standing water and decreasing the duration of soil saturation. This could over time affect local distribution and abundance of plant species (Dwire and Mellmann-Brown 2017). Klamath gentian (*Gentiana plurisetosa*) is a rare species in seasonally dry meadows in southwest Oregon and northern California. Like most rare species found in patchy, spatially disparate habitat, it is particularly vulnerable to directional changes in growth conditions, in this case reduced soil moisture. Although this gentian has winged seeds to facilitate dispersal, reaching suitable habitat is difficult in a montane environment.

The biggest effect of reduced duration of soil saturation would be the development of soil conditions conducive to tree germination and establishment. Many meadows already have mature, cone-bearing conifers around the perimeter, and a longer growing season facilitated by earlier snowmelt will allow trees to advance from the forest edge. This phenomenon has been well-documented for the subalpine forest-meadow interface throughout western North America, including the Pacific Northwest (Holtmeier and Broll 2005, Peterson et al. 2002, Rochefort and Peterson 1996, Woodward et al. 1995, Zald et al. 2012, Zolbrod and Peterson 1999).

5.1.3. Subalpine rock field/outcrop

This community is scattered along high points of moraines on the bottom of Bigelow Basin, comprised of mixed herb and shrub vegetation with sparse tree cover where a combination of rock and prostrate herb/shrub vegetation occurs. Cover of rock is typically >50% (Odion et al. 2013) (Figure 12). Common plant species include greenleaf manzanita (*Arctostaphylos patula*), pinemat manzanita (*A. nevadensis*), spreading phlox (*Phlox diffusa*), sulphur-flowered buckwheat (*Eriogonum umbellatum*), and other herbs and forbs. The subalpine rock field/outcrop community is interspersed with other subalpine communities, occurring in relatively small patches where rocks and boulders are prominent at the soil surface and the soil profile is shallow. These habitats have typically been subject to less historic grazing pressure from domestic livestock than subalpine meadows, and often have relatively intact native flora.

Although a warmer climate is expected to generally reduce soil moisture in summer, most species in this habitat are tolerant of seasonally dry soils. In the absence of additional stressors, it is likely that most species will persist for the foreseeable future, perhaps with lower productivity. Low soil moisture may also inhibit regeneration of trees from adjacent locations.

Figure 12. Subalpine rock field/outcrop in Oregon Caves National Monument and Preserve (from Odion et al. 2013).

5.1.4. Subalpine forest

Subalpine forests are a relatively new addition to ORCA, located entirely within the Preserve, mostly at higher elevations, but also within Bigelow Basin. This community is comprised of open to dense forest of medium to large overstory conifers, mostly with a relatively sparse understory, except in openings (Figure 13). Where the understory is well-developed, it is often due to the presence of deer oak (*Quercus sadleriana*), which occurs in dense patches but is discontinuous (Odion et al. 2013). Mountain hemlock is increasingly dominant at higher elevations, with Shasta red fir dominating the highest elevations, especially on north-facing slopes. In this environment, height growth and radial growth of trees are slow because of the short growing season (Peterson and Peterson 2001), so many trees are shrubby and compact, especially in open-grown situations where they are exposed to heavy snow loads and wind (Malanson et al. 2007). Understory species include common snowberry (*Symphoricarpos albus*), sweetcicely (*Osmorhiza chilensis*), and starry false lily of the valley (*Maianthemum stellatum*). Fire return interval is typically > 200 years.

Figure 13. Subalpine forest in Oregon Caves National Monument and Preserve (from Odion et al. 2013).

In this area, continuous forest transitions to a forest-meadow mosaic, which also includes montane shrublands, lakes and streams, and exposed rock. As such, the subalpine "forest" is often considered an ecotone where tree species are growing at their physiological limit, although this system is quite dynamic in terms of the spatial and temporal distribution and growth forms of trees.

Duration of snowpack is the primary factor controlling growth, establishment, and survival of trees and other species in the subalpine zone, although wind can limit tree distribution and growth in exposed settings, especially at higher elevations. Limiting factors vary spatially with respect to topography (north vs. south slopes, concavities vs. convexities) (Peterson 1998), affecting snow distribution, temperature, and species dominance (Millar et al. 2004, Peterson et al. 2002, Woodward et al. 1995). Patterns of vegetation reflect interactions among climatic, topographic, and biotic factors at multiple spatial scales (Zald et al. 2012). High levels of fragmentation, intermittent riparian areas and open water, and a flora that is mostly distinct from lower elevations, contribute significantly to the diversity of species and habitats for both plants and animals. In addition, subalpine areas are a popular destination for recreation because of the presence of scenic vistas, seasonal wildflower displays, and wildlife.

Despite low productivity and long duration of snowpack in the subalpine zone, these ecosystems can store a considerable amount of carbon (Sanscrainte and Peterson 2003a). Carbon increases as tree "islands" expand and aboveground biomass, belowground biomass, and soil organic matter accumulate over time (Sanscrainte and Peterson 2003b). A cold, wet environment contributes to low decomposition rates, thus promoting this accumulation. Wet soils and lake sediments also tend to accumulate and store organic matter for hundreds to thousands of years, providing long-term carbon sinks.

Subalpine forest boundaries (including treeline) are typically dynamic in space and time (Rochefort et al. 1994), and there is a delay between conditions favoring tree establishment and a noticeable change. For example, pulses of tree establishment respond to favorable periods of climate when limiting factors have been reduced (Klasner and Fagre 2002, Woodward et al. 1995), and wildfire can delay tree establishment for a century, particularly if the seed source (living trees) is downhill (Little et al. 1994). Consequently, current conditions in subalpine forest are simply a context for assessing future change.

Recent ground-based photos of the Bigelow Lakes area, and imagery from the National Agriculture Imagery Program (NAIP) and Google Earth indicate that tree establishment in meadows is gradually spreading from the edges of continuous forest. These patterns correspond to studies showing increasing tree establishment in subalpine meadows elsewhere in the Pacific Northwest (Franklin et al. 1971, Halpern et al. 2010, Rochefort and Peterson 1996, Woodward et al. 1995, Zolbrod and Peterson 1999). These changes are generally attributed to periods of lower snowpack that facilitate a longer growing season for germination and seedling growth. In addition, the "black body effect" of older trees absorbing radiation may promote snowmelt at the edge of continuous forest cover, providing suitable conditions for tree establishment.

A warmer climate is expected to affect subalpine forests in different ways (Walther et al. 2005). The extent and duration of snowpack has already decreased in the Cascade Range and are expected to decrease further with each passing decade (Mote et al. 2018). This will result in earlier snowmelt and longer growing seasons, which in turn are likely to decrease meadow habitat as conifers establish and advance from the forest edge (Holtmeier and Broll 2005, Peterson et al. 2002, Rochefort and Peterson 1996, Woodward et al. 1995, Zald et al. 2012, Zolbrod and Peterson 1999). In addition, tree growth of conifer species may increase in locations where snow severely limits the duration of the growing season (Peterson 1998, Peterson et al. 2002).

Model output from MC2 and LANDIS-II suggest that there will be a significant reduction in climatically suitable habitat for high-elevation forests, which is consistent with a previous assessment for southwest Oregon (Myer 2013). The extent and duration of snowpack have already decreased in the Cascade and Siskiyou Mountains and are expected to decrease further with each passing decade (Dalton et al. 2017, Mote et al. 2018). This will result in earlier snowmelt and longer growing seasons, which in turn are likely to (1) decrease meadow habitat as conifers establish and advance from the forest edge (Holtmeier and Broll 2005, Peterson et al. 2002, Rochefort and Peterson 1996, Woodward et al. 1995, Zald et al. 2012, Zolbrod and Peterson 1999), and (2) increase tree growth of conifer species in some locations (Peterson 1998, Peterson et al. 2002).

Dominant species in the subalpine zone may experience increased competition from species that are currently dominant at lower elevations (Briles et al. 2008, Walther et al. 2005). In MC2, subalpine forest is replaced by moist coniferous forest, suggesting that Douglas-fir, western hemlock, and white fir—species that currently occur on warmer sites in high-elevation forests—could increase in abundance in the future. Paleoecological studies similarly suggest that pines, Douglas-fir, white fir, and cedar replaced subalpine parklands in the Siskiyou Mountains during warmer and drier periods in the past (Briles et al. 2008).

Although much attention has been focused on the movement of treeline in mountains, it has rarely fluctuated more than 300 feet (100 m) during the Holocene throughout North America (Rochefort et al. 1994). In contrast, tree density and proportion of trees and herbaceous/grass species in the forest-meadow mosaic are a more dynamic component of subalpine ecosystem function, fluctuating considerably in response to decadal-to-centennial scale climatic variation (Klasner and Fagre 2002, Woodward et al. 1995) and to disturbance (Little et al. 1994). If wildfire frequency and extent in southwest Oregon increase as expected, younger age cohorts and smaller tree sizes will be more common in the long term (Kerns et al. 2017). Earlier spring snowmelt could result in a longer summer dry period, and area burned by high-severity fires may increase.

5.1.5. Cushion buckwheat gravel fields

This community is located at high elevations on coarse gravelly soil, dominated by a sparse cover of cushion plants and scattered conifer seedlings, saplings, and an occasional small tree or tree thicket (Figure 14). It is found in only one location in ORCA. Common species include cushion buckwheat (*Eriogonum ovalifolium*), cobwebby Indian paintbrush (*Castilleja arachnoidea*), and squirreltail grass (*Elymus elymoides*), with occasional Shasta red fir and incense cedar (Odion et al. 2013).

Cushion buckwheat and squirreltail grass are widespread in western North America and are typically productive even in very dry soils. The gray-green, hairy leaves of buckwheat are an adaptation to the hot, low-moisture environment near the soil surface. Squirreltail grass sprouts from the root crown and regenerates by seed following fire, although surface fuels are quite low in this community. Given the drought tolerance, morphology, and other characteristics of plants in gravel fields, most species are expected to persist for the foreseeable future in the absence of other stressors (e.g., a highly competitive nonnative species).

Figure 14. Buckwheat gravel field in Oregon Caves National Monument and Preserve (from Odion et al. 2013).

5.2. Riparian and Wetland Plant Communities

Riparian areas provide multiple ecological functions to aquatic ecosystems. Root systems of riparian plants help to maintain soil structure and stream bank stability and prevent erosion into streams (Naiman and Décamps 1997). Shade provided by riparian tree canopies reduces stream temperatures, improving habitat for cold-water species (Gregory et al. 1991). Organic matter from riparian vegetation provides food resources for aquatic organisms (Gregory et al. 1991, Naiman and Décamps 1997). Riparian areas also act as a source of large woody debris for in-stream structure and habitat.

Fire exclusion may be affecting plant composition in some riparian habitats in southwest Oregon. Historically, fire probably occurred at lower frequency in riparian areas compared to uplands (Skinner 2003). However, lack of fire, particularly in the lower montane forest zone (that historically experienced more frequent fire), may be shifting composition to favor shade-tolerant species (Messier et al. 2012). Shading by dense overstory conifers may limit the occurrence of shade-intolerant hardwood species such as alder.

Reduced snowpack in a warmer climate may have significant effects on riparian areas and wetlands. In some riparian areas, increased flooding may occur as a result of lower snowpack and increased intensity of winter precipitation events. Less water during the summer would alter local hydrology (Clifton et al. 2017), especially in the absence of adequate ground water supply (Aldous and Bah 2014), reducing the duration and depth of standing water and reducing the duration of wet soil conditions. This could affect local distribution and abundance of plant species associated with riparian areas and wetlands (Dwire and Mellmann-Brown 2017), potentially facilitating establishment of species from adjacent habitat that are more competitive in drier soils.

Wildfires generally burn with lower severity in riparian areas and affect soil to a lesser extent (Halofsky and Hibbs 2008). However, fire exclusion has resulted in denser forests in some riparian areas and adjacent uplands (Messier et al. 2012), and climate change will likely increase area burned. More frequent fire is likely to favor hardwood species and shade-intolerant conifers. More disturbance, in combination with reduced water supply, could further isolate individual riparian areas and wetlands (Ćížková et al. 2013), resulting in reduced genetic diversity of remaining populations.

5.2.1. Alder thicket

This community is found at mid and higher elevations in relatively small, discrete patches (Figure 15). Dominated by dense stands of thinleaf alder (*Alnus incana*), alder thickets are associated primarily with subalpine wet meadows and willow thickets. Additional species include red elderberry (*Sambucus racemosa*), arrowleaf ragwort (*Senecio triangularis*), bleeding heart (*Dicentra formosa*), and common cowparsnip (*Heracleum sphondylium* ssp. *montanum*) (Odion et al. 2013).

Thinleaf alder is highly productive in wetter soils but is also tolerant of relatively dry and infertile soils (it has associated bacteria on its roots that can fix nitrogen), and this drought tolerance should make it more resilient in a warmer and drier climate. Alder can sprout vigorously from the root crown or roots if the stem is killed, so it should be able to persist if fire frequency increases. If soils dry significantly at the periphery of a riparian habitat, conifers may be able to establish over decades, reducing the dominance of alder.

Figure 15. Alder thicket in Oregon Caves National Monument and Preserve (from Odion et al. 2013).

5.2.2. Willow thicket

This community is located in seasonally saturated, perennially moist areas at higher elevations, mainly meadow margins or in large patches in meadows and often on the margin of alder thickets (Figure 16). Willow thickets occur in small (1 to several acres or hectares) patches, dominated by Eastwood's willow (*Salix eastwoodiae*), with a sparse understory that includes arrowleaf ragwort (*Senecio triangularis*) (Odion et al. 2013).

In a warmer climate, lower soil moisture could create some stress for trees in late summer, although the effect would probably be less growth and productivity, rather than mortality. The biggest effects would be on the drier edges of the thickets. Assuming that Eastwood's willow sprouts from the root crown like most other willow species, it should be resilient to increased fire frequency, which is expected in a warmer climate.

Figure 16. Willow thicket in Oregon Caves National Monument and Preserve (from Odion et al. 2013).

5.2.3. Bitter cherry thickets

This community consists of dense, low-stature thickets of bitter cherry (*Prunus emarginata*) with an understory of forbs and grasses. It occurs in small stands along drier edges of seasonally dry meadows (Figure 17). It often forms a narrow band between meadow vegetation and adjacent montane or subalpine conifer vegetation, or between willow and alder thickets and adjacent montane or subalpine conifer vegetation (Odion et al. 2013).

In a warmer climate, lower soil moisture could create some stress for bitter cherry in late summer, although the effect would probably be less growth and productivity, rather than mortality. The biggest effects would be on the drier edges of the thickets. Bitter cherry sprouts following fire, so it should be resilient to increased fire frequency which is expected in a warmer climate. In high-severity fire, this sprouting ability may be a significant advantage compared to adjacent conifer forest. Conversely, conifers may advance into bitter cherry thickets during long fire-free periods.

Figure 17. Bitter cherry thicket in Oregon Caves National Monument and Preserve (from Odion et al. 2013).

5.3. Montane Shrublands

5.3.1. Montane chaparral

This community is composed of moderately dense to dense, medium-sized shrubs with greenleaf manzanita and snowbush (*Ceanothus velutinus*) being the most frequent and abundant (Figure 18). Deciduous shrub components, such as western serviceberry (*Amelanchier alnifolia*) and bitter cherry, are often common in openings, which may be associated with rock outcrops (Odion et al. 2013). Soils are often well-drained and rocky. After logging, the sclerophylls may not be as common as after fire and oceanspray may be more common. Small conifers including Shasta red fir, white fir, and Douglas-fir are typically scattered throughout.

Figure 18. Montane chaparral in Oregon Caves National Monument and Preserve (from Odion et al. 2013).

Chaparral shrubs are adapted to fire and quickly establish after fire via sprouting or a long-lived soil seed bank (Keeley 1991). Once established, shrubs impede tree seedling establishment and development of forests (Conard and Radosevich 1982, Nagel and Taylor 2005). Patches of chaparral, tens to hundreds of hectares in size, were part of historical landscapes in fire-prone forests in southwest Oregon (Lauvaux et al. 2016). It is likely that these patches have been historically maintained by relatively low-frequency (compared to the surrounding forest), but high-severity fire (Lauvaux et al. 2016). However, many areas historically dominated by shrublands have now converted to conifers in areas where fire has been excluded (Duren et al. 2012).

With increasing fire frequency and summer water deficit in a warming climate, it is likely that shrublands will expand in locations where tree regeneration is not successful (Lavaux et al. 2016, Myer 2013, Tepley et al. 2017). The LANDIS-II model projected forest loss and expansion of shrubland and chaparral (see section 4.2). MC2 also projected expansion of shrublands in hotter and drier scenarios (see section 4.1). Paleoecological studies in the region suggest that chaparral expanded during warm and dry periods in the past (Daniels et al. 2005).

As climatic water deficit increases, areas at the dry end of the present distribution of conifer forests may shift to shrublands after severe fire. Chaparral shrub species establish well in forests burned at high severity (Lavaux et al. 2016). Repeated fire could perpetuate chaparral vegetation because short intervals between severe fires and drought conditions do not allow for forest establishment (Tepley et al. 2017). However, if fire return intervals are sufficiently short, some chaparral species could be replaced by grasses and/or forbs, including invasive species.

Nearby trees provide the seed source for conversion of chaparral to forest (Lavaux et al. 2016). However, in large, high-severity fire patches, long distances to tree seed sources could slow or prevent establishment of forests. Conversion to shrubland may occur in some locations where mature forest is killed by high-severity fire; if the frequency of short-interval high-severity reburns increases, each successive fire will kill regenerating conifers and eliminate potential seed trees (Tepley et al. 2017). Drought conditions will likely further limit tree seedling regeneration on dry sites.

5.3.2. Montane open shrubland

This community consists of open-shrub vegetation on coarse-textured, very rocky soils, often along ridges (Figure 19). Low soil moisture is chronic (e.g., south-facing aspects, shallow soils) and trees rarely become established. Species composition is generally less diverse, lower density, and lower stature than in montane chaparral. Similar to montane chaparral, montane open shrubland is dominated by drought-tolerant shrub species including greenleaf manzanita and snowbush, with a diversity of understory species and other shrubs, including western serviceberry and oceanspray (Odion et al. 2013).

In a warmer climate, montane open shrubland can be expected to persist, because it contains plant species that tolerate low soil moisture, although shrub density could decrease in drier locations. The extent of this community could increase in some locations, perhaps through a transition from montane chaparral. Increasing frequency and extent of both drought and fire will also contribute to the maintenance of shrubland species, making it unlikely that tree species will be able to regenerate.

Figure 19. Montane open shrubland in Oregon Caves National Monument and Preserve (from Odion et al. 2013).

6. Adapting Vegetation Management to Climate Change

Based on the vulnerability assessment information presented in Halofsky et al. (in press), and on documented adaptation principles (e.g., Millar et al. 2007, Peterson et al. 2011, Swanston et al. 2016), adaptation options for southwest Oregon were identified by participants in a workshop that took place in Grants Pass in April 2018. Workshop participants identified strategies, or general approaches, for adapting vegetation management to climate change, then identified on-the-ground tactics associated with each adaptation strategy and considered the implementation of those tactics.

Adaptation options focused on key climate change sensitivities for vegetation, including (1) altered disturbance regimes (e.g., drought, fire, insects), (2) potential for mortality events and regeneration failures, (3) altered species distribution, (4) stresses on endemic and relict species, and (5) phenological mismatches between flowering plants and pollinators. The adaptation options considered most relevant for ORCA vegetation in general are summarized in Table 1. Adaptation options considered most relevant for riparian and wetland systems (Dwire and Mellmann-Brown 2017), a prominent component of ORCA focal communities, are summarized in Table 2.

The magnitude of changes in current management approaches that are dictated by projected effects of climate change will vary by location, vegetation assemblage, and species. Climate-related responses will manifest in different ways in different landscapes, requiring that expert judgment be exercised for local landscapes, regardless of what broad-scale models may say. An evaluation of uncertainty will be useful for documentation in planning and project management that consider climate change effects.

Table 1. General vegetation adaptation options for southwest Oregon, developed during a workshop in the spring of 2018.

Sensitivity to climate change	Adaptation strategy	Adaptation tactic
Hotter and drier conditions will increase forest drought stress, lead to reduced forest productivity, and increase susceptibility to secondary stressors such as disease, insects, and hydraulic failure.	Increase drought resilience and forest vigor.	Manage vegetation density to reduce soil moisture stress. Leverage disturbance events to restore ecosystem function. Be ready for rapid response. Reduce density of post-disturbance tree planting. Consider using genetically improved seedling stock. Plant resistant species or genotypes where species-specific insects or pathogens are a concern.
	Maintain and enhance forest productivity regardless of tree species; focus on functional ecosystems and processes.	Manage species densities to maintain tree vigor and growth potential. Prepare for species migration by managing for multiple species across large landscapes.

Table 1 (continued). General vegetation adaptation options for southwest Oregon, developed during a workshop in the spring of 2018.

Sensitivity to climate change	Adaptation strategy	Adaptation tactic
Hotter and drier conditions will increase forest drought stress, lead to reduced forest productivity, and increase susceptibility to secondary stressors such as disease, insects, and hydraulic failure. (continued)	Increase forest landscape resilience to large and extensive insect or pathogen outbreaks.	Design forest gaps that create establishment opportunities. Increase diversity of patch sizes. Consider planting desired species (assisted migration) rather than relying on natural regeneration and migration.
	Promote diversity of forest age and size classes.	Diversify large contiguous areas of single age and size classes.
	Increase resistance to invasion by insects, pathogens, and invasive plants.	Assertively apply early detection, rapid response to limit nonnative insects.
Climate change will likely lead to altered disturbance regimes.	Promote conditions to facilitate response or transition while maintaining function.	Monitor disturbance events. Identify trends and use information to improve forecasting. Align planning process to proactively address shifting disturbance regimes. Promote social acceptance and public awareness of management.
Climate change will increase the potential for vegetation mortality during drought.	Promote regeneration of desired species to ensure adaptation of progeny to future conditions.	Thin dense vegetation where appropriate; encourage natural regeneration; plant seedlings where appropriate.
	Use judicious assisted migration of genotypes where appropriate	Relax seed zone guidelines to a variety of genotypes rather than just one.
Increased temperatures and lower snowpack will result in more fire (larger aerial extent and more high-severity patches).	Plan and prepare for more area burned.	Anticipate opportunities to use wildfire for resource benefit. Plan post-fire response for large fires. Manage forest restoration for future range of variability.
	Increase resilience of existing vegetation by reducing hazardous fuels and density of dominant vegetation.	Consider using prescribed fire where scientific evidence supports change to more frequent fire regime. Increase interagency coordination. Use regeneration and planting to influence vegetation structure.
	Increase resilience through post-fire management.	Consider climate change in post-fire rehabilitation. Determine where native seed may be needed for post-fire planting. Anticipate greater need for seed sources and propagated plants.

Table 1 (continued). General vegetation adaptation options for southwest Oregon, developed during a workshop in the spring of 2018.

Sensitivity to climate change	**Adaptation strategy**	**Adaptation tactic**
Higher temperatures may increase stress for some species.	Protect rare and disjunct species.	Plant and encourage regeneration of rare and disjunct species in appropriate locations.
	Protect subalpine forests by restoring forests at lower elevations, thus reducing spread of large crown fires.	Create targeted fuel breaks at strategic landscape locations. Thin dry forests to densities low enough to reduce fire intensity and spread.
	Accelerate restoration of subalpine forests where appropriate.	Increase the availability of nursery stock and seed for tree species in cold upland and subalpine forests.
Disturbances will change large-scale patterns, including structure, and species distribution and abundance.	Increase knowledge of patterns, characteristics, and rates of change in species distributions.	Expand long-term monitoring programs.
	Create landscape patterns that are resilient to past and expected future disturbance regimes.	Continue research on expected future disturbance regimes; evaluate potential transitions and thresholds. Improve communication across boundaries. Manage for diversity of structure and patch size.
Climate change may threaten endemic, refugial, and relict species.	Conserve genetic and phenotypic diversity and increase species resilience to conditions based on climate change projections.	Develop a gene conservation plan for ex situ collections for long-term storage. Identify areas important for in situ gene conservation. Maintain a seed inventory with high-quality seed for a range of species, particularly species that may do well in the future under hotter and drier conditions. Use seeding of native plant species in areas with nonnative species. Prepare for species migration by managing for multiple species across large landscapes. Plant and encourage regeneration of rare and disjunct species in appropriate locations. Collect seed that will cover a wide range of seed zones and species. Modify genetic movement guidelines to allow more flexibility. Identify conditions that provide climate refugia for rare species (e.g., slope, aspect, soil type) and restore refugia where there is need.

Table 1 (continued). General vegetation adaptation options for southwest Oregon, developed during a workshop in the spring of 2018.

Sensitivity to climate change	**Adaptation strategy**	**Adaptation tactic**
Climate change will increase nonnative species establishment.	Minimize establishment and spread of nonnative and invasive species.	Implement early detection and rapid response for nonnative species treatment. Coordinate invasive species management, funding, and support between agencies. Include invasive species prevention strategies in all projects. Inventory regularly to detect new populations and species. Maintain permits for aggressive treatment of invasive species (e.g., burning and herbicides). Emphasize use of plant species for restoration that will be robust in a warmer climate. Plant genetically adapted species from appropriate seed zones. Promote weed-free seed. Prevent nonnative plant introductions during projects, fire response, and post-fire activities. Ensure weed-free policies are included in planning documents. Coordinate weed-free seed standards and regulations among agencies.
Earlier flowering may lead to phenological mismatch such that pollinators are not present when flowering begins.	Maintain and increase genetic diversity. Try to minimize mismatches in timing between flowering and pollinator timing.	Create different microsites to change phenology within a species (e.g., sun versus shade, wet versus dry) Thin trees to increase and prolong snow cover. When planting, use diverse genetic material within a species as well as a variety of different species.

Table 2. Vegetation adaptation options for riparian and wetland systems (adapted from Dwire and Mellmann-Brown 2017).

Sensitivity to climate change	Adaptation strategy	Adaptation tactic
Shifts in hydrologic regime include changes in timing and magnitude of flows, lower summer flows, and higher, more frequent winter peak flows. Reduced snowpack will decrease water supply during growing season.	Maintain appropriate densities of native species, propagate drought tolerant native species.	Plant species that have a broad range of moisture tolerance. Eradicate and control invasive species where possible. Remove infrastructure where appropriate (e.g., camp sites, trails).
	Maintain or restore natural flow regime.	Develop integrated tactics to maintain or restore natural flows. Restore riparian areas and beaver populations to maintain summer baseflows. Reconnect and increase off-channel habitat and refugia inside channels. Revegetate areas that have been damaged by livestock and people.
Reduced snowpack will decrease water supply during growing season, thus reducing productivity in groundwater-dependent systems, including springs and wetlands.	Manage for resilience of springs and wetlands by including uplands.	Consider impacts and potential benefits of vegetation management treatments (e.g., prescribed fire). Protect groundwater recharge areas.
	Manage water to maintain springs and wetlands, improve soil quality and stability.	Decommission roads and trails where appropriate, reduce road connectivity. Maintain water on site through water conservation techniques (e.g., float valves, diversion valves) Use no more water than is sufficient to meet the intended purpose (e.g., campground). Include implementation and effectiveness monitoring to evaluate water conservation projects.
Reduced snowpack will decrease water supply during growing season and lead to more variable streamflow, thus reducing productivity in riparian systems.	Reduce stresses from conifer encroachment, past livestock grazing, and recent ungulate browsing.	Consider riparian fuel reduction strategies in subalpine areas, including small-scale fuel breaks. Rehabilitate areas degraded by past livestock grazing.

7. Conclusions

Climate change is expected to affect the vegetation communities of the Oregon Caves National Monument and Preserve in myriad ways. Increased temperatures, soil moisture deficits, and wildfire may affect species composition and structure of vegetation in ORCA. Increased temperatures and reduced snowpack may lead to a reduction in climatically suitable habitat for high-elevation species. Dominant species in the subalpine zone may experience increased competition from species that are currently dominant at lower elevations, including Douglas-fir, western hemlock, and white fir. Earlier snowmelt and longer growing seasons are likely to increase tree growth but will also lengthen the summer dry period. Area burned by high-severity fires may increase.

Moist and mesic forests in southwest Oregon will likely continue to be dominated by Douglas-fir and other early-seral species with increasing temperature and disturbance rates. Fire- and drought-intolerant species, including western hemlock, Pacific silver fir, and western redcedar, are likely to decrease in abundance in moist forests, and white fir may decrease in mesic forests. Mesic forests could transition to more xeric forest limited by summer drought stress and maintained by more frequent fire.

Shifts from dry forest to woodlands or shrublands may occur in the driest portions of the current dry forest range. Drought stress and large, high-severity fire patches may impede forest development in some locations. Conversion to shrubland would likely occur with increasing loss of mature forest in high-severity fire and increasing frequency of short-interval, high-severity reburns will likely kill more regenerating conifers and potential seed trees with each successive fire. Growth of dry forest species will likely decrease, whereas growth of subalpine species may increase. Tree mortality may also increase in some locations because of the interacting effects of drought, disturbance, pathogens and insects.

Overall, topographic heterogeneity and varied microclimates in ORCA may create climate refugia and allow for species persistence (Briles 2017). However, where fuels have accumulated as a result of fire exclusion, forests and woodlands are at risk of high-severity fire. High-severity fire may also affect adjacent vegetation communities with historically low fire frequencies. Second-growth forests may be particularly vulnerable to drought, fire, and insect outbreaks because of their high density and low species and structural diversity. Interactions among multiple stressors may result in rapid change in vegetation composition and structure, and in ecosystem services provided (Millar and Stephenson 2015).

Using active management to reduce risk of high-severity fire and increase resilience to climate change is an important adaptation strategy for ORCA, as it is for all of southwest Oregon. Other adaptation strategies and tactics are focused on helping ecosystems and communities transition to new conditions, promoting plant species or community types of concern, and reducing the effects of existing non-climatic stressors on ecosystems. Multiple values, ecosystem services, and ecosystem stressors (e.g., fire, insects and disease, and invasive species) need to be considered when determining the type, scale, and intensity of adaptation actions (Halofsky et al. 2016).

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