Chapter 6: Effects of Climatic Variability and Change on Upland Vegetation in the Blue Mountains

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Introduction

The Blue Mountains ecoregion (BME) extends from the Ochoco Mountains in central Oregon to Hells Canyon of the Snake River in extreme northeastern Oregon and adjacent Idaho, and then north to the deeply carved canyons and basalt rimrock of southeastern Washington (see fig. 1.1 in chapter 1). The BME consists of a series of mountain ranges occurring in a southwest to northeast orientation, allowing the BME to function ecologically and floristically as a transverse bridge between the Cascade Mountains province to the west, and the main portion of the middle Rocky Mountains province to the east.

Powell[2] defines six vegetation zones within the BME, which range from low-elevation grasslands to high-elevation alpine areas, and Johnson (2004), Johnson and Clausnitzer (1995), Johnson and Simon (1987), and Johnson and Swanson (2005) described upland plant associations. The lowest elevation plains zone contains grasslands and shrublands because moisture is too low to support forests except along waterways. The foothills zone is usually dominated by western juniper (*Juniperus occidentalis* Hook.), often with a mixture of curl-leaf mountain-mahogany (*Cercocarpus ledifolius* Nutt.) and antelope bitterbrush (*Purshia tridentata* [Pursh] DC.) shrublands. In the northern Blue Mountains, the foothills zone generally supports tall shrublands (with, for example, western serviceberry [*Amelanchier alnifolia* [Nutt.] Nutt. ex M. Roem.], black hawthorn [*Crataegus douglasii* Lindl.], and western chokecherry [*Prunus virginiana* var. *demissa* [Nutt.] Torr.]), rather than western juniper, curl-leaf mountain-mahogany, and antelope bitterbrush. Located just above the western juniper woodlands is the lower montane zone, which

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[2] **Powell, D.C. 2012.** Active management of dry forests in the Blue Mountains: silvicultural considerations. White Paper F14-SO-WP-Silv-4. Unpublished report. Pendleton, OR: U.S. Department of Agriculture, Forest Service, Umatilla National Forest.

contains dry conifer forests characterized by ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), and grand fir (*Abies grandis* [Douglas ex D. Don] Lindl.). Although some of these forests are mixed conifer, many of the warm, dry forests are composed almost exclusively of ponderosa pine.

Warm, dry forests tend to be the most common forest zone in the Blue Mountains, and because they occur at lower and moderate elevations, they have a long history of human use both for commodity purposes (e.g., livestock grazing) and as an area where effective fire exclusion occurred and led to well-documented changes in species composition, forest structure, and stand density. The upper montane zone includes moist forests characterized by Douglas-fir, grand fir, and subalpine fir (*A. lasiocarpa* [Hook.] Nutt.). The upper montane zone may be very narrow or nonexistent in the southern portion of the Blue Mountains. High elevations support a subalpine zone with Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), subalpine fir, and whitebark pine (*Pinus albicaulis* Engelm.) or an alpine zone near mountain summits where trees are absent. Subalpine and alpine environments are mostly confined to the high ridges of the Strawberry Mountains, the Elkhorn Crest, the Wallowa Mountains, and the Seven Devils in eastern Oregon and western Idaho.

This chapter assesses potential climate change effects on upland vegetation in the BME using information from the literature and output from simulation models. The focus is on the BME in general, but emphasis is placed on the Malheur, Umatilla, and Wallowa-Whitman National Forests (see fig. 1.1 in chapter 1). First, we review information sources that were used as the basis of this assessment. Then we present the vegetation assessment, which is organized around an established vegetation classification system, followed by a brief conclusion.

Altered disturbance regimes will likely be the most important catalyst for vegetation change.

Understanding Climate Change Effects

Climate change is expected to profoundly alter vegetation structure and composition, terrestrial ecosystem processes, and the delivery of important ecosystem services over the next century. Climate influences the spatial distribution of major vegetation biomes, the abundance of species and communities within biomes, biotic interactions, and the geographic ranges of individual species. Climate also influences the rates at which terrestrial ecosystems process water, carbon, and nutrients and deliver ecosystem services like fresh water, food, and biomass. Finally climate influences disturbance processes that shape vegetation structure and composition, and altered disturbance regimes will likely be the most important catalyst for vegetation change (box 6.1). Climate-induced vegetation changes have important

Box 6.1—Ecological Disturbance and Climate Change

Ecological disturbances include fires, floods, windstorms, and insect outbreaks, as well as human-caused disturbances such as forest clearing and the introduction of nonnative species. Disturbances are part of the ecological history of most forest ecosystems, influencing vegetation age and structure, plant species composition, productivity, carbon storage, water yield, nutrient retention, and wildlife habitat. Natural disturbances, which are influenced by climate, weather, predisturbance vegetation conditions, and location, can have profound and immediate effects on ecosystems across large spatial and temporal scales.

Climate (and weather at a shorter time scale) influences the timing, frequency, and magnitude of disturbances in any particular location. For example, a 1-year drought may not have significant direct effects on a forest, but it can reduce tree resistance to insect attack and can desiccate living and dead vegetation sufficiently to increase fire hazard. As climate continues to warm during the 21st century, the most rapidly visible and significant short-term effects will be caused by altered disturbance, often occurring with increased frequency and severity. Increased disturbance will be facilitated by more frequent extreme droughts, amplifying conditions that favor wildfire, insect outbreaks, and invasive species (Adams et al. 2010, Allen et al. 2009, Anderegg et al. 2012). The type and magnitude of disturbances will differ regionally and will pose significant challenges for resource managers to mitigate damage to resource values and transition systems into new disturbance regimes.

A warmer climate will cause an increase in the frequency and extent of wildfire in most dry forest and shrubland ecosystems (e.g., Westerling et al. 2006, 2011). By around 2050, the annual area burned in most of the Western United States is projected to be at least two to three times higher than it is today (Littell et al. 2010; Littell, n.d. cited in Ojima et al. 2014; McKenzie et al. 2004). The Blue Mountains ecoprovince is also expected to experience increased area burn by the mid 21st century (fig. 6.1). Recent research shows that the occurrence of large fires in the Western United States has increased since around 1980 (Dennison et al. 2014). Many dry forests that have not burned for several decades have high fuel accumulations, and initial fires may cause uncharacteristic tree mortality compared to low levels of mature tree mortality associated with historical surface-fire regime. If these areas recover as forested ecosystems, recurrent fires (if allowed to burn, and are not suppressed) may more closely resemble the frequency characteristic of presettlement, low-severity fire regimes. **However, in the driest portions of the Blue Mountains, it is possible that these areas will not recover to forested conditions, and uncharacteristic fires combined with climatic warming could initiate a transition to shrub- or herb-dominated ecosystems.**

Critical thresholds in ecosystem structure and function may be exceeded in a warmer climate. Warmer temperatures may increase the potential for insect and disease outbreaks, particularly as a transient response in colder temperate zones where insect and pathogen vigor has been limited by suboptimal temperatures (Bentz et al. 2010). Higher warm-season temperatures should also increase growth rates for temperate insect herbivores, although the rate of increase will vary by species (Bale et al. 2002). For some species, faster growth rates and reduced development time could enhance juvenile survivorship by reducing predation rates during the larval and nymphal feeding stages (Bernays 1997, cited by Bale et al. 2002). Increased growth

Box 6.1 (continued)

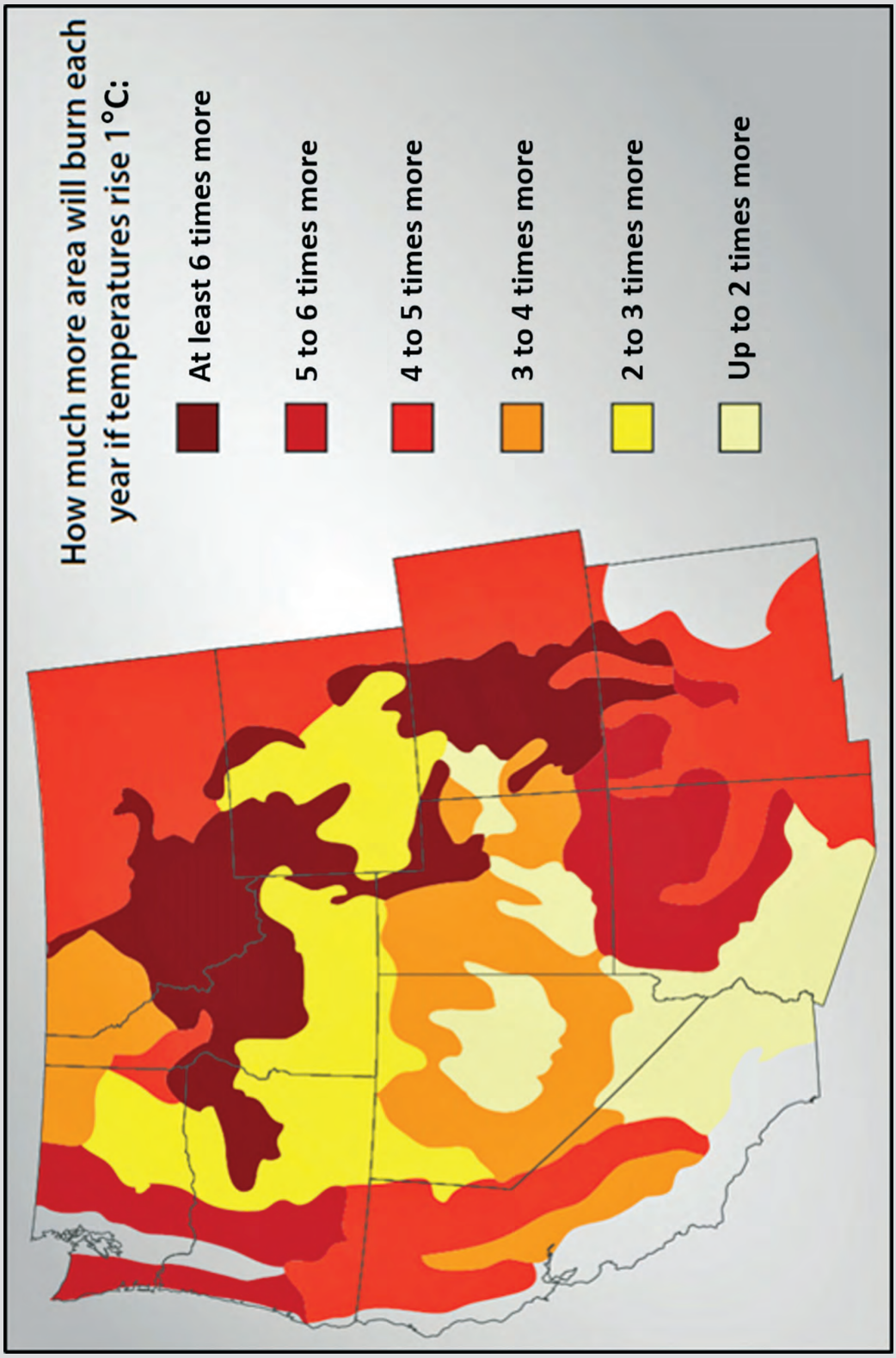

Figure 6.1—Projected increase in area burn by wildfire as associated with a mean annual temperature increase of 1 °C, shown as the percentage change relative to the median annual area burned during 1950–2003 (Littell, n.d. cited in Ojima et al. [2014]). Results are aggregated to ecoprovinces of the Western United States based on Bailey (1995).

Box 6.1 (continued)

rates could reduce generation times for some species, which could significantly increase population growth rates (Bale et al. 2002, Mitton and Ferrenberg 2012). Increasing population success increases the potential for insect outbreaks to develop, although outbreaks can be limited by host and predator constraints as well (Bale et al. 2002, Boone et al. 2011). Increasing temperatures may also facilitate migration of insects and diseases toward higher elevations and latitudes (Bale et al. 2002, Bentz et al. 2010). Similarly, species ranges could contract at lower elevations and latitudes if warm-season temperatures exceed tolerance levels during the juvenile (or other) growth stages (Bale et al. 2002). Depending on the future scenario, some research indicates that western spruce budworm (*Christoneura occidentalis* Freeman) defoliation in Oregon would decrease under climate change scenarios that do not have an increase in both temperature and precipitation (Williams and Liebhold 1995). Warmer temperatures could also increase host stress levels and the ability to defend against insect attacks (Boone et al. 2011).

The recent proliferation of mountain pine beetles (*Dendroctonus ponderosae* Hopkins, 1902) in lodgepole pine forests in western North America is a good example of how a warmer climate can propagate widespread disturbance (Bentz et al. 2009, Kurz et al. 2008, Raffia et al. 2008). **Mountain pine beetles have affected 36 million ha of forests in the Western United States and British Columbia over the last two decades**, largely as a consequence of increasing temperatures in mostly older, low-vigor forests. Both wildfire and insect outbreaks can rapidly "clear the slate" in landscapes, initiating a regeneration phase in which species will compete, allowing for a potential change in dominant species. A rapid change in dominant vegetation suggests that a threshold has been crossed, at least temporarily, and that new conditions established in a warming climate may be difficult to reverse. Higher atmospheric carbon dioxide levels could reduce insect impacts by increasing carbon availability for defenses in plants and reducing substrate quality in host plants (Stiling and Cornelissen 2007). However, the inhibitory effects of increasing carbon dioxide concentrations may be offset by the stimulatory effects of warmer temperatures on insect activity (Zvereva and Kozlov 2006).

Interacting disturbances and other stressors, termed stress complexes, are a normal component of forest ecosystems, affecting species composition, structure, and function. Altering one particular factor can potentially magnify the effects of other stressors, leading to a rapid and possibly long-lasting change in forest ecosystems. The effects of disturbance across large geographic areas are especially pronounced where forest regeneration is slow or delayed, leading to a potential change in dominant vegetation. A warmer climate is expected to alter and often exacerbate the effects of stress complexes (McKenzie et al. 2009). To examine the intersection of multiple stressors that affect Oregon and Washington, Kerns et al. (n.d.) examined spatial data regarding wildfire potential, insect and disease risk, projected urban and exurban development, and a warmer climate. They mapped where these stressors occur in concentrated places on the landscape and where they occur in combination (Kline et al. 2013) (fig. 6.2). **For the Blue Mountains, insect and disease risk is the most spatially extensive stressor.**

Forest resource managers are tasked with assessing the risk of disturbances and opportunities for mitigation. Most federal lands have fire management plans that describe current fuel conditions, potential fire occurrence, likely effects of wildfire,

Box 6.1 (continued)

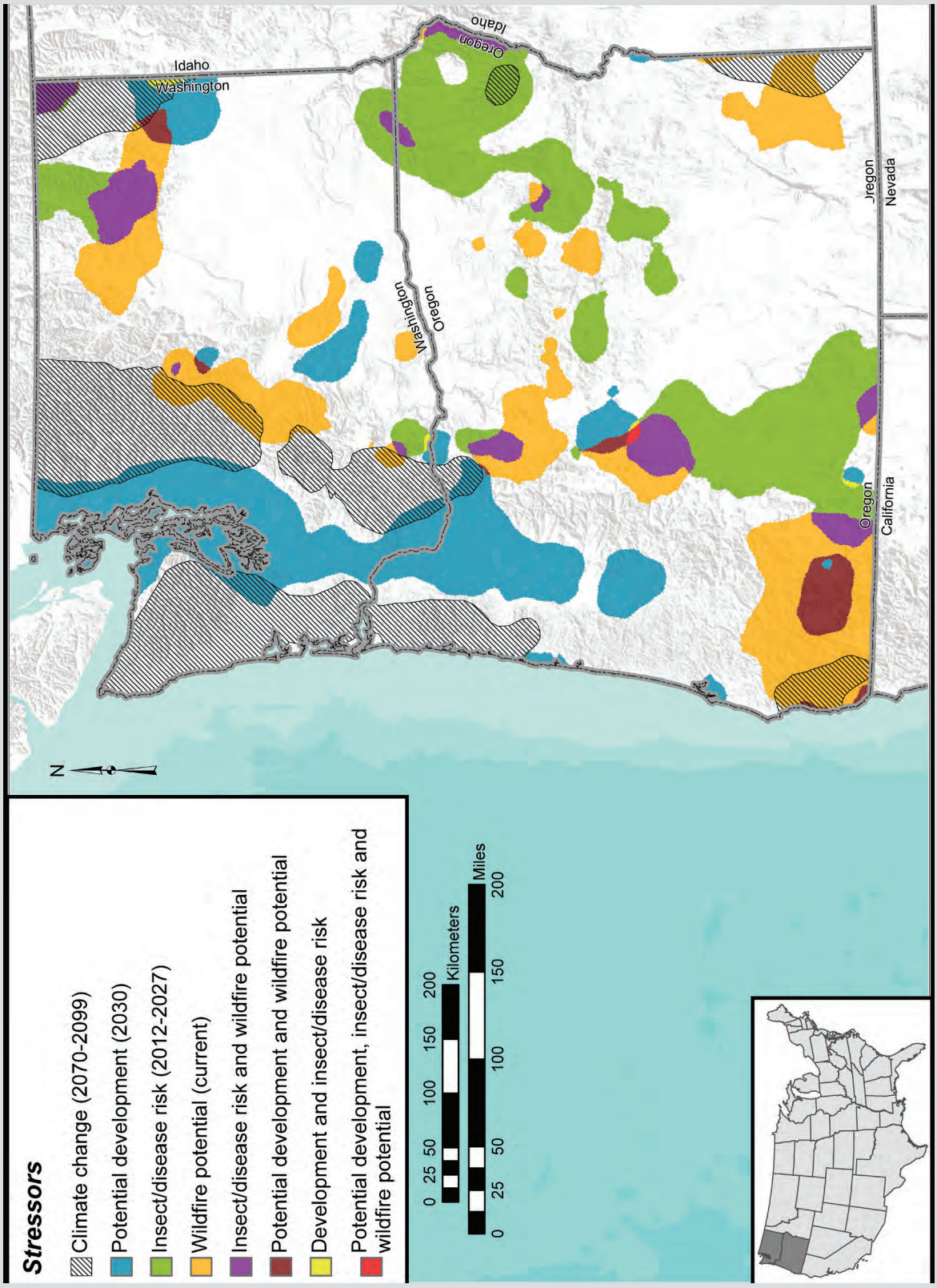

Figure 6.2—Results from a neighborhood analysis for concentrated exposure across Oregon and Washington to four stressors and their combinations.

Box 6.1 (continued)

fire suppression strategies, and often postfire activities designed to reduce secondary damage such as erosion. Similarly, plans assessing the management of insects and nonnative plants evaluate the risk of their occurrence as the basis for developing appropriate responses. In some cases, the effectiveness of predisturbance and postdisturbance actions may be limited. For example, many nonnative species are so well established that it is not feasible to remove them from a particular location; therefore, prevention or rapid response is usually the best approach. Following large, intense wildfires, it may be impractical and expensive to install erosion control across a mountainous landscape with minimal access. Active management on certain public lands—wilderness, riparian habitat conservation areas, designated old growth, lands with endangered species—may be severely restricted owing to regulations or lack of social license, thus limiting mitigation of known risks.

Disturbances, which have always been a dominant influence on the dynamics of forest ecosystems in western North America, will be even more important if climate change leads to increased frequency and magnitude of extreme weather. In most cases, it will be advisable to develop strategies that allow us to live with increasing disturbance and stress in forests, especially as the climate continues to warm. Active management and planning in anticipation of these changing conditions can reduce risk and the severity of short- and long-term hazards.

implications for wildlife habitat, biodiversity, hydrology, future disturbance regimes, ecosystem services, and the ability of ecosystems to absorb and sequester carbon from the atmosphere.

Several information sources are useful for assessing potential climate change impacts on vegetation and future forest composition and structure, including long-term paleoecological records; evidence from climate, carbon dioxide, and other experimental and observational studies; and model predictions for the future. We use these sources to conduct our assessment of potential effects of climate change on vegetation. When different lines of evidence are in conflict, we often rely on our ecological and local knowledge and derived logical inference to weigh different lines of evidence. For example, we emphasize the importance of interactions between climate change and disturbance, which many agree is likely to be the most important driver of change in the next few decades (see box 6.1). However, these disturbances are not included in many of the models that we discuss. In addition, paleoecological evidence or empirical evidence from highly relevant studies is often given greater weight than model output.

Paleoecological Records

Paleoecology uses data from some form of preserved plant material, or fossils and subfossils, to reconstruct vegetation of the past. More recent records (e.g., the Holocene or last 11,700 years) are commonly reconstructed by using plant materials

such as pollen and phytoliths. For example, pollen grains that are washed or blown into lakes can accumulate in sediments and provide a record of past vegetation. Different types of pollen in lake sediments reflect the vegetation that was present around the lake and, therefore, the climate conditions favorable for that vegetation. Phytoliths are morphologically distinct silica bodies from plants that can exist in soil, sedimentary deposits, or in archaeological material. In addition, much historical vegetation information can be obtained from biological remains in nests and middens (waste accumulation) created by several species of birds and small mammals, and they may be preserved for thousands of years, providing a detailed fossil record of past environmental conditions (Betancourt et al. 1990, Rhode 2001).

Global paleoecological records indicate that during warm periods of the past, tree species tended to move poleward and upward in elevation.

Global paleoecological records indicate that during warm periods of the past, tree species tended to move poleward and upward in elevation. A species "leading edge" is fundamentally important under global change, as it is commonly accepted that range expansions depend mostly on populations at the colonization front. The leading edge is also seen as controlled by rare long-dispersal events followed by exponential population growth (Hampe and Petit 2005). Unfortunately, existing environmental reconstructions of the Pacific Northwest are based mostly on pollen records obtained from lakes in forested areas west of the Cascade Range, as fewer of these sites exist in the the dry interior of the region. However, important insights can be derived from the existing literature (Blinnikov 2002, Hansen 1943, Mehringer 1997, Whitlock 1992, Whitlock and Bartlein 1997). In particular, as recent glaciation waned during the early Holocene period, a warmer and drier climate than today is inferred (Anderson and Davis 1988). Thus, inferred vegetation during the early to mid-Holocene (9,000 to 6,000 BP) is used in this assessment as one line of evidence with respect to potential future changes. However, early and mid-Holocene conditions may be drier than current projections from global climate models (GCMs).

The rate of future climate change may be faster than any period in the Holocene record (500 to 1,000 years) (Whitlock 1992). Hansen (1943) reports on pollen profiles from a peat deposit located at Mud Lake, one of the Anthony Lakes that lie near the crest of the Blue Mountains. The site is at 2134 m elevation in a subalpine forest. The author reports on a cursory survey of the area surrounding the peat deposit: 50 percent lodgepole pine (*Pinus contorta* Douglas ex Loudon), 25 percent alpine fir (assumed to be subalpine fir), and 25 percent Engelmann spruce. Two peat cores were removed above a layer of impenetrable ash. Hansen states that they probably represent most, if not all, of the post Pleistocene (Holocene), but no radiocarbon dates were obtained. The source of the ash layer is unknown. A rapid gain in ponderosa pine pollen roughly mid-profile is one of the more important climate

indicator trends identified, signifying a warmer and drier climate as the influence of recent glaciation waned. This trend is consistent with early to mid-Holocene warming. A western larch (*Larix occidentalis* Nutt.) maximum suggests that fire slowed this trend, but as fires decreased, ponderosa pine continued to increase, again suggesting that warmer and drier conditions prevailed. A general decline in whitebark pine from the bottom of the profile to the level of ponderosa pine maximum is substantiating evidence for this climatic trend. Ponderosa pine pollen then declines from its peak to proportions that are generally maintained to the surface, marking the shift to a cooler and wetter climate (compared to the earlier warm and dry period) and present-day conditions. Today, ponderosa pine is not within the vicinity of the Anthony Lakes area (it is found at much lower elevations).

Mehringer[3] reports on a pollen and macrofossil record from Lost Lake, which is located in the Umatilla National Forest on the edge of the Vinegar Hill-Indian Rock Scenic Area about 50 km southeast of Dale, Oregon, and only 3 km north of the Malheur National Forest border. At 1870 m elevation, the surrounding slopes of the lake are within a mixed-conifer forests above the elevation of the ponderosa pine zone. Grand fir, Douglas-fir, western larch, and lodgepole pine are common. Engelmann spruce is restricted to cold area drainages and moist areas around the lake. A 3.8-m core located above the Mazama ash tephra layer was examined, thus the record spans the last 7,600 years. The record reveals that lodgepole pine, Douglas-fir, and western larch have been the dominant conifers at Lost Lake for the past 7,600 years. However, a transition at about 4,000 years BP is noted. Relatively more pine, spruce, and fir pollen, and less Douglas-fir and western larch pollen are noted at that point in the record. This change is attributed to an increase in effective moisture, coinciding with a transition from open woodland conditions to closed-canopy forests. Although the author does not specifically discuss mid-Holocene warming, this change in vegetation conditions probably relates to the end of mid-Holocene warming and the development of present-day cooler and wetter conditions. This change brought an episode of intense wildfires starting about 3,600 years BP and concluding around 1,860 years BP; sagebrush pollen is also a more important component of the pollen profile 9,000 to 5,600 years BP.

Whitlock and Bartlein (1997) present a pollen record from Carp Lake, located at 714 m elevation in the eastern Cascades. Presently, the site is dominated by ponderosa pine. The Carp Lake record indicates the development of a pine-oak woodland at 9,000 BP. Blinnikov et al. (2002) presents a phytolith record in loess

[3] **Mehringer, P.J., Jr. 1997.** Late Holocene fire and forest history from Lost Lake, Umatilla National Forest, Blue Mountains, Oregon. Unpublished report. On file with: U.S. Department of Agriculture, Forest Service, Malheur National Forest. John Day, OR.

from four different sites in the interior Columbia Basin, two of which were located in the Blue Mountains, using the Carp Lake record and the phylioth sites to infer vegetation in the Columbia Basin during the last 21,000 years (fig. 6.3).

Climate, Carbon Dioxide, and Other Studies

Dendroecological records (tree rings) from the past several hundred years also provide evidence about changes in tree growth with climatic variability. Experimental studies involving altered temperature, water availability, and carbon dioxide concentration provide information about how individual species may respond to climate change. Observational studies regarding response of species to current warming trends also provide clues to potential future responses of vegetation to climate. For example, van Mantgem et al. (2009) note that unmanaged old forests in the Western United States showed that background mortality rates have increased in recent decades. Increases occurred across elevations, tree sizes, dominant genera, and past fire histories, and the authors attributed elevated mortality to regional warming and consequent increases in water deficits.

Much of the evidence from climate, carbon dioxide, and other studies cited in this chapter is covered in greater detail in Peterson et al. (2014), which documents that climatic warming during the 20th century has led to a variety of plant responses, including altered phenology and geographic distributions of species (Crimmins et al. 2011, Kullman 2002, Parmesan 2006, Walther et al. 2005, 2012). Results from these types of studies are more appropriately discussed within the relevant vegetation type in the assessment section below.

Model Output

Many vegetation models have the ability to simulate the effects of climate change on vegetation, including altered productivity, distribution shifts, and disturbance regimes. These climate-aware vegetation models can be divided into three classes: species distribution (also called empirical, correlative, statistical, bioclimatic envelope, or niche), process based (also called mechanistic), and landscape (Littell et al. 2011).

Species distribution models (SDMs) use historical correlations of climate and historical species distributions to develop relationships and then project future suitable habitat under climate change scenarios (Hamann and Wang 2006, Iverson et al. 2008, Kerns et al. 2009, McKenzie et al. 2003a). These models are typically species specific. Model output is not the actual species range or habitat, although it is often presented as such; it is simply the climate and or environmental conditions that are correlated with the species historical presence or abundance. Many types of statistical relationships and approaches are used to make projections about the future (i.e., generalized linear models, machine learning, maximum entropy).

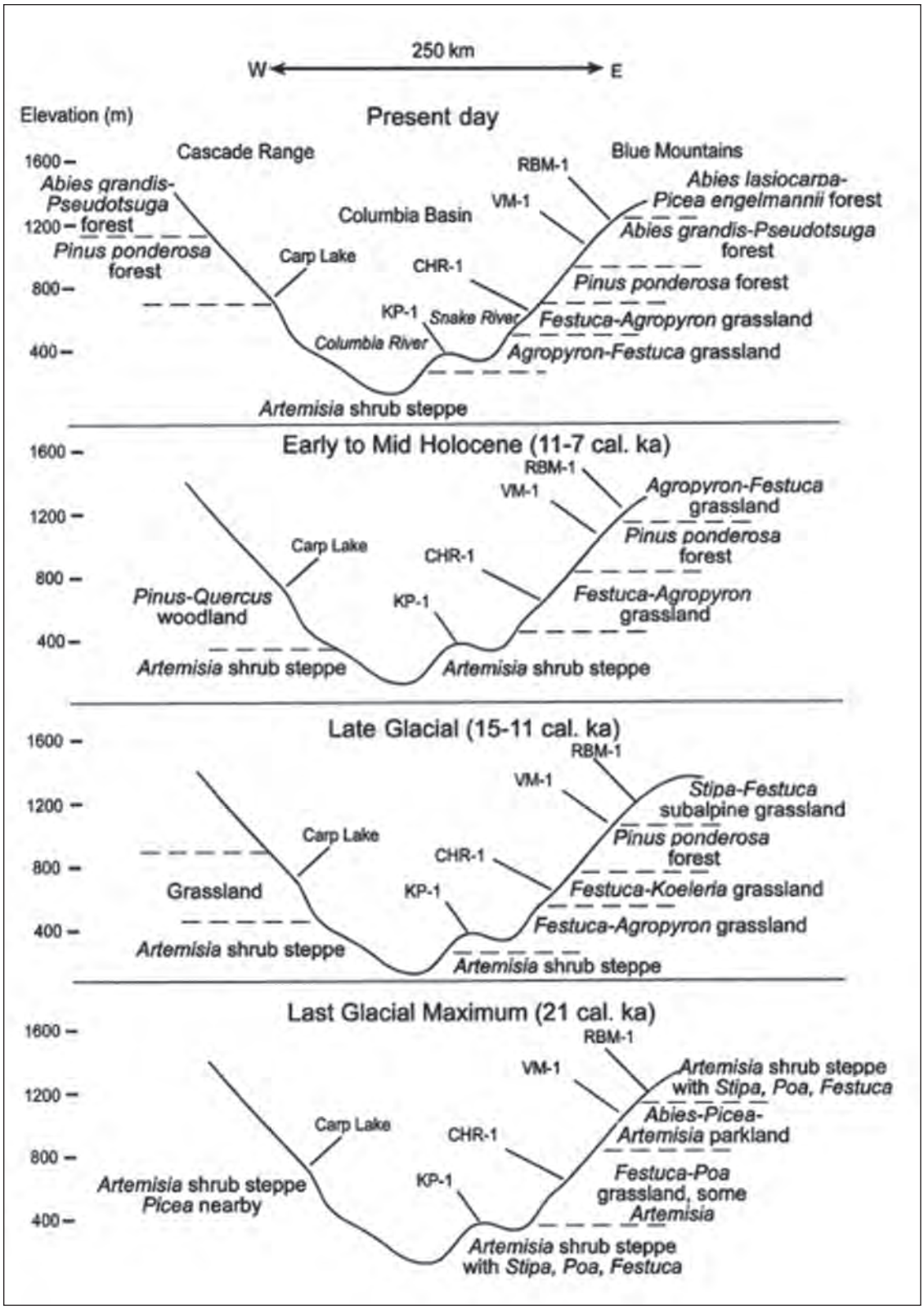

Figure 6.3—Inferred vegetation in the Columbia Basin during the last 21,000 years based on the pollen record at Carp Lake and phytoliths in four loess sections in the Columbia Basin. Figure from Blinnikov et al. 2002.

Process models can be species specific, such as forest gap models (Bugmann 2001), or they can simulate groups of species with similar form and function in ecosystems (plant functional types), including dynamic global vegetation models (DGVMs) (Prentice et al. 2007). Although some dismiss this class of model because it does not include species-specific information (Morin and Thuiller 2009), relevant information can be inferred from these models, including regionally specific and biome-scale information from projections based on plant functional types. Moreover, information at multiple taxonomic levels (e.g., species vs. vegetation type) about how climate might affect vegetation change, and the difference between projections and their sources, are themselves useful tools with which to assess uncertainty (Littell et al. 2011). Process models are parameterized from theoretical or experimental, and observational information; they project the response of an individual or a population to environmental conditions by explicitly incorporating biological processes (Bachelet et al. 2001, Bugmann 2001, Campbell et al. 2009, Cramer et al. 2001, Morin and Thuiller 2009). Some operate by using average climate while others use transient climate data. Vegetation or other output by the models is an "emergent property" in relation to key model input information (e.g., climate, soils, carbon dioxide emissions). Some argue that process models may be more robust at projecting the future given their basis in known mechanisms and their theoretical capability to extrapolate into nonanalog conditions (Coops and Waring 2011a, Thuiller 2007).

Landscape models expand either of the first two types to explicitly simulate landscape change and ecological processes such as succession and fire. The models are often spatial but may be distributional (Baker 1989, Cushman et al. 2007), such as nonspatial state-and-transition simulation models (Kerns et al. 2012). These models are well suited to simulations of the effects of disturbance on vegetation, which can be a major limitation for some of the empirical and process models. They can use climate information as inputs, but climate is not always explicitly incorporated into landscape models (Littell et al. 2011).

The inclusion of disturbance and extreme events in most models is still early in the developmental process.

Despite the specific strengths and weaknesses of different modeling approaches, there are caveats specific to all vegetation models. For example, most of the models do not deal with inertia (the tendency of vegetation to remain unchanged through various mechanisms) in vegetation, although some landscape models have the capacity to deal with inertia if they are parameterized to include climate change (Halofsky et al. 2013). Most of the models also do not incorporate disturbance processes, although MC2, a DGVM, does address fire (box 6.1). The inclusion of disturbance and extreme events in most models is still early in the developmental process (Keane et al. 2004, Lenihan et al. 1998, Thonicke et al. 2001).

Most models do not consider biotic interactions (although some process models do consider competition) and phenotypic plasticity (box 6.2), and no models deal with dispersal or genetic adaptation. A new generation of models incorporating climatic variability with many of these important processes is clearly needed. In addition, many models are used without being methodically calibrated and without careful validation of model skill or sensitivity analysis, such that model uncertainty is not understood. For all models, if future conditions differ greatly from conditions used to build and calibrate a model, then accuracy for projecting ecological phenomena will be relatively low (Williams et al. 2007).

Multiple projections using different vegetation models can be used to assess a range of potential changes in climate habitat and implications for changes in species distribution under future climate scenarios. We assess vegetation model projections for the end of the 21st century for the BME using different types of vegetation models. Our goal is to summarize and discuss agreement or disagreement in the models regarding general trends relating to potential changes in climate habitat for vegetation, and implications of the changes for future vegetation distribution. Although climate change projections simulated by more than 20 different GCMs have been available in recent years, most of the models we examined used climate data as inputs that were derived from only a small subset of the available GCM output. We tried to limit our assessment to common climate projections (GCM and greenhouse gas emission scenario combination) among the models, although this was not always possible. In addition, we assessed model outputs under only the "business as usual" emission scenarios (A2, A1FI, and RCP 8.5), which significantly depart from other scenarios after the mid 21st century (Nakicenovic et al. 2014, Nakicenovic and Swart 2000).

We selected species-specific model output from websites and peer-reviewed literature supporting the model output (table 6.1). We also used the MC2 model, a new version of MC1 rewritten for improved computational efficiency.[4] MC2 simulates response of plant functional types to climate change at a monthly type step, including plant physiology, biogeography, water relations, and interactions with fire. We selected four GCM outputs published by Coupled Model Intercomparison Project Phase 5 (CMIP5) for the RCP8.5 greenhouse gas emissions scenario using impact-relevant sensitivities as a guide (Vano et al.).[5] The four selected

[4] Kim, J.B.; Conklin, D. [N.d.]. MC2 Dynamic Global Vegetation Model: improvements in model structure and computational efficiency. Manuscript in prepartion. On file with: J. Kim, U.S. Department of Agriculture, Pacific Northwest Research Station, 3200 SW Jefferson Way, Corvallis, OR 97331.

[5] Vano, J.A.; Rupp, D.E.; Kim, J.B.; Mote, P.W. [N.d.]. Selecting climate change scenarios using impact-relevant sensitivities. Manuscript in prepartion. On file with: J. Kim, Pacific Northwest Research Station, 3200 SW Jefferson Way, Corvallis, OR 97331.

Box 6.2—Model limitations: phenotypic plasticity and biotic interactions

Models that predict vegetation responses to climate change, particularly species distribution models, are generally unable to account for the effects of several important factors such as phenotypic plasticity (Nicotra et al. 2010, Valladares et al. 2014) or the outcomes of **biotic interactions** (Araújo and Peterson 2012). However, it is well-established that biotic interactions can profoundly influence how plants respond to changing environmental conditions (Tylianakis et al. 2008), and that competition will play moderate species distribution patterns and responses to climate change (Brooker 2006). Moreover, long-lived organisms, such as trees can substantially modify their ability to tolerate stress or acquire resources as a consequence of plastic responses to external environmental conditions experienced in their lifetimes. This often results in significant differences in phenotypes among individuals, resulting in individualistic responses to environmental change (Nicotra et al. 2010, Richter et al. 2012). Therefore, assessing the effects of climate change on vegetation requires understanding (1) how climate affects species distributions across landscapes, (2) how these relationships are modified by biotic interactions and expressions of phenotypic plasticity (Araújo and Guisan 2006), and (3) how management actions influence resilience and adaptive capacity of forest ecosystems (Choi 2007).

Phenotypic plasticity refers to the range of functional traits that can be expressed by a particular genotype or individual plant in response to the environment. Traits that can vary in response to the environment include plant physiological processes (e.g., respiration rates, growth phenology), morphology (e.g., height, allocation to roots), and reproduction (e.g., method, timing) (Peterson et al. 2014). However, costly tradeoffs make plasticity less common than one might expect, given the apparent benefits to individuals and populations (Peterson et al. 2014). If plasticity cannot allow plants to adjust to changing environmental conditions, genetic adaptation or migration may be required to maintain species viability.

Phenotypic plasticity allows plants to adjust to seasonal changes in climate and longer term climatic variability, and helps avoid extreme vegetation responses to typical (historical) climatic variability. The ability to persist as a juvenile (advance regeneration) facilitates rapid response to increased resource availability following disturbance (or a favorable period of climate), whereas the ability to persist as an adult provides a continuing opportunity to reproduce during favorable periods of climate (Peterson et al. 2014). High phenotypic plasticity may be associated with ecological generalists and benefit plants as the climate changes, because high plasticity allows a single genotype to occupy different environments (Matesanz et al. 2010).

Phenotypic plasticity can help plants become established and persist under low resource availability caused by climatic variability, biotic interactions, and management actions. Unlike most short-lived species, trees can modify their stress tolerance or acquire resources in response to environmental conditions, resulting in individualistic responses to environmental change. For example, the effects of long-term adaptations to water stress (changes in carbon allocation and hydraulic architecture) may influence the ability to cope with environmental variability. Altering water relations by shifting biomass allocation from leaves to woody tissue is an adaptive

Box 6.2 (continued)

response of trees exposed to increased warming and drying (Cregg 1994, Parmesan 2006, Zwiazek and Blake 1989).

Competition and stand dynamics affect plant communities and vegetation dynamics (Connell and Slatyer 1977, Grime 1979, Tilman 1982). Assessing the relationship between competition and climate sensitivity for mature trees requires information on the proximate effects of neighbors on resource availability and understanding how expressions of phenotypic plasticity and long-term adaptations to competitive stress may influence the ability to cope with environmental variability (Barnard et al. 2011, Woods 2008). At a local scale, stand density and structure, which can be modified through silvicultural treatments, can affect environmental conditions and create sharp gradients in factors that regulate tree growth (light, water, temperature) (Aussenac 2000, Zhu et al. 2000). As a result, trees growing with high competition or in subdominant canopy positions are consistently exposed to different environmental conditions that affect morphological and physiological characteristics, including a lower ratio of leaf area to sapwood area (McDowell et al. 2006, Renninger et al. 2007), higher shoot-to-root ratio (Newton and Cole 1991), and reduced rooting depths (McMinn 1963). Trees experiencing high competition have reduced allocation to roots, which reduces water available to support transpiring leaves and necessitates large amounts of sapwood for water storage. However, these adaptations could also buffer the negative effects of stressful years and decrease drought sensitivity.

Differences in climate sensitivity between dominant and subdominant trees vary significantly among species depending on life history traits. In one study, dominant ponderosa pines were nearly twice as sensitive to high temperatures as ponderosa pines (*Pinus ponderosa* Douglas ex P. Lawson & C. Lawson) growing in subdominant canopy positions, but sensitivity of Douglas-fir did not vary across canopy classes (Carnwath et al. 2012). The contrasting effects of canopy position on temperature responses of these two species likely reflect fundamental differences in their physiology, morphology, and hydraulic conductivity.

Biotic interactions can significantly influence species distributions and community composition at multiple scales (Araújo and Luoto 2007). For example, rising carbon dioxide concentrations and changing climate could alter the outcomes of biotic interactions if competing individuals or species respond differently to changes in the environment. If plants respond differently, the results could lead to increased or reduced relative rates of growth and reproduction. Changing climate and carbon dioxide concentrations could also alter the intensity of biotic interactions by increasing or reducing overall resource availability. A better understanding of the effects of climate on biotic interactions will assist resource managers with implementation of treatments that can affect those interactions (silviculture, fuel reduction, etc.).

Facilitation, or positive interactions in community structure, is not as well understood as competition (Brooker et al. 2008, Callaway 1995, Callaway and Walker 1997). Plants often rely on beneficial biotic interactions for initial establishment (e.g., nurse plants or logs, mycorrhizal infection), pollination, seed dispersal, and protection from herbivores. They also respond to or defend against detrimental biotic interactions such as competition for limited resources, herbivory, and seed predation. Positive

Box 6.2 (continued)

interactions can occur in all types of conditions but appear to be most important in stressful environments, like alpine, semiarid, and arid ecosystems (Callaway 1995, Callaway et al. 2002).

Large trees and shrubs can facilitate establishment and persistence of understory plants (including tree seedlings) by modifying the microclimate (e.g., shading) and availability of soil nutrients and water (McPherson 1997, Scholes and Archer 1997). For example, it has been demonstrated that sagebrush (*Artemisia* spp.) can increase water availability near the soil surface by transporting water from deeper layers at night via "hydraulic lift" (Caldwell et al. 1998, Caldwell and Richards 1989, Richards and Caldwell 1987), but this may also benefit invasive species such as cheatgrass (*Bromus tectorum* L.) (Griffith 2010). Subalpine conifers can reduce snowpack duration and increase growing season length, thereby facilitating establishment of herbaceous plants and tree seedlings (Brooke et al. 1970), or can increase local soil moisture by altering windflow patterns and enhancing local snow deposition (Holtmeier and Broll 2005).

Table 6.1—Summary of species-specific model output and scenarios examined[a]

Model name	Scenario/GCM	End of century	Citation	Link
3-PG (hybrid)	A2 CGCM2	2080	Coops and Waring (2011b)	http://www.pnwspecieschange.info/index.html
ForeCASTS	A1(A1FI) HADCM3 no elevation	2100	Hargrove and Hoffman (2005)	http://www.geobabble.org/~hnw/global/treeranges3/climate_change/atlas.html
Plant hardiness	HADGEM and Composite-AR5	2170–2100	McKenney et al. (2011)	http://planthardiness.gc.ca/ph_futurehabitat.pl?lang=en
Plant species and climate profile projections	A2/CGCM3 and A2/Hadley	2090	Rehfeldt et al. (2006)	http://forest.moscowfsl.wsu.edu/climate/species/index.php

[a] CGCM and Hadley were the common global climate models (GCMs) among the output available

GCM outputs—CSIRO-MK360, HADGEM2-ES, MRI-CGCM3, and NORESM1-M—were selected based on their performance ranking and climate representation. We avoided poorly performing models based on Rupp et al. (2013), then selected models that represented the ensemble or multimodel mean for the hottest, coolest, and wettest projections of all 30 available projections, respectively (fig. 6.4). The selected GCM outputs were downscaled to 30 arc-second spatial resolution using the delta method (Fowler et al. 2007) for model input. MC2 was calibrated for the historical period (1895–2009) using a hierarchical approach, using PRISM climate data (Daly et al. 2008) as input.[6] We used MODIS net primary productivity data (Zhao and Running 2010), an internally generated potential vegetation type map, and fire-return interval data from LANDFIRE (Rollins 2009) as reference data sets for calibration. Future vegetation conditions were simulated for 2010–2100 using the four downscaled GCM data as input; output variables included vegetation type, carbon fluxes and stocks, and fire occurrence and effect.

Climate change is expected to result in substantial areas that have novel climates with no modern analog.

Table 6.2 lists each species examined (from the species-specific models) and an overall summary of the model output. Projections indicate some loss to complete loss of habitat for most species. Specific model results are discussed in the vegetation assessment section below. The results in table 6.2 are typical for SDMs, showing large reductions in available climate habitat for many species. The SDMs often produce projections with dramatic reductions in suitable habitat for a species simply because future novel climates do not correspond to modern conditions under which the species occurs. Climate change is expected to result in substantial areas that have novel climates with no modern analog ("novel climates," Williams and Jackson 2007, Williams et al. 2007). This may be especially true of the Western United States where almost half the land area may have novel climatic conditions by the end of the century (Rehfeldt et al. 2006). Furthermore, as models of modern realized niches increase in complexity (more climatic predictors and interactions), the less likely they are to project that species will exist under future climate scenarios.

In some cases, a novel climate could be quite favorable for a species (e.g., Kerns et al. 2009). As a result, novel climates often create a bias in SDM projections toward reduced area for most species under future climate, without identifying which species would replace them. Thus, the projected complete loss of habitat of species may simply illustrate the widespread nature of novel future climate conditions in the BME. Some species have relatively broad ecological amplitudes

[6] Kim, J.B.; Conklin, D.; Kerns, B.K. [N.d.]. A hierarchical approach to calibrating MC2 Dynamic Global Vegetation Model. Manuscript in prepartion. On file with: J. Kim, U.S. Department of Agriculture, Pacific Northwest Research Station, 3200 SW Jefferson Way, Corvallis, OR 97331.

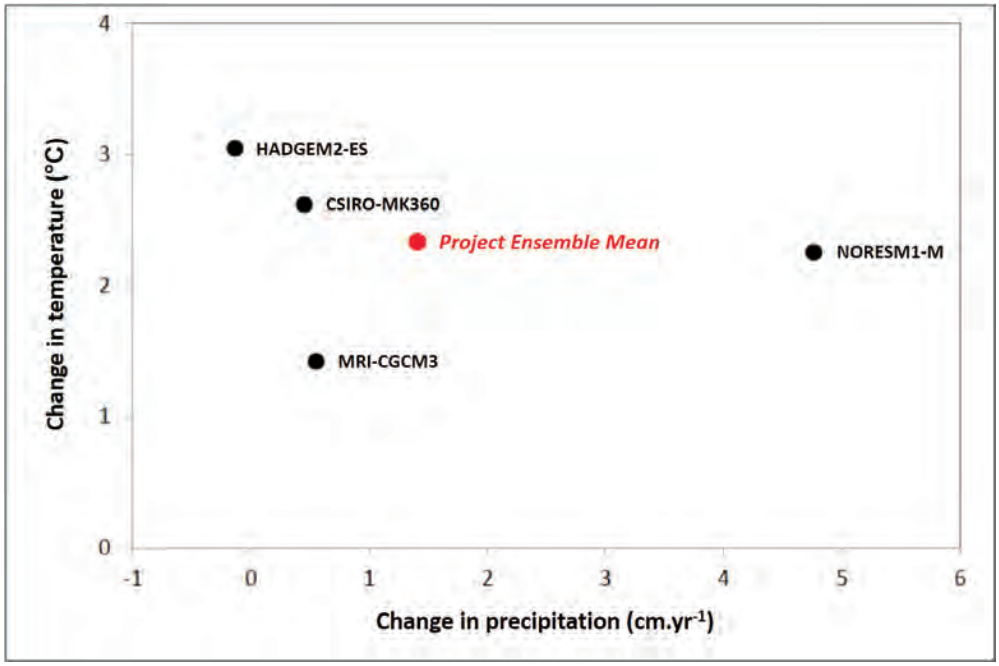

Figure 6.4—Projected annual change in precipitation and temperature compared to average annual historical Pacific Northwest climate for the global climate model (GCM) scenarios used as input to the MC2 model. All GCMs were run under Representative Concentration Pathway (RCP) 8.5, and the ensemble of all run under RCP 8.5 is shown for comparison.

Table 6.2—Summary of model projections and vulnerability assessment scores for some common upland species in the Blue Mountains

Potential vegetation or species	Common name	Species model output summary	Vulnerability assessment score[a]
Callitropsis nootkatensis	Alaska cedar	Complete loss[c]	Elevated[b]
Pinus flexilis	Limber pine	Major to complete loss[c]	Elevated[b]
Tsuga mertensiana	Mountain hemlock	Major to complete loss	Elevated[b]
Pinus albicaulis	Whitebark pine	Major to complete loss	74
Abies lasiocarpa	Subalpine fir	Moderate to complete loss, potential refugia in Wallowa Mountains	70
Picea engelmannii	Engelmann spruce	Moderate to complete loss, potential refugia in Wallowa Mountains[c]	61
Pinus monticola	Western white pine	Some loss to major; other models show shifts and minor loss	57
Populus tremuloides	Quaking aspen	Major to complete loss; one model showed minor loss	57
Abies grandis	Grand fir	Some loss to almost complete loss	51
Abies concolor	White fir	Major to complete loss[c]	51
Pinus contorta var. *latifolia*	Lodgepole pine	Moderate to complete loss	43
Pseudotsuga menziesii	Douglas-fir	Minor to major loss	42
Juniperus occidentalis	Western juniper	Major to complete loss[c]	30
Larix occidentalis	Western larch	Minor loss, shifts to complete loss	32
Pinus ponderosa	Ponderosa pine	Minor to major loss; one model shows minor shift	22
Artemisia tridentata	Big sagebrush	Complete loss[d]	NA
Cercocarpus ledifolius	Curl-leaf mountain-mahogany	Major loss[f]	NA
Purshia tridentata	Antelope bitterbrush	Complete loss[e]	NA

NA = not applicable.

Note: Loss or gain refers to climate habitat, not species range. Species are ordered from the highest vulnerability score to the lowest. Model output is summarized from all four models shown in table 6.1 unless otherwise noted.

[a] Based on Table 15 in Devine et al. (2012).

[b] These species were not offically ranked in Devine et al. (2012). However, they are considered to have an elevated vulnerability to climate change because they are rare in the Blue Mountains.

[c] Based on three models.

[d] Based on one model.

[e] Based on two models.

(e.g., lodgepole pine, juniper; Miller et al. 2005, Miller and Wigand 1994, Pfister and Daubenmire 1975) and may be very competitive in a novel environment. In contrast, outputs from MC2 indicate that although some forest types may decrease in the future, increases in shrublands and grasslands might be expected (fig. 6.5). Agreement among the four simulation outputs is high, especially for the arid portions of the ecoregion (fig. 6.6). The northern portion of the Umatilla National Forest has considerable uncertainty in MC2 projections.

Although some forest types may decrease in the future, increases in shrublands and grasslands might be expected.

Effects of Climate Change on Vegetation

To structure the vegetation assessment and development of adaptation options, we used the potential natural vegetation (PNV) concept and an existing vegetation classification scheme. Potential natural vegetation is defined as the plant community that would establish under existing environmental conditions in the absence of disturbance and without interference by humans (Chiarucci et al. 2010, Cook 1996). The PNV implies that over the course of time, similar types of plant communities will develop on similar sites, thus a potential plant association is an indicator of a particular biophysical condition or setting of a site.

The PNV concept has been debated in the literature (e.g., Chiarucci et al. 2010, Jackson 2013), and climate change effects on vegetation will probably not be consistent with the PNV concept. Paleoecological evidence suggests that species responses to climate are individualistic and no-analog plant communities should be expected in response to future climate change. Nevertheless, similar biophysical settings that support broad potential vegetation types will still exist in the future and occupy similar habitat types, but it is likely these habitat types will be redistributed across the landscape. For example, it is likely that some form of upland shrublands, grasslands, and dry forest types will exist in the future, although the species that make up these groups could be quite different.

The PNV concept is still useful for discussing changes in future vegetation, and U.S. Forest Service management is largely based on publications, classification systems, and models that rely on PNV. We used potential vegetation groups (PVGs; Powell et al. 2007), a mid-scale hierarchy system typically used for planning and strategic assessments at large spatial scales. These PVGs are similar to the potential plant functional types used for global biome vegetation modeling (e.g., Prentice et al. 1992), and they crosswalk reasonably well to MC2 output and potential vegetation functional types (table 6.3). Although these broad vegetation groups may exist in the future as representations of biophysical settings, the species that comprise them at fine spatial scales may differ greatly.

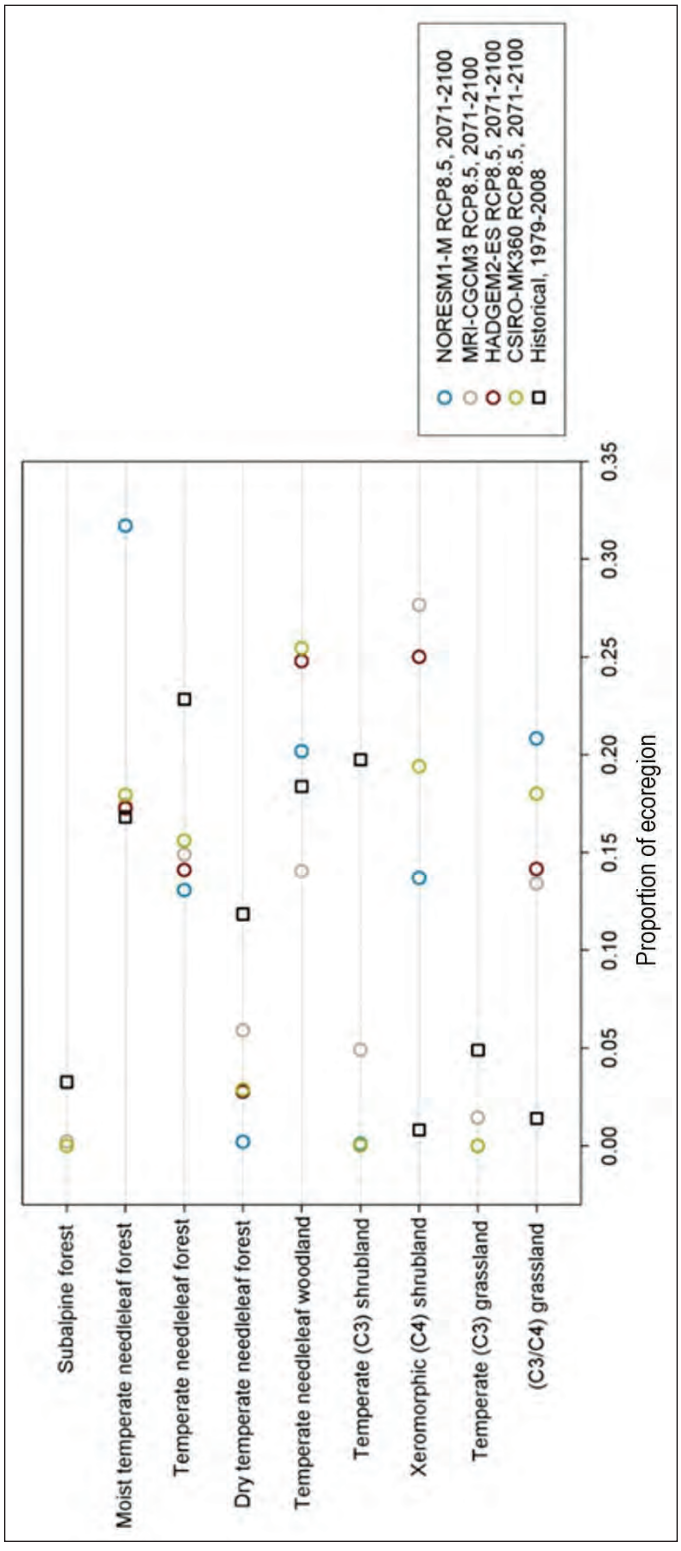

Figure 6.5—Projected changes in potential vegetation functional types for the end of the 21st century. Data are based on output from the MC2 model and four future climate scenarios (see inset box). Historical data were generated from simulated MC2 data and may be different than other potential vegetation maps.

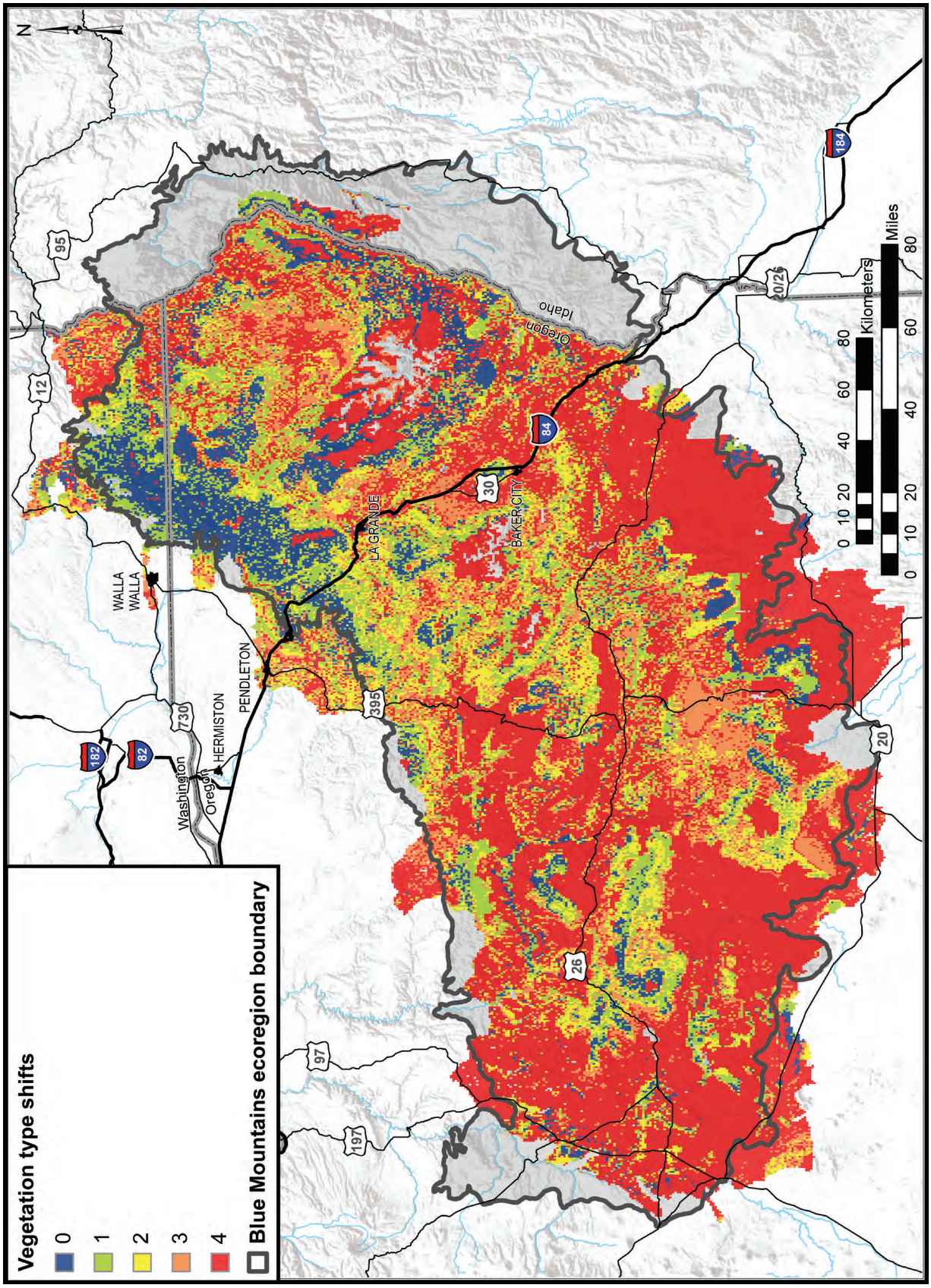

Figure 6.6—Number of projections resulting in vegetation type shift by the MC2 model from the historical period (1979–2008) to the late 21st century (2071–2100). The modeled ecoregion boundary is slightly different than the U.S. Environmental Protection Agency Level II Ecoregion (shown by the heavy black line).

Table 6.3—Summary of the upland potential vegetation groups (PVGs) used in this chapter, and the crosswalk between these PVGs and MC2 potential vegetation functional types

PVG	Typical species	MC2 potential vegetation functional type
Cold Upland Forest	Subalpine fir, Engelmann spruce, lodgepole pine, whitebark pine	Subalpine forest
Cold Upland Shrub	Mountain big sagebrush, Sitka alder, shrubby cinquefoil	NA
Cold Upland Herb	Greenleaf fescue, Idaho fescue, forbs, sedges	NA
Dry Upland Forest	Ponderosa pine, grand fir, or Douglas-fir	Temperate and dry temperate needleleaf forest
Dry Upland Herb	Bluebunch wheatgrass and Sandberg bluegrass	Temperate (C3) grassland
Dry Upland Woodland	Western juniper, mountain big sagebrush, curl-leaf mountain-mahogany	Temperate needleleaf woodland
Dry Upland Shrub	Low, scabland, and threetip sagebrush	Temperate (C3) shrubland
Moist Upland Forest	Subalpine fir, grand fir, Douglas-fir, lodgepole pine, western larch	Moist temperate needleleaf forest
Moist Upland Woodland	Western juniper, bluebunch wheatgrass, low and scabland sagebrush	Temperate needleleaf woodland
Moist Upland Shrub	Mountain big sagebrush, antelope bitterbrush, snowberries, bitter cherry	Temperate (C3) shrubland
Moist Upland Herb	Idaho fescue, bluebunch wheatgrass	Temperate (C3) grassland

NA = not applicable.

National forests in the Blue Mountains differ in their distribution of upland PVGs (table 6.4; figs. 6.7 through 6.9). Malheur National Forest uplands are dominated by the Dry Upland Forest (UF) PVG, and includes some Cold UF and moist forests. Umatilla National Forest uplands are dominated by Moist, Dry and Cold UF PVGs, a large amount of moist forest, and a considerable amount of Dry upland herbland. Wallowa-Whitman National Forest uplands are dominated by Dry, Cold, and Moist UF PVGs, with more Cold UFs than the other two Blue Mountains national forests.

To aid our assessment, we also used Potential Soil Drought Stress maps developed by the U.S. Forest Service Pacific Northwest Region (Ringo et al.[7]) (box 6.3). The following information, organized by PVG, was evaluated by workshop attendees to create the final assessment worksheets.

Cold Upland Forest

Cold UFs occur at moderate or high elevations in the subalpine zone, characterized by cold, wet winters, and mild, relatively cool and dry summers. Deep, persistent winter snowpacks are common, although the depth and persistence of winter snowpacks have been declining since the 1950s in response to climate change (Furniss et al. 2010, Karl et al. 2009, Stewart et al. 2004). Cold UFs have relatively short growing seasons, low air and soil temperatures, and slow nutrient cycling rates.

Late-seral stands are typically dominated by subalpine fir, grand fir, Engelmann spruce, whitebark pine, and lodgepole pine, but whitebark pine, lodgepole pine, and western larch are often persistent, early-seral species. Whitebark pine will successfully dominate late-seral stands at timberline, on low-moisture soils and southern exposures. Cold UFs are adjoined by a treeless alpine zone at their upper edge (often separated by a narrow zone of dwarf or krummholz trees), and by Moist UFs at their lower edge.

The Cold UF PVG consists of three plant association groups (PAGs), two in the cold temperature regime (Cold Moist and Cold Dry) and one in the cool temperature regime (Cool Dry). The Cold Dry UF PAG is the most widespread member of the Cold UF PVG and includes the more xeric of the high-elevation forested communities. Cold Dry UF plant associations occur on all aspects and many different substrates, often on sites with moderate to high impact from wind scour and, at lower elevation, in cold air pockets and drainages.

[7] Ringo, C.; Bennett, K.; Noller, J. [N.d.]. Climate change vulnerability assessment: resources for national forests and grasslands in the Pacific Northwest. Soil drought index. Unpublished report. On file with: Oregon State University, Department of Crop and Soil Science, 3017 Agricultural and Life Sciences Building, Corvallis, OR 97331.

Table 6.4—The distribution of upland potential vegetation groups (PVGs) among the three Blue Mountains Forests[a]

Potential vegetation group	Malheur National Forest	Umatilla National Forest	Wallowa-Whitman National Forest
		Percent	
Cold Upland Forest	11	12	18
Cold Upland Shrub	<1	1	1
Cold Upland Herb	<1	1	4
Dry Upland Forest	70	29	32
Dry Upland Woodland	<1	<1	<1
Dry Upland Shrub	6	1	2
Dry Upland Herb	3	11	13
Moist Upland Forest	5	38	10
Moist Upland Woodland	3	2	<1
Moist Upland Shrub	<1	2	1
Moist Upland Herb	<1	3	2

[a] Data are based on maps and boundaries shown in figures 6.12 through 6.14. Percentages do not add up to 100 because other PVGs (e.g., riparian areas) are not included.

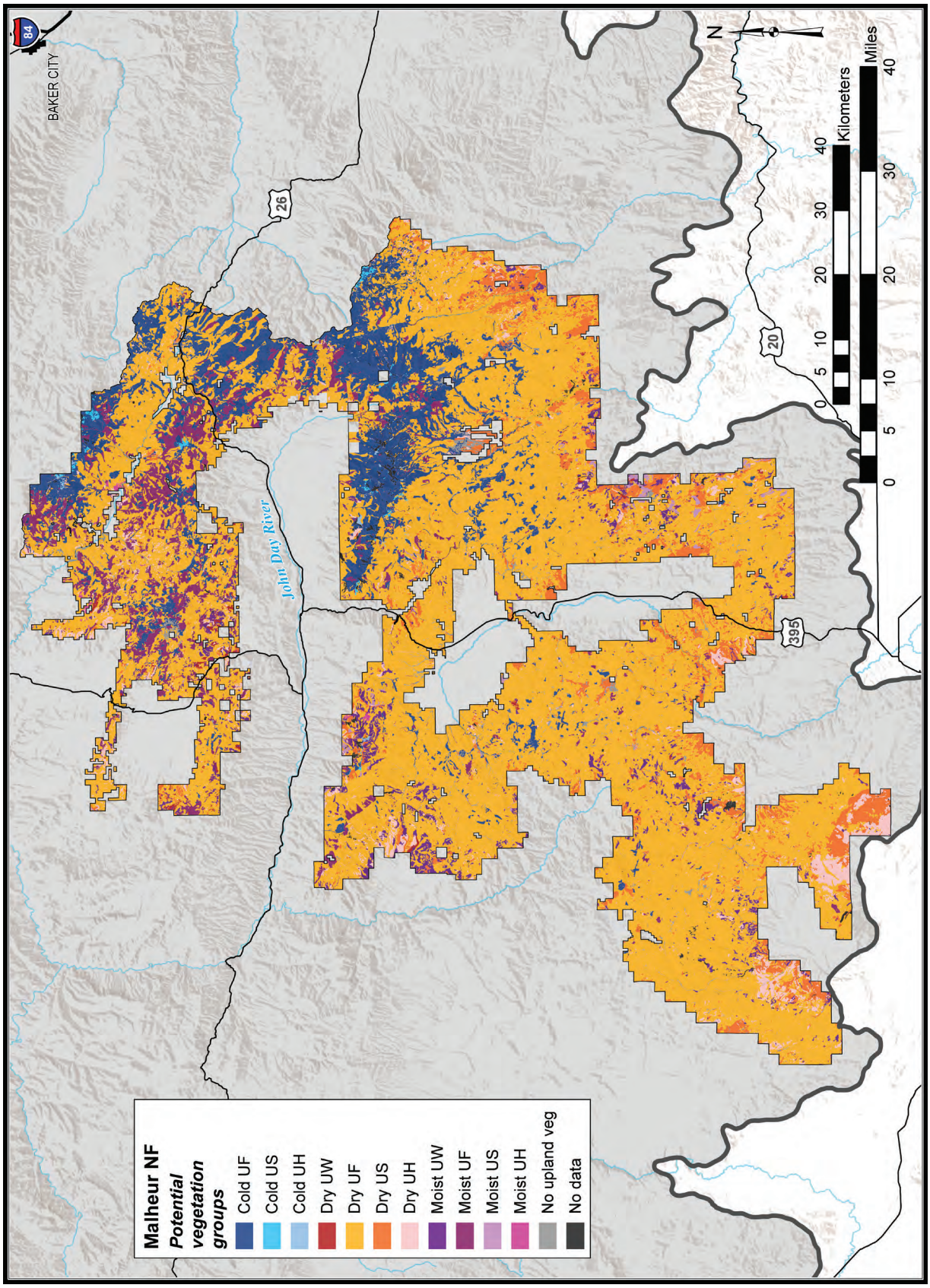

Figure 6.7—Potential vegetation groups (PVGs) for the Malheur National Forest (NF). Forest boundaries are proclaimed, and PVG data from small inholdings were removed. UF = Upland Forest, US = Upland Shrub, UH = Upland Herb, UW = Upland Woodland.

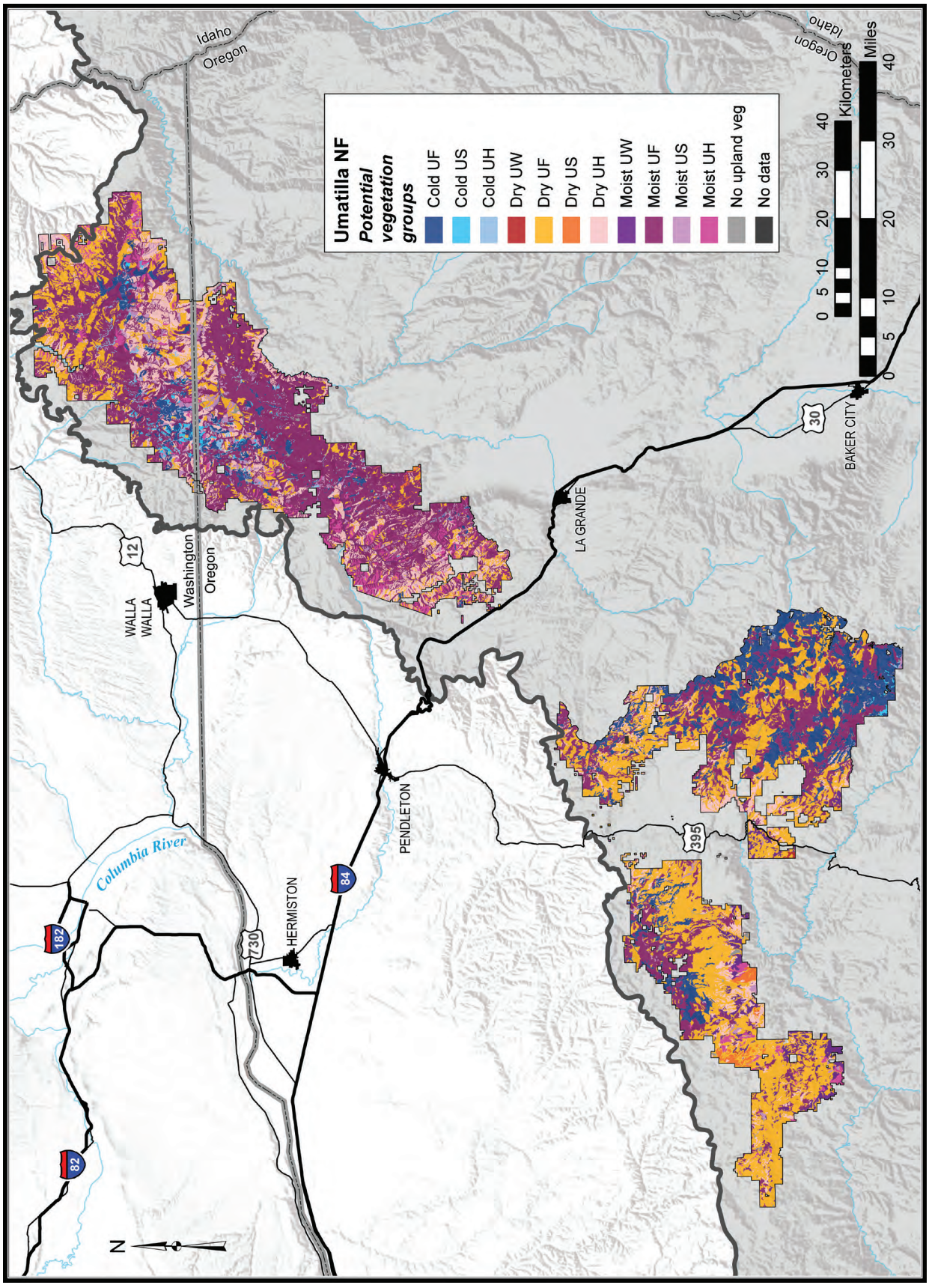

Figure 6.8—Potential vegetation groups (PVGs) for the Umatilla National Forest (NF). Forest boundaries are proclaimed, and PVG data from small inholdings were removed. UF = Upland Forest, US = Upland Shrub, UH = Upland Herb, UW = Upland Woodland.

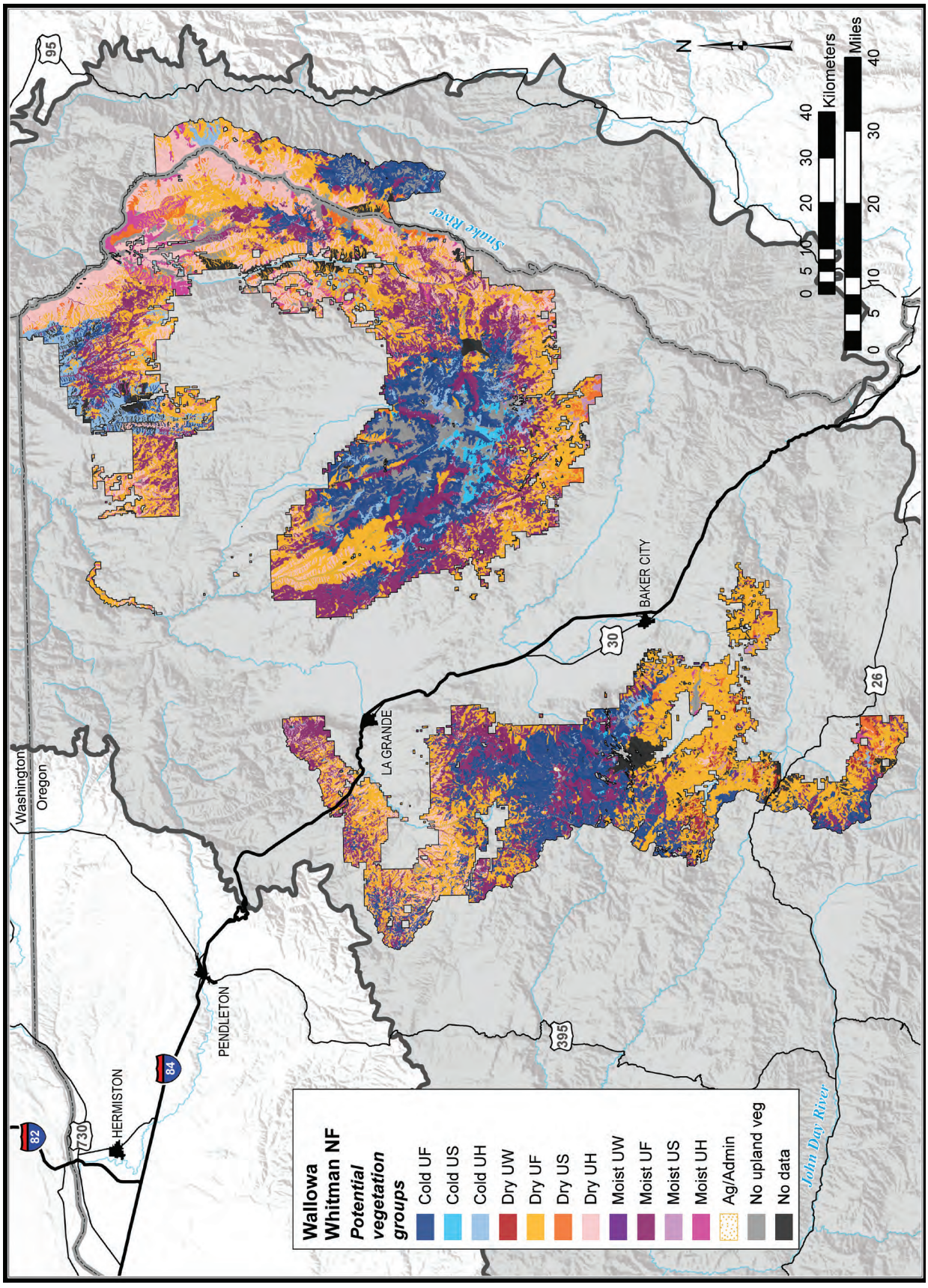

Figure 6.9—Potential vegetation groups (PVGs) for the Wallowa-Whitman National Forest (NF). Forest boundaries are proclaimed, and PVG data from small inholdings were removed. UF = Upland Forest, US = Upland Shrub, UH = Upland Herb, UW = Upland Woodland.

Box 6.3—Soil drought in forests of the Blue Mountains

One of the most important mechanisms by which climate change can influence vegetation composition and dynamics is through its effects on soil water availability, water uptake by plants, and evapotranspiration (Peterson et al. 2014). Soil water availability depends on precipitation inputs (form, amount, intensity, and seasonality), surface and subsurface water movement, evaporative demand, and vegetation structure and composition. Evapotranspiration is driven primarily by temperature and humidity, but can be constrained by reductions in stomatal conductance or reduced water uptake owing to low soil water availability. Peterson et al. (2014) reviewed the likely effects of projected changes in annual and seasonal mean temperature and precipitation on important components of the hydrologic cycle, including the fraction of winter precipitation received as snowfall, snowpack water storage, snowmelt rates, evapotranspiration, and soil water availability.

By examining soil indices related to drought throughout the Blue Mountains, land managers can potentially identify areas with drought-prone soils, although the relationship between low soil moisture in soils and vegetation vulnerability may not be straightforward. For example, soil parent materials that differ widely in texture may have little effect on mortality of trees and shrubs (Koepke et al. 2010). In addition, some experimental studies have shown that conifers preconditioned by exposure to water stress can actually have higher survival rates and improved water relations during subsequent drought events (Cregg 1994, Zwiazek and Blake 1989). Therefore, the existence of drought-prone soils may not necessarily imply vulnerability.

We examined the Potential Soil Drought Stress (PSDS) index (under development by U.S. Forest Service Pacific Northwest Region) to assess soil drought stress in the Blue Mountains in spring (April, May, June) and summer (July, August, September). The PSDS index ranged from low (1) to high (5) and was developed using available water storage, soil, and modeled actual and potential evapotranspiration data. Methods are summarized below, based on Ringo et al. (see footnote 7).

Available water holding capacity (AWHC) is the amount of plant-available water that can be held in each inch of soil if that soil is at field capacity (i.e., after free drainage of water following a storm has ceased). AWHC was calculated from Natural Resources Conservation Service soil surveys in the Soil Survey Geographic Database (SSURGO) at a scale of 1:24,000 and U.S. Forest Service Soil Resource Inventories (SRIs) at a scale of 1:63,560 where SSURGO was not available. Available water storage (AWS) for the entire soil profile was calculated by multiplying the AWHC for each inch of soil by the depth of the soil to 59 in or to a root restricting layer or bedrock, whichever was shallower. All calculations were for the dominant soil in the soil map unit. Data were classified into five AWHC categories from low to high.

Potential evapotranspiration (PET) is an estimate of the evaporation and transpiration that would occur if an adequate supply of moisture were available. **Actual evapotranspiration** (AET) measures the actual loss of moisture from soil and plant surfaces, and so the degree to which AET falls below PET may be interpreted as an indicator of moisture limitation. Modeled AET and PET data sets were derived from the Numerical Terradynamic Simulation Group at the University of Montana (http://www.ntsg.umt.edu/project/mod16). In their MODIS Global Evapotranspiration Project, they used remotely sensed land cover, leaf area index, and albedo data together with daily meteorological data for air temperature, air pressure, humidity, and shortwave radiation to model AET and PET monthly averages for the years

Box 6.3 (continued)

2000–2012. Departure from PET was calculated and classified into one of five categories that ranged from low (AET/PET = 0.80 to 1) to high (AET/PET = 0.20 or less) (table 6.5). Inaccuracies of this approach include (1) atmospheric interference from aerosols and cloud cover, and reflectance from snow creates inaccuracies in the reflectance data and derivative products used to create the PET data set; and (2) there are some years in which sensors were malfunctioning and data were estimated rather than measured.

A map of PSDS index was created by overlaying AET/PET and AWHC data (table 6.4). If there is little departure from PET (AET/PET is low) then vegetation is transpiring at full potential and there is enough moisture to allow this level of transpiration, then climatic factors, not soil water storage, limit available moisture for plant use. However, the degree to which AWHC or climatic factors are governing plant transpiration is not actually known.

As the departure from PET increases (e.g., AET/PET is much less than 1), then plant transpiration is below full potential. If AET/PET is extremely low (< 0.20 or close to 0), then it follows that all forms of available moisture for plants have largely been exhausted. However, the AET/PET spatial data describe average conditions across each grid cell and the AWHC data are finer scale. It is possible that even if AET/PET is classified as low within a grid as an average, areas that have soils with high AWHC within that grid cell may not be as dry as soils that have a lower AWHC. This means that although AET/PET may be mapped as extremely low, soils with higher AWHC may still have the ability to sustain growth. The PSDS index allows the combination of two scales of data (coarser scale AET/PET data and finer scale AWHC data) to provide a potentially more refined understanding of plant drought stress.

The PSDS maps are shown for each national forest in the Blue Mountains in figures 6.10 through 6.12. The PSDS index has not yet been peer reviewed and relies heavily on the accuracy of the SSURGO data at a very fine scale. Additional field

Table 6.5—Potential soil drought stress index (PSDI) categories derived from an overlay of departure from potential transpiration (AET/PET) and available water-holding-capacity data

Available water holding capacity (0-150 cm)	Departure from potential evapotranspiration				
	Low (0.8-1.0)	Medium low (0.60-0.80)	Medium (0.4-0.6)	Medium high (0.2-0.4)	High (≤ 0.20)
High	1	1	2	2	3
Medium high	1	2	3	3	3
Medium	1	2	3	3	4
Medium low	1	2	3	4	5
Low	2	3	4	5	5

Note: Additional details are provided in box 6.3. PSDI categories are an estimate of potential stress and range from 1 to 5, where 1 = low, 2 = medium low, 3 = medium, 4 = medium high, and 5 = high. AET = actual evapotransporation.

Box 6.3 (continued)

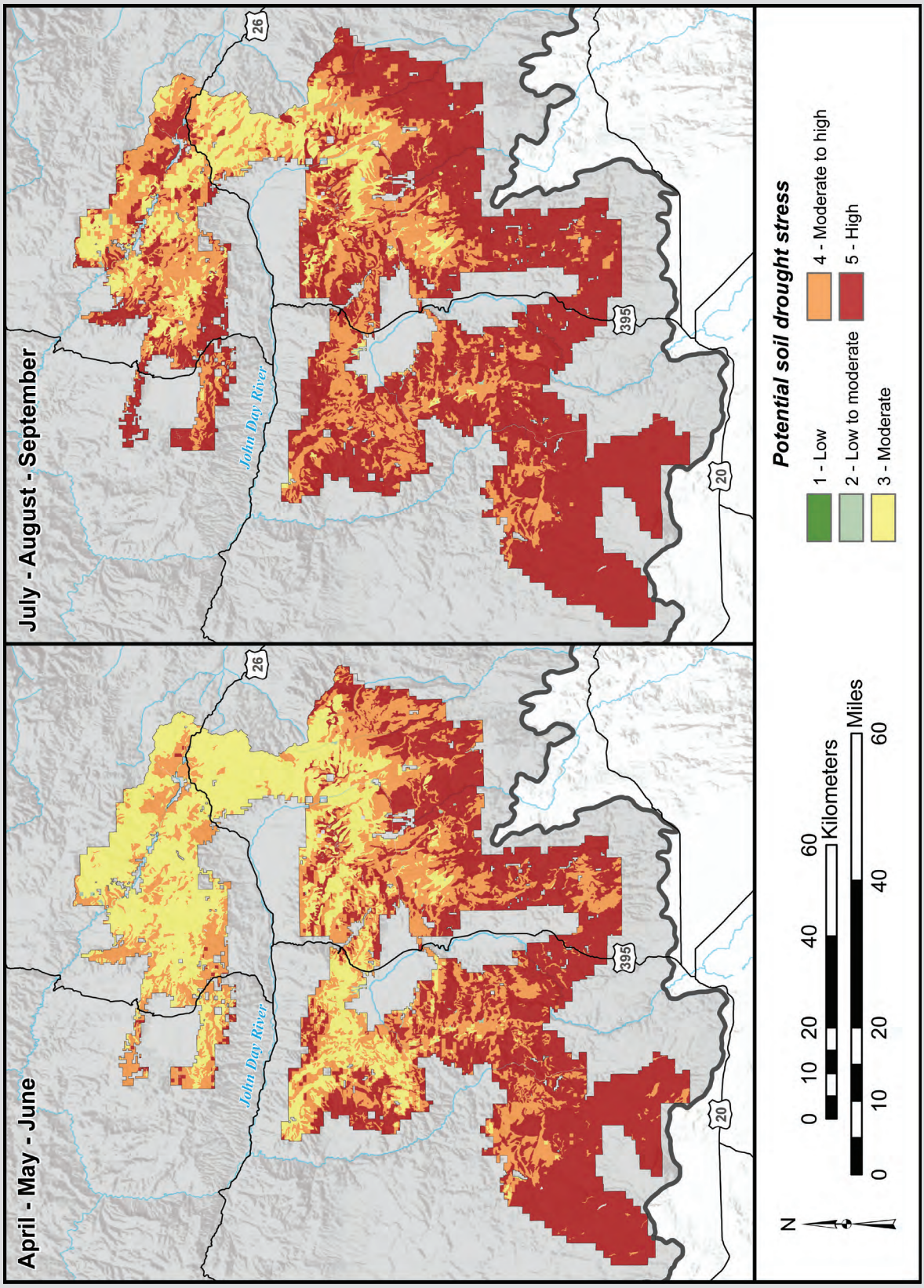

Figure 6.10—Potential soil drought stress in the spring and summer for the Malheur National Forest.

Box 6.3 (continued)

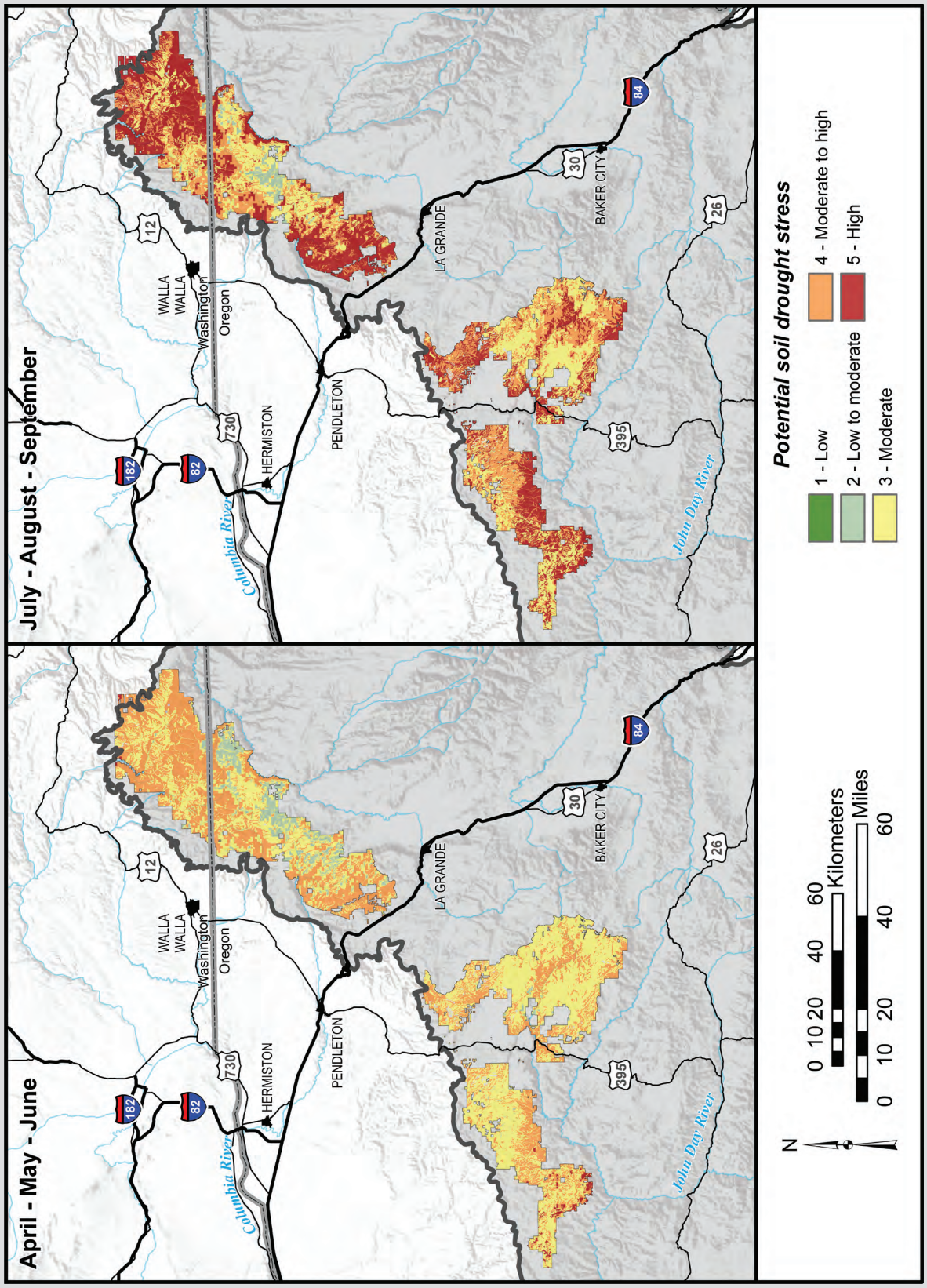

Figure 6.11—Potential soil drought stress in the spring and summer for the Umatilla National Forest.

Box 6.3 (continued)

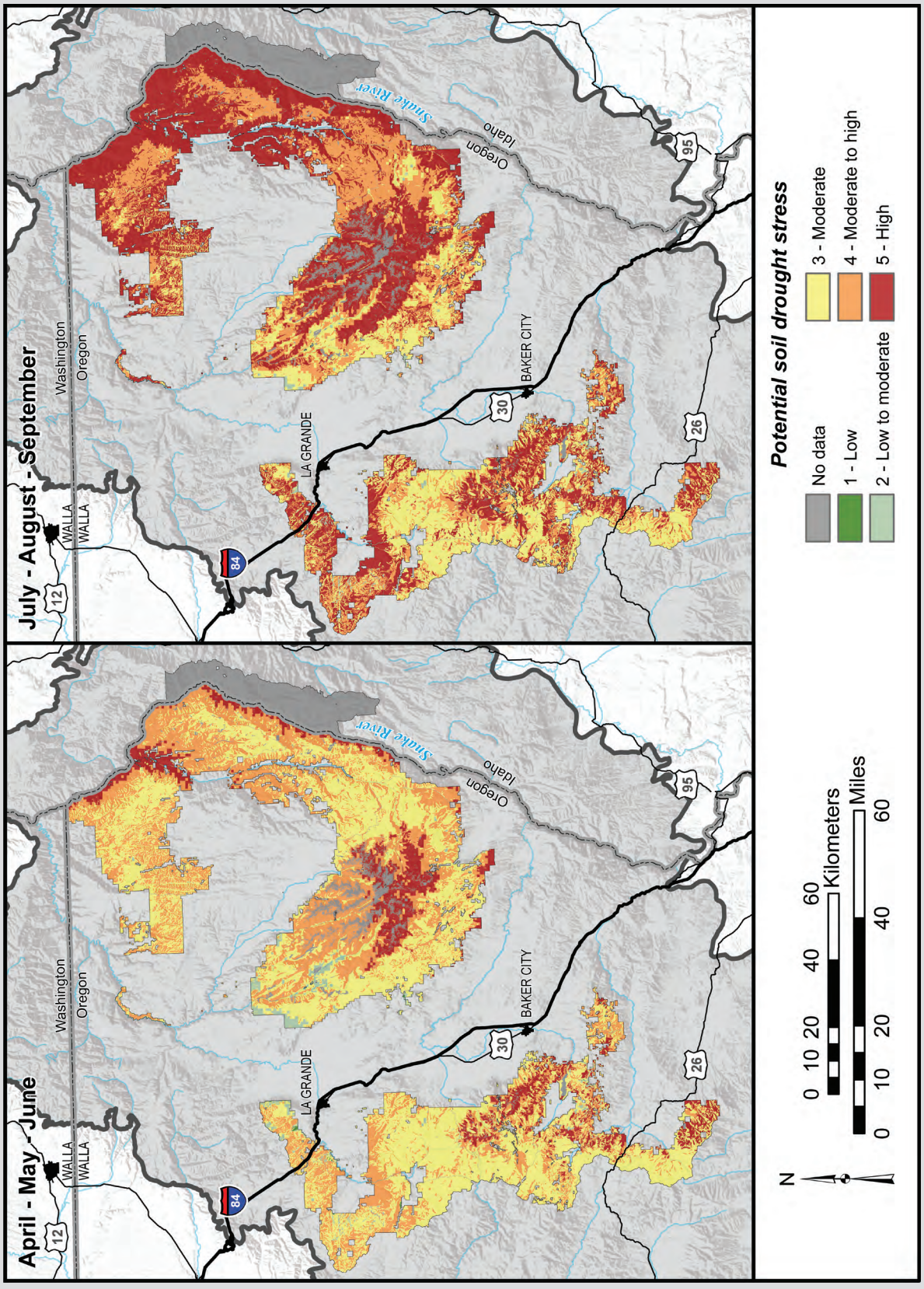

Figure 6.12—Potential soil drought stress in the spring and summer for the Wallowa-Whitman National Forest.

Box 6.3 (continued)

testing will help validate many assumptions and test data accuracy. Reliance on the fine-scale data could lead to an inappropriate spatial focus and assumptions about accuracy ("my favorite pixel"). Although the SSURGO soil data are available at relatively fine spatial scales, soil can be heterogeneous at even finer scales (as small as a few feet). Examination of the two spatial layers separately may be more appropriate. The PSDS index may simply provide an indication of where on the landscape vegetation may be vulnerable to drought stress, particularly across similar vegetation types or species.

Malheur National Forest

- Overview—Patterns in the Malheur National Forest are distinctive from other areas in the Blue Mountains, owing to the southerly location, lack of maritime influence, and lower elevation terrain (fig. 6.10). Soils are typified by fewer volcanic soils and more clay-dominated soils. A substantial portion of the landscape (41 percent) is classified as high PSDS (table 6.6). Most of the nonforested PVGs cover a small spatial extent, so we might expect more error in classification associated with these groups.
- Spring—Seventy-one percent of the forest is classified into the two highest PSDS categories combined. Only 11 percent of the forest remains in the moderate PSDS category. All PVGs have substantial portions in the moderate and moderate to high PSDS categories (table 6.7). Nonforested PVGs have more area in the moderate-to-high categories compared to forested PVGs. The Dry UW, US, UH and Moist UW and US groups all have 90 percent or more area in moderate-to-high or high PSDS categories combined. However, the Dry UF group actually has a higher proportion in the two highest PSDS categories (78 percent) than some of the other nonforested PVGs. The Dry US group has the highest spatial extent in moderate-to-high and high PSDS (98 percent).

Table 6.6—Potential soil drought stress index for the three national forests in the Blue Mountains

Potential soil drought stress index	Umatilla National Forest		Wallowa-Whitman National Forest		Malheur National Forest	
	Spring	Summer	Spring	Summer	Spring	Summer
	Percent					
1–Low	0	0	0	0	0	0
2–Low to moderate	5	2	2	1	0	0
3–Moderate	52	29	46	22	28	11
4–Moderate to high	42	30	34	26	30	28
5–High	1	39	9	43	41	60

Note that not all percentages within a forest add up to 100 because of rounding or because some areas contain no data. Categories are based on mapped data in figures 6.12 through 6.14

Box 6.3 (continued)

Table 6.7—Potential vegetation groups (PVGs) and their relative proportion of area mapped in each potential soil drought stress category for each national forest in spring (April, May, June) and summer (July, August, September)

	Potential soil drought stress: spring					Potential soil drought stress: summer				
Potential vegetation group	**1**	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
	Relative percent									
Malheur National Forest:										
Cold Upland Forest	0	0	70	21	10	0	0	40	39	21
Cold Upland Shrub	0	0	31	27	42	0	0	5	23	72
Cold Upland Herb	0	0	28	30	42	0	0	6	16	77
Dry Upland Forest	0	0	23	34	44	0	0	6	30	64
Dry Upland Woodland	0	0	9	51	40	0	0	1	8	92
Dry Upland Shrub	0	0	1	19	79	0	0	1	3	97
Dry Upland Herb	0	0	10	28	62	0	0	2	9	89
Moist Upland Forest	0	0	77	19	4	0	0	43	38	19
Moist Upland Woodland	0	0	4	31	65	0	0	0	5	95
Moist Upland Shrub	0	0	5	19	75	0	0	1	8	91
Moist Upland Herb	0	1	15	69	14	0	0	7	9	83
Umatilla National Forest:										
Cold Upland Forest	0	1	78	21	0	0	1	46	36	18
Cold Upland Shrub	0	1	29	70	0	0	3	23	40	34
Cold Upland Herb	0	2	21	77	0	0	1	16	31	52
Dry Upland Forest	0	2	55	42	1	0	1	24	33	42
Dry Upland Woodland	0	0	4	96	0	0	0	3	1	96
Dry Upland Shrub	0	1	14	83	2	0	0	3	9	88
Dry Upland Herb	0	1	13	85	1	0	0	3	15	82
Moist Upland Forest	0	12	60	28	0	0	4	41	32	23
Moist Upland Woodland	0	0	14	73	14	0	0	5	6	89
Moist Upland Shrub	0	1	24	74	1	0	0	9	32	59
Moist Upland Herb	0	1	15	82	2	0	0	2	12	86
Wallowa-Whitman National Forest:										
Cold Upland Forest	0	0	57	30	12	0	0	32	28	39
Cold Upland Shrub	0	0	7	31	61	0	0	2	13	85
Cold Upland Herb	0	0	32	51	17	0	0	2	16	81
Dry Upland Forest	0	2	58	34	5	0	1	27	33	38

Box 6.3 (continued)

Table 6.7—Potential vegetation groups (PVGs) and their relative proportion of area mapped in each potential soil drought stress category for each national forest in spring (April, May, June) and summer (July, August, September) (continued)

	Potential soil drought stress: spring					Potential soil drought stress: summer				
Potential vegetation group	1	2	3	4	5	1	2	3	4	5
	Relative percent									
Dry Upland Woodland	0	0	20	35	45	0	0	6	18	76
Dry Upland Shrub	0	0	14	54	32	0	0	2	10	88
Dry Upland Herb	0	0	17	68	15	0	0	2	12	85
Moist Upland Forest	0	4	73	21	2	0	2	39	41	19
Moist Upland Woodland	0	0	8	40	52	0	0	3	10	86
Moist Upland Shrub	0	1	21	51	27	0	0	8	21	72
Moist Upland Herb	0	0	22	63	16	0	0	2	14	83

Note: Values less than 0.5 percent are listed as 0. Categories are based on mapped data shown in figures 6.12 through 6.14. Note that for Wallowa-Whitman National Forest, not all mapped PVGs had associated PSDS data. "Unmapped" PSDS data were removed to calculate the relative percentages.

- Summer—Sixty percent of the forest is mapped as high potential drought stress and 28 percent as moderate to high. Therefore, almost 90 percent of the forest could be under extreme drought conditions in the summer. All PVGs have more area classified in moderate-to-high and high PSDS categories. Nonforested PVGs have high proportions of mapped drought-prone soils compared to forested PVGs. Ninety-seven percent of the Dry US PVG and 95 percent of Moist UW are in high categories. Values are lower for the forested PVGs, but all forested PVGs have substantial portions in the high category. Ninety-four percent of the Dry UF PVG is in the moderate-to-high and high combined categories.

Umatilla National Forest

- Overview—About half of the Umatilla National Forest is classified as having moderate PSDS (table 6.6, fig. 6.11); the relative percentage PSDS index classified within each upland PVG is shown in table 6.7. Most of the nonforested PVGs cover a small spatial extent, so we might expect more error in classification associated with these groups.
- Spring—Although 94 percent of the forest is in the moderate and moderate-to-high categories combined, only 1 percent of the forest is mapped with high soil drought stress during this time period. Only 5 percent of the forest has low-to-moderate drought stress. All PVGs have substantial portions in moderate and moderate-to-high PSDS categories. All nonforested PVGs have more area classified in the moderate-to-high categories (70 percent or more) compared to forested PVGs. The Dry UW PVG has the highest spatial extent in the high PSDS (14 percent). The spatial extent of category 4 and 5 PSDS for the three

Box 6.3 (continued)

forested PVGs is much lower than the nonforested PVGs. As might be expected, the most drought-prone soils for the forested PVGs is for the Dry UF group, with more than 40 percent in moderate to high. However, 28 percent of the Moist UF and 21 percent of the Cold UF are in moderate to high.

- Summer—About 69 percent of the forest is mapped as having moderate-to-high and high drought stress. All PVGs have more area classified in the moderate to high, and high PSDS categories. Nonforested PVGs have high proportions of drought-prone soils compared to forested PVGs. Ninety-six percent of the Dry UF PVG is in high PSDS, and even 23 percent of the Moist UW and 18 percent of the Cold UF are high. However, almost twice as much of the Dry UF group is in the high category (42 percent), and about 75 percent of this PVG is in the moderate-to high-and high categories combined.

Wallowa-Whitman National Forest.

- Overview—Similar to the Umatilla National Forest, about half of the Wallowa-Whitman National Forest is classified in moderate PSDS (table 6.6, fig. 6.12), and 80 percent is in moderate and moderate-to-high categories combined.
- Spring—About 9 percent of the forest is in the high PSDS category, and only 2 percent is in the low-to-moderate category. All PVGs have substantial portions in moderate and moderate-to-high PSDS (table 6.7). Unlike the Umatilla National Forest, substantial portions of the PVGs are in high PSDS status. This forest has large areas of glaciated terrain with scoured landscapes and shallow rocky soils and fewer deep ash caps as compared to the Umatilla. All of the nonforested PVGs have more area classified in moderate-to-high categories compared to forested PVGs. The Moist UW PVG has the highest spatial extent in high PSDS (>50 percent). Most of the nonforested PVGs cover a small spatial extent, so we might expect more error in classification associated with these groups. The spatial extent of categories 4 and 5 PSDS for the three forested PVGs is lower than for nonforested PVGs. The Cold UF PVG has the most drought-prone soils (30 percent) in moderate-to-high, most likely a function of shallow rocky, soils and glaciated terrain in these areas.
- Summer—About 69 percent of the forest is mapped as moderate-to-high and high potential drought stress (categories 4 and 5 combined). As with the Umatilla National Forest, all PVGs have more area in the moderate-to-high and high PSDS categories. Similar to spring patterns described above, nonforested PVGs have high proportions of drought-prone soils compared to forested PVGs. Eighty-eight percent of the Dry US PVG and 86 percent of the Moist UW have high PSDS. Values are lower for the forested PVGs, but both the Cold and Dry UF PVGs have substantial portions in the high category.

Common undergrowth species include herbs and dwarf shrubs. Areas with physiographic and soil characteristics suitable for supporting forests with at least moderate canopy cover are frequently dominated by grouseberry (*Vaccinium scoparium* Leiberg ex Coville). Areas with steeper slopes or shallower soils support open-canopy stands and herb-dominated undergrowth, often featuring elk sedge (*Carex geyeri* Boott), Ross' sedge (*C. rossii* Boott), and western needlegrass (*Achnatherum occidentale* ssp. *occidentale* [Thurb. ex S. Watson] Barkworth).

Cold UFs at the highest elevations often contain whitebark pine, and these open forest communities often have undergrowth similar to flora in subalpine and alpine meadows, including greenleaf fescue (*Festuca viridula* Vasey), prickly sandwort (*Eremogone aculeata* [S. Watson] Ikonn.), mountain heath (*Phyllodoce empetriformis* [Sm.] D. Don), and poke knotweed (*Aconogonon phytolaccifolium* var. *phytolaccifolium* [Meisn. ex Small] Small ex Rydb.).

Species of concern in this PVG include whitebark pine, limber pine (*Pinus flexilis* E. James), and mountain hemlock (*Tsuga mertensiana* [Bong] Carrière) (box 6.2). These forests are an important component of the landscape from a biodiversity perspective, and for snow retention and hydrologic function.

Potential future changes—

At broad scales, forests of western North America can be partitioned into energy-limited versus water-limited domains (Littell and Peterson 2005, Littell et al. 2008, McKenzie et al. 2003b, Milne et al. 2002). Energy-limiting factors are chiefly light (e.g., productive forests where competition reduces light to most individuals) and temperature (e.g., high-latitude or high-elevation forests). Tree growth in energy-limited forests appears to be responding positively to warming temperatures over the past 100 years (McKenzie et al. 2001). Productivity is projected to increase in subalpine and alpine zones across the Pacific Northwest (Latta et al. 2010).

High-elevation forests may increase in productivity in response to moderate warming and elevated atmospheric carbon dioxide.

High-elevation forests may increase in productivity in response to moderate warming and elevated atmospheric carbon dioxide. Longer growing seasons, reduced snowpack and depth, and warmer summer temperatures (day and night) associated with future warming may promote increased tree growth within the treeline ecotone and an upward advance of treeline in some locations. However, a recent worldwide study concluded that there was evidence of treeline advance over the last century at only about half of the sites throughout the world with long-term measurements (Harsch et al. 2009). Upward migration of treeline must include the successful recruitment of tree seedlings, first within and then above the current treeline ecotone. This mechanism depends on microsite facilitation (Smith et al. 2003) and may be limited by unsuitable topographic and edaphic conditions of

upslope areas, wind exposure, and patterns of snow distribution (Holtmeier and Broll 2012, Macias-Fauria and Johnson 2013). Zald et al. (2012) noted that the importance of snow, the mediation of snow by interacting and context-dependent factors in complex mountain terrain, and the uncertainty of climate-change impacts on snow creates a challenge for understanding how these ecotones may respond to future climate conditions.

A high-elevation site in the Blue Mountains that is presently at the transition between Cold UF and Moist UF and was forested (fig. 6.3) supported an *Agropyron*-dominated grassland/ponderosa pine parkland during early- to mid-Holocene warming. A rapid gain in ponderosa pine in the middle of this record indicates the influence of this species under drier and warmer conditions (Hansen 1943). Paleoecological evidence suggests Cold UFs may be converted to herbaceous parklands with ponderosa pine, or the importance of ponderosa pine may increase under warmer and drier scenarios. Douglas-fir woodland types, which are found in the Okanogan valley of southern and central British Columbia and western Montana, may also be a potential analog for what could happen in some areas of the Blue Mountains that are currently dominated by cold forest species.

Species distribution models project that suitable climate available for most cold upland tree species will be moderately reduced to nonexistent in the Blue Mountains by the end of the 21st century (table 6.2). MC2 projects major loss of subalpine forest climate habitat as well (fig. 6.5). Based on model output, Cold UFs may be vulnerable to climate change, and high-elevation mountains (e.g., Wallowa Mountains, Seven Devils) may serve as refugia for subalpine species. Devine et al. (2012) considered subalpine fir, Engelmann spruce, and western white pine (*Pinus monticola* Douglas ex D. Don) to be highly susceptible to climate change (table 6.2), although lodgepole pine has a lower susceptibility score. Although western white pine also has a high susceptibility score (Devine et al. 2012), its generalist life history (Rehfeldt et al. 1984) may confer phenotypic plasticity, allowing it to better adjust to changing environmental conditions (box 6.2). Another study, which suggests that subalpine species were among the most vulnerable to recent climatic variation, assigned lodgepole pine and western hemlock as the first and second most vulnerable species, respectively, in the Blue Mountains (Waring et al. 2011).

Disturbance—

Historically, large-scale disturbances have been infrequent in the subalpine and alpine zones but can still play an important role in shaping vegetation distribution and abundance. Smaller scale disturbances, such as windthrow, are common in the Cold UF (fig. 6.13). Deep snowpacks allow only a short fire season, fuels are often wet,

Figure 6.13—Windthrow disturbance in a Cold Upland Forest.

and spatial discontinuities inhibit fire spread (Agee 1993). However, forest vegetation can be greatly altered by rare wildfire events, and tree establishment following stand-replacing wildfires can require decades to centuries for trees in subalpine forests that do not include lodgepole pine (Agee and Smith 1984, Little et al. 1994). In the future, wildfire events in subalpine systems may be more common as the summer dry period grows longer. Recovery of subalpine forests following wildfire requires nearby seed sources, an extended period of favorable climate, and favorable biotic and abiotic microsite conditions (Bansal et al. 2011, Stueve et al. 2009,

Zald et al. 2012). Seral whitebark pine communities could benefit from increased fire occurrence but depend on Clark's nutcracker (*Nucifraga columbiana* (A. Wilson, 1811)) for dispersal (Arno and Hoff 1990, Barringer et al. 2012, Keane et al. 2012). However, even with warmer temperatures, most Cold UFs would continue to support high- and mixed-severity fire regimes.

A warmer climate could increase the potential for insect and disease outbreaks (box 6.2). Insects in Cold UFs include nonnative balsam woolly adelgid (*Adelges piceae* (Ratzeburg)) in subalpine fir, spruce beetle (*Dendroctonus rufipennis* [Kirby, 1837]) in Engelmann spruce, mountain pine beetle in lodgepole pine, and larch dwarf mistletoe (*Arceuthobium campylopodum* Engelm.) in western larch. Whitebark pine populations are currently declining owing to introduced white pine blister rust (*Cronartium ribicola* A. Dietr.) and mountain pine beetles. The status of limber pine, another species susceptible to white pine blister rust, is unknown. The balsam woolly adelgid is increasing in the Blue Mountains and may be a major stressor of subalpine fir in future decades.

Synthesis—

Based on multiple lines of evidence from modeling and autecological assessment, climate change is likely to produce significant changes in Cold UFs over time, including altered growth, altered phenology, and establishment and persistence of trees in current meadow communities. However, results from experimental and observational studies are not as clear and even suggest potential contrary responses. Cold UFs may be converted to high-elevation herbaceous parklands or woodlands with ponderosa pine or Douglas-fir under warmer and drier scenarios. Remnant populations may persist in the highest of elevations within the Blue Mountains (e.g., Wallowa Mountains). The cool dry PAG may be more tolerant of future warming compared to the cold dry and cold moist PAGs. However, increased wildfire may constrain tree reestablishment in these slow-growing systems, particularly for sites without serotinous lodgepole pine as a common, prefire component. Increased insect and disease activity with climate change may also increase stress and mortality in these forests.

Cold Upland Shrub

Cold Upland Shrublands (US; fig. 6.14) occur at moderate to high elevations in climate conditions similar to Cold UFs. This PVG is not common in the Blue Mountains (table 6.4). The principal species characterizing these shrublands form associations that range from xeric to mesic, although this system may be associated with exposed sites, rocky substrates, cold air drainages, and dry conditions that limit tree growth. This PVG includes cold very moist, cold moist, cool dry, and cool moist

Figure 6.14—An example of a Cold Upland Shrubland community from the south slope of the Greenhorn Mountains.

US PAGs. Typical species include Sitka alder (*Alnus viridis* ssp. *sinuata* [Regel] Á. Löve & D. Löve), mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana* [Rydb.] Beetle), and shrubby cinquefoil (*Dasiphora fruticosa* [L.] Rydb.). However the habitat conditions for these species are quite different. Cold moist shrublands are common in subalpine and alpine systems in the Wallowa Mountains at about 2200 m elevation. Cold very moist shrublands consist of Sitka alder in snow slides in the northern portion of the Blue Mountains. The cooler upper shrubland PAGs are more common in the montane zone.

Paleoecological studies generally suggest an increase in sagebrush during warmer conditions in the early to mid Holocene.

Potential future changes—

Species distribution model results for sagebrush indicate complete loss of habitat for this species in the future, although only one model was available for assessment. MC2 model results project a major loss of subalpine forest climate habitat, and an analogous loss of Cold US habitat might be inferred from this result (fig. 6.5). In contrast, the Biome-BGC model projects increased net productivity in eastern Oregon shrublands after 2030, including areas currently occupied by sagebrush (Reeves et al. 2014). Note that paleoecological studies generally suggest an increase in sagebrush during warmer conditions in the early to mid Holocene (fig. 6.3), providing an important context for interpretation of modeling studies.

Disturbance—

Large-scale disturbances are infrequent in subalpine and alpine systems but can still play an important role in shaping vegetation distribution and abundance. Deep snowpacks produce a short fire season, fuels are often wet, and spatial discontinuities can inhibit fire spread (Agee 1993). In the future, wildfire events in subalpine systems may be more common, and shrub species may be able to more rapidly regenerate compared to subalpine tree species. Sitka alder and shrubby cinquefoil can survive low-to-moderate-intensity fires and resprout vigorously. However, mountain big sagebrush is readily killed by fire and requires at least 15 years or more to recover after fire (Bunting et al. 1987).

Synthesis—

Climate change is likely to produce changes in Cold US over time, including changes in plant growth and phenology. Although limited model data suggest that sagebrush may be highly vulnerable to climate change, paleoecological evidence does not support this inference. Warming at higher elevations and a longer growing season may increase productivity in Cold US. Increased future fire activity in subalpine environments may not constrain shrub establishment as much as tree establishment in these slow-growing areas, although recovery times may still be long.

Cold Upland Herb

Subalpine and alpine meadows are found at high elevations where temperatures are too cold or snow covers the ground too long for trees to grow. This PVG is not common in the Blue Mountains (table 6.4) but includes cold moist, cold dry, cool moist, and cool dry upland herb (UH) PAGs. Typical plant communities consist of grasslands with greenleaf fescue and Idaho fescue (*Festuca idahoensis* Elmer), elk sedge and Hood's sedge (*Carex hoodii* Boott). Many of these high-elevation grasslands have been degraded from livestock and elk grazing (Irwin et al. 1994; Johnson 2003). Partial to complete loss of vegetation cover resulted in significant erosion of deep, fine textured, loess soils and decreased productivity, and greenleaf fescue grasslands in the Wallowa Mountains are still recovering (Johnson 2003, Reid et al. 1991). Some sites appeared to have transitioned to stable forb communities with poke knotweed, Nuttall's linanthus (*Leptosiphon nuttallii* ssp. *nuttallii* [A. Gray] J.M. Porter & L.A. Johnson) and alpine golden buckwheat (*Eriogonum flavum* Nutt.). This PVG includes many rare and endemic meadow species (fig. 6.15).

Potential future changes—

Snowpack melting and subsequent soil heating have been shown to influence flowering, growth phenology, and vegetation community patterns in alpine meadows

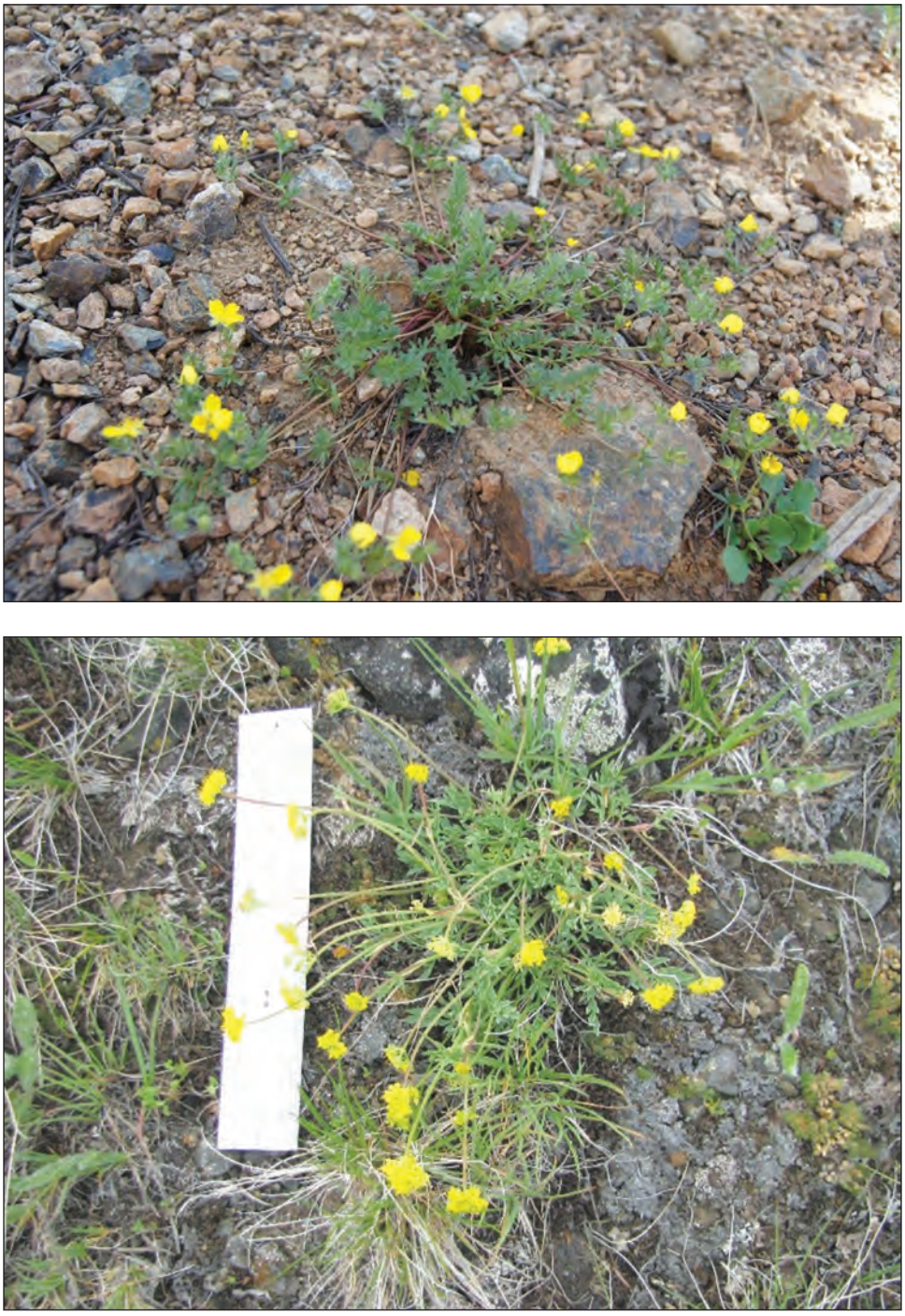

Figure 6.15—Examples of rare and locally endemic species found in alpine and subalpine environments. On the top is *Potentilla* sp. from serpentine substrates in the Greenhorn Mountains. On the bottom is Greenman's biscuitroot from Mount Howard, Wallowa County, Oregon.

(Dunne et al. 2003, Inouye 2008). Spatial variability in snowpack persistence influences flowering and growth phenology within species and, to the extent that spatial patterns of snowmelt are consistent among years, also influences species composition and community types in alpine meadows (Canaday and Fonda 1974, Evans and Fonda 1990).

Subalpine conifers have been documented as infilling alpine tundra and meadows in the Pacific Northwest, a trend related to periods of warmer climate (Rochefort and Peterson 1996, Woodward et al. 1995). Snow-dominated meadows on Mount Rainier (Washington) (Rochefort and Peterson 1996) and in the Olympic Mountains (Woodward et al. 1995) have been infilled by subalpine fir and mountain hemlock during the 20th century in association with warm phases of the Pacific Decadal Oscillation. Tree establishment has also been documented in the Wallowa and Elkhorn Mountains (Skovlin et al. 2001) and in some locations, may have been aided by the heavy grazing pressures in the early 1900s. Because of the shallow, rocky soils that characterize this PVG, it is unlikely that tree encroachment will be a major problem in the future. Meadow plant species are able to colonize new habitat that was previously covered by ice or bare ground under more favorable climatic conditions, but the process of soil formation is slow. It is more likely that subalpine trees will establish in Cold US areas, many of which are characterized by deeper soils.

The effects of climate change on alpine or subalpine meadow species have not been modeled. However, model results for subalpine tree species and MC2 results generally show loss of subalpine and alpine habitat, which may suggest loss of habitat for other subalpine and alpine life forms.

Disturbance—

As noted above, large-scale disturbances are infrequent in subalpine and alpine systems. In the future, wildfire events in subalpine systems may be more common, and herbaceous species may be able to more rapidly regenerate compared to shrubs and trees. Many species in cold UH communities are fire survivors because they sprout or sucker from roots, root crowns, corms, or rhizomes.

Synthesis—

There is considerable uncertainty about the future spatial extent of subalpine and alpine meadows. Continued warming in future decades could cause the geographic range of upland grass and forbs to contract, expand, or remain the same. Trends will most likely depend on the rates at which meadow species colonize exposed soil following disturbance.

Moist Upland Forests

Moist UFs occur at moderate elevations in the montane vegetation zone, or at low elevations in the subalpine zone (fig. 6.16). They are adjoined by cold forests at their upper edge and by dry forests at their lower edge. They are characterized by slightly longer growing seasons compared to the Cold UF, and have cooler temperatures and higher precipitation than Dry UF.

Late-seral stands are generally dominated by subalpine fir, grand fir, or Douglas-fir, and lodgepole pine or western larch often occur as early-seral species (except where lodgepole pine dominates). Douglas-fir and western white pine are mid-seral species (except in three potential vegetation types where Douglas-fir is the dominant (climax) species). Species of concern in this PVG include Alaska cedar (*Callitropsis nootkatensis* [D. Don] D.P. Little) (box 6.4) and quaking aspen (*Populus tremuloides* Michx.) (box 6.5).

For the Blue Mountains, the Moist UF PVG consists of five PAGs—three in the cool temperature regime (Cool Wet UF, Cool Very Moist UF, Cool Moist UF), and two in the warm temperature regime (Warm Very Moist UF, Warm Moist UF). The Cool Moist UF is the most common component of this PVG. Cool moist forest understories are dominated by forbs, several mid-height shrubs, and a few tall shrubs on the warm end of the PAG. These forests tend to occupy the most productive forested environments in the Blue Mountains because moisture is less

These forests tend to occupy the most productive forested environments in the Blue Mountains because moisture is less limiting.

Figure 6.16—Moist Upland Forest in the Blue Mountains.

Box 6.4—Cold and moist upland conifer species of concern: whitebark pine, limber pine, mountain hemlock, Alaska cedar

Limber pine, Alaska cedar, and mountain hemlock in the Blue Mountains are represented by populations that are separated from the main parts of the species' distributions. A single population of Alaska cedar occurs in eastern Oregon, the Cedar Grove Botanical Area in the Malheur National Forest (Frenkel 1974). This population is likely a relict of a time period when cooler and wetter conditions prevailed and the species occurred across a larger area (Devine et al. 2012). The distribution of limber pine is discontinuous throughout its range and two relict populations occur in the Eagle Cap Wilderness. Infrequent mountain hemlock stands occur in the northern half of the Wallowa Mountains. These rare, disjunct populations represent important genetic resources that may be at risk to climate change and other stressors.

Whitebark pine and limber pine are also threatened by white pine blister rust (Smith et al. 2013, Tomback and Achuff 2010), a nonnative fungus that forms cankers of necrotic tissue that girdle tree stems. Alternate hosts for the fungus are currant (*Ribes*) or the herbs paintbrush *Castilleja*) and lousewort (*Pedicularis*) (Geils et al. 2010). Indirect pathogen-related effects could occur if climate increasingly favors blister rust. Higher variability in weather conditions may create conditions conducive to infection, although drier summers could inhibit the formation and spread of rust spores and fruiting body development.

High-elevation species can also be rapidly altered by rare wildfire events, because recovery from stand-replacing wildfires is slow. This is especially true for species with limited distributions. Mountain hemlock in the Wallowa Mountains is favored by a lack of fire, because a stand-replacing fire here would likely favor lodgepole pine (Devine et al. 2012). The Alaska cedar grove in the Malheur National Forest burned in 2006, and most of the mature trees were killed. In 2014, wildfire burned through some of the limber pine habitat in the northern Wallowa Mountains, although the effects on the trees is unknown.

Fire effects on whitebark pine are difficult to generalize. Mixed-severity and stand-replacing fires are beneficial to seral whitebark pine communities, because the pine is better adapted to recolonize burned sites compared to more shade-tolerant subalpine fir that may outcompete pine in the absence of fire (Arno and Hoff 1990, Keane et al. 2012). With declining populations, loss of cone-bearing trees with potential resistance to blister rust will limit future management options. In addition, probability of dispersal by Clark's nutcracker declines with diminishing cone production (Barringer et al. 2012).

During the past two decades, warmer temperatures have allowed mountain pine beetles to shift upward and persist in higher elevation forests (Logan et al. 2010). Mountain pine beetle is the primary cause of whitebark pine mortality at Crater Lake National Park, presently at a rate of 1 percent annually (Murray 2010), and may have been facilitated by an increasing late-summer dry season (Daly et al. 2009). Longer, high-elevation growing seasons could enhance the growth of subalpine tree species (e.g., Bunn et al. 2005, Peterson and Peterson 2001). However, because whitebark pine grows so slowly, it may be at a competitive disadvantage compared to other species.

Box 6.5—Quaking aspen in upland forests

Because quaking aspen stands in the Blue Mountains are typically less than 1 ha and fragmented across the landscape, they are vulnerable to a warmer climate and other stressors (table 6.2). Aspen is an early seral component of forest communities, and stands have been declining in number, area, and stocking density. Current stressors include competition with shade-tolerant conifers, herbivory by livestock and native ungulates, and fire exclusion. Aspen can occur in cold, moist, and dry upland forest potential vegetation groups, although the only aspen potential vegetation type is in the Cool Very Moist plant association group in Moist Upland Forest (Swanson et al. 2010).

Sudden aspen death (SAD) was noted in the Rocky Mountains over a decade ago and has affected large areas of aspen in southwestern Colorado (Worrall et al. 2010). Sudden aspen death is characterized by rapid, synchronous branch dieback, crown thinning, and mortality of stems, without the involvement of primary pathogens and insects. Some affected stands may fail to produce suckers in response to crown loss and mortality. Worrall et al. (2008) proposed a disease model for SAD as follows: (1) predisposing factors include low elevations, south/southwestern aspects, physiological maturity, and low stand density, (2) inciting factors are acute drought with high temperatures during the growing season, and (3) contributing factors are insects and pathogens that tend to invade and kill stressed trees.

A warmer climate and drier growing season could increase susceptibility to SAD in the Blue Mountains, because aspen requires mesic soil moisture conditions, and moisture stress is an underlying factor for SAD (Worrall et al. 2010). Aspen stands in the Dry Upland PVG are already near their soil moisture limit for survival, and increased loss of aspen in the Malheur National Forest might be expected at lower elevations.

Increased frequency of stand-replacing fires in the future may favor aspen regeneration, because aspen respond to stand-replacing fire with abundant vegetative propagation (suckering) and rapid growth. Increased low-severity fire may benefit aspen if competition with conifers is reduced and soil moisture status is improved, although once competing conifers like ponderosa pine, Douglas-fir, and grand fir are relatively large, their fire tolerance exceeds that of aspen (Swanson et al. 2010). In addition, severe fires and reburns may kill shallow root systems and eliminate small clones, particularly in stressed stands.

limiting, and their temperate nature is demonstrated by high species diversity and closed forest structure. High species diversity pertains to both forest overstory and understory composition.

Moist-forest understories are dominated by forbs, some mid-height shrubs, and a few tall shrubs in warmer locations. Moist-site plants such as bride's bonnet (*Clintonia uniflora* [Menzies ex Schult. & Schult. f.] Kunth), twinflower (*Linnaea borealis* L.), false bugbane (*Trautvetteria caroliniensis* [Walter] Vail), western swordfern (*Polystichum munitum* [Kaulf.] C. Presl), and British Columbia wildginger (*Asarum caudatum* Lindl.) occur here, but most mesic environments in the Moist UF PVG

are dominated by thinleaf huckleberry (*Vaccinium membranaceum* Douglas ex Torr.). Moist forests at the warm end of the temperature spectrum (Warm Very Moist and Warm Moist PAGs) include mid or tall shrubs such as Rocky Mountain maple (*Acer glabrum* Torr.), mallow ninebark (*Physocarpus malvaceus* [Greene] Kuntze), and oceanspray (*Holodiscus discolor* [Pursh] Maxim.).

Potential future changes—

Many (but not all) productive moist upland forests at higher elevation are more energy limited than water limited. Light is a limiting factor in productive forests where competition reduces light to most individuals. Tree growth in energy-limited forests appears to be responding positively to warming temperatures over the past 100 years (McKenzie et al. 2001). In the Blue Mountains, lower elevation moist forests may transition to being primarily water limited, particularly areas without much ash or loess, which would enhance water holding capacity.

Moderate warming may lead to a positive response and increased productivity. However, more extreme warming and increased drought stress, particularly at lower elevations and in the southern portion of the Blue Mountains (e.g., Malheur National Forest) will likely cause decreased tree growth and forest productivity. However, suitable climate habitat currently occupied by Cold UFS may offset these losses.

Paleoecological evidence demonstrates that during early to mid-Holocene warming, areas that are currently between Cold UF and Moist UF were forested (fig. 6.3), supporting *Agropyron*-dominated grassland/ponderosa pine parkland. Areas currently dominated by grand fir and Douglas-fir were dominated by ponderosa pine.

Species distribution models (table 6.1) project that the suitable climate available for most tree species characteristic of this PVG will be reduced to nonexistent by the end of the 21st century, although some models project minor loss of climate habitat for Douglas-fir. Losses were more extreme for the Malheur National Forest (where this PVG is currently limited in distribution) and southern portions of the Umatilla and Wallowa-Whitman National Forests. In contrast, MC2 projects either very small (three scenarios) or moderate (one scenario) increases in this forest type (fig. 6.5), particularly in the northern Blue Mountains. The latter scenario has a large projected increase in annual precipitation, including some increased precipitation in the summer, coupled with only moderate warming. None of the MC2 scenarios project a decrease in this forest type.

As discussed earlier, SDMs tend to produce large reductions in species climate habitat with future novel conditions, so the model results for these species are not surprising. MC2 is not constrained as much by future novel conditions. Moreover,

most GCM scenarios project increased precipitation for the Pacific Northwest (fig. 6.4). Higher precipitation, coupled with warming, and the availability of climate habitat currently occupied by colder forest species, confirms the logic of MC2 model output. One recent study suggests that precipitation may decrease, because slower westerlies associated with jet-stream changes will result in less orographic lifting (enhancement) of precipitation for mountainous regions (Luce et al. 2013). MC2 does not account for decreasing orographic effects and tends to underestimate the effects of decreased snowpack expected in the future. Even with these shortcomings, the MC2 projections are more robust, and it is unlikely that any future climate would result in a large loss of moist forest habitat.

Disturbance—

Moist upland forests generally support mixed-severity fire regimes, although low- and high-severity regimes also occur (Stine et al. 2014). High levels of coarse woody debris, litter, and live biomass can produce occasional large, high-severity wildfires when fire weather and dry fuel conditions coincide. Increased frequency and duration of summer drought would allow wildfires to burn wetter and cooler sites, where high fuel loads become more available when fuel moisture is low. However, the primary effect of severe fire in these forests is to reduce total standing biomass rather than change forest composition.

Warming temperatures could increase the potential for insect and disease outbreaks (box 6.1). Insect and disease agents in Moist UF include western spruce budworm (*Choristoneura occidentalis* Freeman), Douglas-fir tussock moth (*Orgyia pseudotsugata* [McDunnough, 1921]), Douglas-fir beetle (*Dendroctonus pseudotsugae* Hopkins, 1905), fir engraver (*Scolytus ventralis* LeConte, 1868), spruce beetle, mountain pine beetle in lodgepole pine, Douglas-fir dwarf mistletoe (*Arceuthobium douglasii* Engelm.), larch dwarf mistletoe, and root pathogens (particularly *Armillaria*, *Annosus* root rot (*Heterobasidion annosum* [Fr.] Bref.), and laminated root rot (*Phellinus sulphurascens* [Pilat]).

Many Moist UFs (e.g., those not burned in recent decades) have high stand densities, small trees, and few large fire-tolerant trees, and are dominated by shade- and fire-intolerant tree species (Stine et al. 2014). Because of the increase in surface and canopy fuel loads, there is a greater risk of large and severe wildfires, particularly when fire weather conditions are severe. Increased stand densities also increase competition for water and nutrients, which leads to higher susceptibility to insect and disease outbreaks. These conditions threaten the long-term survival of large trees and sustainability of these forests in general.

Synthesis—

Paleoecological and some model evidence suggest that climate change will cause moderate to extreme loss of Moist UFs and characteristic species. However, MC2 model results suggest the opposite. Future warming with increased precipitation may lead to increased importance of this PVG across the landscape. This outcome is somewhat supported by recent trends in response to warming in energy-limited forests. Unlike Cold UFs, these forests may be able to adapt to future climate change by expanding into new available habitats. Warm and very warm moist forest PAGs may be able to better adapt to warming compared to cool PAGs. However, increased summer drought stress may make these forests more vulnerable to other stressors, particularly at lower elevations and on southern sites in the Blue Mountains. Wildfire activity and insect and disease outbreaks will most likely increase with future warming, and may reduce the distribution of this PVG.

Future warming with increased precipitation may lead to increased importance of this PVG across the landscape.

Dry Upland Forest

Dry UFs generally occur at low to moderate elevations in the montane vegetation zone (fig. 6.17). Climate varies with elevation, but common features include warm, dry summers, with warm to hot daytime temperatures and cool nighttime temperatures, and cold, wet winters. Much of the annual precipitation falls as snow in winter or during spring rainstorms. Late-seral stands are dominated by ponderosa pine, grand fir, or Douglas-fir, and ponderosa pine or Douglas-fir also function as early- or mid-seral species depending on plant association. Western juniper is expanding rapidly into this PVG as a result of fire exclusion and climate change, moving upward from a foothills woodland zone located below the montane zone (Knapp and Soulé 1998, Miller et al. 2005). Dry UFs are adjoined by Moist UFs at their upper edge, and by the woodlands and shrublands of the foothills vegetation zone at their lower edge.

The Dry UF PVG consists of three PAGs—one from the warm temperature regime (Warm Dry UF), and two from the hot temperature regime (Hot Moist UF, Hot Dry UF). Warm, dry forests are the most common forest zone in the Blue Mountains (table 6.4), and they have a long history of human use for commodity purposes (e.g., livestock grazing and timber production; fig. 6.18). Effective fire exclusion has led to significant changes in species composition, forest structure, and stand density. Dry UF sites were historically dominated by ponderosa pine, which is well adapted to survive in low-severity fire regimes.

Common dry UF understory species include graminoids and mid-height shrubs. Elk sedge and pinegrass (*Calamagrostis rubescens* Buckle) are ubiquitous, and

Figure 6.17—View of Dry Upland Forest from Lone Rock, Malheur National Forest.

Becky Kerns

Figure 6.18—Domestic livestock grazing is common in the Blue Mountains, particularly in dry upland ponderosa pine forests.

white spirea (*Spiraea betulifolia* Pall.), snowberry (*Symphoricarpos* spp.), mallow ninebark, antelope bitterbrush and curl-leaf mountain-mahogany are common shrubs. On hot and dry sites, mountain big sagebrush, bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] Á. Löve), and western juniper are important species.

Potential future changes—

Dry UFs in the Blue Mountains are water limited, and productivity is projected to decline in a warmer climate (Latta et al. 2010). Water stress during the warm season is the primary factor limiting tree growth at low elevations in the Pacific Northwest (Brubaker 1980), and negative water balances constrain photosynthesis (Hicke et al. 2002), although this may be partially offset if carbon dioxide fertilization significantly increases water-use efficiency in trees (Neilson et al. 2005). For example, Littell (2006) found that most montane Douglas-fir forests in the northwestern United States appear to be water limited, and water limitation will increase in warmer climate (assuming that precipitation does not increase in the summer).

Kusnierczyk and Ettl (2002) found that ponderosa pine growth is positively correlated with precipitation in the fall and winter prior to the growing season, but is not significantly correlated with temperatures, suggesting that ponderosa pine growth is more sensitive to changes in water balance than to temperature. Ponderosa pine may be able to adapt to increased summer drought stress by allocating more biomass to sapwood conducting area, reducing the ratio of leaf area to sapwood area and presumably reducing risks of hydraulic failure (Callaway et al. 1994). This is an example of phenotypic plasticity and not genetic adaptation, suggesting that this could be a future response to changing climate (Maherali et al. 2002).

Increased drought stress will likely result in decreased tree growth and forest productivity in Dry UFs in the Blue Mountains, particularly at the current dry forest-steppe ecotone. Areas with increased tree density owing to recent fire exclusion may be particularly vulnerable to future climate change because of increased drought stress, although suitable climate habitat currently occupied by Moist UFs may offset these losses.

Ponderosa pine was generally abundant during periods of major climatic change in the past (Whitlock 1992). Hansen (1943) documents an increase in ponderosa pine pollen during the past (inferred early to mid-Holocene) at a site currently dominated by subalpine species. Phytolith data from sites in the Blue Mountains indicate that the ponderosa pine forests were located at a higher elevation than today and during the early to mid-Holocene (fig. 6.3). Paleoecological evidence suggests that Dry UFs may persist but will be located farther north and at higher elevations.

Species distribution models suggest that climate habitat for ponderosa pine and Douglas-fir will decrease greatly, particularly in the southern Blue Mountains (table 6.2). Other model results suggest that Douglas-fir distribution would decrease only slightly or even increase. Vulnerability scores suggest that Douglas-fir might be more vulnerable to climate change than ponderosa pine (table 6.2). Output from MC2 suggests that temperate needleleaf forest and dry temperate needleleaf forest (both crosswalk to the dry upland PVG) will decrease in relative abundance on the landscape. In summary, model output suggests that dry upland forests will decrease somewhat in abundance across the landscape, particularly in the southwestern portions of the Blue Mountains. However, there is considerable model disagreement about the magnitude of this effect.

Disturbance—

For dry, frequent-fire forests, present-day stand structure, species composition, fuel accumulation and associated risk of severe fires, and insect and disease outbreaks are generally regarded as historically uncharacteristic and undesirable. In the Blue Mountains, grand fir has recently encroached in many ponderosa pine stands, and ponderosa pine has moved down in elevation into adjacent woodlands. These conditions are commonly attributed to decades of fire exclusion and suppression, timber harvesting, historical periods of overgrazing, and shifts in climate (Hessburg et al. 2005, Tiedemann et al. 2000, Wright and Agee 2004). The role and importance of fire as a disturbance process (Agee 1993, Fulé et al. 1997, Hessburg and Agee 2003) and disruption of fire regimes coinciding with Euro-American settlement and associated fire exclusion (Covington and Moore 1994, Hessburg et al. 2005, Swetnam et al. 1999) have been extensively documented. Management goals for these forests, such as reducing the risk of severe wildfires and sustaining and promoting biodiversity, have prompted the use of thinning and prescribed fire to reduce fuels, lower stand densities, and alter stand composition. Prescribed fire simulates the frequent low-intensity surface fires considered characteristic of the historical environment of dry, fire-prone forests throughout the interior West prior to nonindigenous settlement (Agee 1993, Cooper 1960, Covington and Moore 1994, Weaver 1943). However, prescribed fires are typically conducted mostly in the spring and autumn, whereas historical wildfires occurred mostly during summer and early autumn. The ecological implications of the current seasonal timing of prescribed burns have rarely been explored (but see Kerns et al. 2006).

Dry forest sites were historically dominated by ponderosa pine because it is well-adapted to survive in a low-severity fire regime (Agee 1996, Hall 1976, 1980). Heyerdahl et al. (2001) noted that within the same plant association, fires were more

Longer summer droughts will potentially lengthen the fire season for most dry coniferous forest types.

than twice as frequent at plots in the southern watersheds of the Blue Mountains compared to northern watersheds. Plots to the south experienced 15 and 13 fires on average (1687–1900), whereas plots to the north experienced only six, leading to mean fire-return intervals of 14 to 35 years. Heyerdahl et al. (2001) also noted that fire frequency varied more based on location (south versus north) rather than forest type. It is likely that at least some mixed- and high-severity fires also occurred within the matrix of primarily low-severity fire prior to Euro-American settlement (Hessburg et al. 2007).

Longer summer droughts will potentially lengthen the fire season for most dry coniferous forest types and may increase the risk of large wildfires. Because of the current accumulation of live and dead fuels, large and severe wildfires may become the norm for these forest types. At lower elevations, these fires may cause conversion to shrublands or grasslands, a trend that is supported by MC2 output, particularly for hotter and drier scenarios.

A warmer climate could increase the potential for insect and disease outbreaks (box 6.1). Insect and disease agents include western spruce budworm, Douglas-fir tussock moth (frequently where Douglas-fir and grand fir invaded stands historically dominated by ponderosa pine), Douglas-fir dwarf mistletoe, western dwarf mistletoe (*Arceuthobium campylopodum* Engelm.), and bark beetles (*Dendroctonus* spp.). Pine white (*Neophasia menapia* [C. Felder and R. Felder, 1859]), or pine butterfly, is also important in ponderosa pine. A recent pine white outbreak in the Blue Mountains caused significant defoliation (Kerns and Westlind 2013) (fig. 6.19), although mortality from this event is not expected to be high.

Synthesis—

Some Dry UFs may undergo undesirable changes in the face of future climate change. These forests have already experienced a long history of human land use. Many Dry UFs are already experiencing severe and uncharacteristic wildfire, and equally uncharacteristic insect and disease outbreaks, which will most likely increase in the future. It is likely that the hottest and driest sites will shift to woodland or steppe vegetation. Species characteristic of hot dry PAGs may be better adapted to future conditions and these species may become more common. Some model output suggests that Douglas-fir and ponderosa pine may decrease in the future, although paleoecological evidence conflicts somewhat with this conclusion, suggesting that ponderosa pine was able to adapt to warmer climate by migrating north or up in elevation. However, the extent to which these species can adapt under current and future stressors is unclear. The overall vulnerability assessment score for ponderosa pine is quite low, whereas Douglas-fir has a somewhat higher score

Figure 6.19—Pine white butterfly defoliation is extensive in a ponderosa pine stand in the southern Blue Mountains (Malheur National Forest, autumn 2011).

(table 6.2). Given the strong paleoecological evidence regarding the persistence of ponderosa pine, coupled with its potential low vulnerability and availability of habitat currently occupied by moist forests, it is likely that this forest type will persist and remain an important component of the landscape, although shifts in the distribution of Dry UFs and changes in relative abundance of different PAGs might be expected (or the formation of novel plant associations).

Dry and Moist Upland Woodlands

Upland woodlands (UW) in the Blue Mountains occupy the transition zone between shrublands at lower elevation and dry upland forests at higher elevation (fig. 6.20). These woodlands occupy the driest of the tree-dominated vegetation zones in the Blue Mountains. Summers are hot and very dry, and winters are cold and relatively wet. Annual precipitation in western juniper savannas and woodlands ranges from 13 to 75 cm, but most sites fall within the range of 25 to 50 cm yr^{-1} (Gedney et al. 1999). Much of this precipitation falls during the winter as rain or snow.

There are two woodland PVGs in the Blue Mountains—Moist UW (hot moist UW PAG) and Dry UW (hot dry UW PAG). These two PVGs are discussed together

Figure 6.20—A stand with a mixture of western juniper and ponderosa pine.

for this assessment, although the Dry UW PVG is the least common PVG. Western juniper is dominant in all of the PAGs. The Moist UW PVG is characterized by understory shrubs such as mountain big sagebrush, curl-leaf mountain-mahogany, antelope bitterbrush or the grasses Idaho fescue and bluebunch wheatgrass. The Dry UW PVG is characterized by understory species such as bluebunch wheatgrass, low sagebrush (*Artemisia arbuscula* Nutt.), and scabland sagebrush (*A. rigida* [Nutt.] A. Gray).

Western juniper has expanded its range in the interior Pacific Northwest during the past 130 years, invading and creating savannas and woodlands in semiarid ecosystems that were formerly shrub-steppe and grassland communities (Miller et al. 2000). More than 90 percent of the 3.2 million ha of current juniper savannas and woodlands developed in the past 100 years (Miller et al. 2000). The area of juniper forest and woodland is estimated to have increased fivefold between 1936 and 1988 (Gedney et al. 1999). Much of this expansion is attributed to heavy livestock

grazing and reduced fire frequencies (Miller et al. 2000), but woodland expansion may have started between 1850 and 1870 in some areas during wet, mild climatic conditions (Miller et al. 2005). Western junipers tolerate very dry conditions and can live for up to 1,000 years in the absence of disturbance (Miller et al. 2000).

Potential future changes—

Minimal experimental research has been conducted on the response of junipers to elevated carbon dioxide or warming temperatures. Western junipers might be expected to benefit from increasing atmospheric carbon dioxide if it reduces stomatal conductance and delays depletion of deep soil water, although it is unclear if improved water-use efficiency would significantly increase growth, or simply reduce drought stress. Increased growth of western juniper in recent decades suggests that elevated carbon dioxide may be increasing growth through increased water-use efficiency (Knapp et al. 2001), but such evidence is not conclusive.

Tree-ring analyses suggest that western juniper growth is driven primarily by soil moisture availability and drought (Knapp et al. 2004, Knutson and Pyke 2008). Growth was positively correlated with winter and spring precipitation (October to June) and negatively correlated with spring and summer temperatures (Knutson and Pyke 2008). Growth sensitivity to drought was greatest at lower elevations and on steep, rocky sites. Cold winter temperatures also exert influence on vegetation communities and are almost as important as drought for limiting photosynthesis in juniper woodlands (Runyon et al. 1994). Extreme cold temperatures can also function as a disturbance agent (Knapp and Soule 2005).

Juniper woodlands may increase in abundance in future scenarios associated with increased winter and spring precipitation. However, higher spring and summer temperatures may negatively affect juniper woodlands, particularly at lower elevations and for hot dry UW PAGs. Areas with increased juniper density from recent land use history may be particularly vulnerable to future climate change. However, suitable climate habitat currently occupied by dry UFs may offset these losses.

Recent research suggests that increased precipitation intensity (the same amount of precipitation falling in fewer storms) can result in moisture reaching deeper portions of the soil profile, and that juniper and other woody plants are able to access the deeper moisture better than grasses and herbs. Woody plant encroachment observed over the last century could continue into the future should precipitation intensity increase (Kulmatiski and Beard 2013).

As temperatures warmed during the early Holocene, western juniper began migrating north into its present range in the Pacific Northwest. Since the arrival

of western juniper in central and eastern Oregon (circa 6,600 to 4,800 BP), northeastern California, and southeastern Idaho, its abundance and distribution have fluctuated (Miller et al. 2005). Dry climatic periods can cause regional declines of juniper, whereas wetter periods (wet summers, mild winters) can cause expansion. Paleoecological evidence suggests that juniper woodlands may increase in abundance in future scenarios associated with increased winter temperatures (virtually all scenarios) and increased spring and summer precipitation. The Moist UW PVG may become more abundant with this type of scenario. A warm, dry scenario may result in decreased western juniper abundance, although species characteristic of the Dry UW PVG may be better adapted to these conditions. Data from the Blue Mountains indicate that the shrub-steppe boundary was higher in elevation in the past during the warmer and drier Holocene (fig. 6.3), and a similar shift in the forest-woodland boundary might be expected.

Model output—

Species distribution models indicate that climate habitat for western juniper, curlleaf mountain-mahogany, and big sagebrush will be almost nonexistent (table 6.2). However, MC2 output shows that temperate needleleaf woodlands will most likely increase in abundance, except for one scenario that represents the least amount of change (fig. 6.5). Assuming process models are more robust than species distribution models for projecting vegetation responses to climate change, we would expect an increase in these PVGs across the landscape, an inference that is consistent with the temperature and moisture tolerances of major species associated with woodland PVGs.

Disturbance—

Historical fire regimes are not well described for juniper savannas and woodlands in Oregon (Agee 1993, Young and Evans 1981). Young junipers have thin bark and are readily killed by fires. Junipers that avoid fires early in their lifespan can subsequently escape injury and death from fire by having thicker bark and suppressing understory herbaceous fine fuels through competition for water (Agee 1993). Consequently, fire-scarred junipers are limited to microsites with limited fine fuel production. Fire scars in scattered or adjacent ponderosa pine forests might suggest a mixed-severity fire regime, with mean fire-return intervals of 15 years to more than a century and occasional large fires (Agee 1993, Miller and Rose 1999, Miller et al. 2005). Romme et al. (2009) noted that in many places in the West, fire-return intervals in juniper woodlands were very long (generally measured in centuries). In this predominantly fuel-limited biome, climate change effects on fire frequency

and severity will likely depend on changes in soil water availability and its effect on understory plant productivity (fuel generation).

Many of these woodlands were also homesteads at the turn of the century, have been subject to livestock grazing for over a century, and are still used for grazing. These systems are also highly changed because of exotic annual grass invasion, particularly after tree harvest and natural disturbances. In general, sites supporting moist and cooler upland woodlands and sites dominated by mountain big sagebrush are more resistant to invasion by juniper than sites dominated by other species or subspecies of sagebrush (e.g., Wyoming big sagebrush [*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young]) found in warmer and drier conditions (Miller et al. 2013).

Higher spring and summer temperatures may negatively affect some very hot and dry juniper woodlands at lower elevations.

Synthesis—

Higher spring and summer temperatures may negatively affect some very hot and dry juniper woodlands at lower elevations. However, it is unlikely that these PVGs will be greatly reduced across the landscape (although locations of juniper woodlands could shift), given the (1) adaptability of juniper to drought, (2) potential availability of habitat currently occupied by dry upland forests, and (3) continued fire suppression policies. In the future, years of wet and mild climatic conditions, particularly above-average spring and summer precipitation, will most likely facilitate the continued expansion of juniper. Increased fire frequency and severity associated with future warming could reverse this trend and lead to conversion of some of these woodlands to persistent grasslands. This disturbance-mediated effect may help reduce ongoing conversion of shrublands and grasslands to juniper woodlands, although grassland dominance may be accompanied by increasing dominance of nonnative annual grasses.

Moist and Dry Upland Shrublands

Moist USs (warm moist, hot moist, very hot moist) and Dry USs (hot dry and warm dry) are not common in the Blue Mountains and are discussed together for this assessment. These PVGs tend to occupy the transition zone between woodlands at the upper elevation and grasslands at the lower elevation, but can also be found in forest openings or near ridge tops at higher elevation (e.g., snow openings, snowberry shrublands). Numerous community types differ according to the dominant grasses and shrubs, and species distribution largely reflects underlying gradients in annual mean precipitation and soil properties (Franklin and Dyrness 1973). Characteristic species for the moist US PVG include mountain big sagebrush, antelope bitterbrush, snowberries, bitter cherry (*Prunus emarginata* [Douglas] Eaton),

curl-leaf mountain-mahogany, and cool season (C3) bunchgrasses (e.g., Idaho fescue, bluebunch wheatgrass, Sandberg bluegrass [*Poa secunda* ssp. *secunda* J. Presl]). The dry US PVG is characterized by low, scabland, and threetip sagebrush (*Artemisia tripartita* Rydb.). The climate is classified as arid to semiarid with low precipitation, hot dry summers, and relatively cold winters, and these shrublands have often been described as cold desert or high desert (Franklin and Dyrness 1973). Some of these sites, particularly those in the Dry US PVG, have high value for traditional and tribal use. Camas (*Camassia* spp.), bitterroot (*Lewisia rediviva* Pursh), cous biscuitroot (*Lomatium cous* [S. Watson] J.M. Coult. & Rose), yampah (*Perideridia* spp.), and other American Indian first-food plants are often associated with scabland environments assigned to the Dry US PVG.

Potential future changes—

Soil moisture and winter temperature are the major limiting factors influencing vegetation composition and productivity in Pacific Northwest steppe communities; precipitation, temperature, soil texture, and soil depth are the primary abiotic determinants of soil moisture (Bates et al. 2006, Comstock and Ehleringer 1992, Schlaepfer et al. 2012). Winter snow and rain are important for recharging water storage in deep soil layers (Schlaepfer et al. 2012, Schwinning et al. 2003), and most precipitation in the Blue Mountains occurs during the winter and spring. However, high temperatures and low summer precipitation combine to produce extended periods of soil moisture deficits each summer. Although summers are warm and dry, winters can be quite cold throughout much of the steppe region in the Pacific Northwest (Comstock and Ehleringer 1992).

Shrubs in this PVG are generally well-adapted to both cold winter temperatures and summer drought, particularly the Dry US PVG. Sagebrush is tolerant of summer drought and is unresponsive to shifts in the seasonality of precipitation in regard to cover and density (Bates et al. 2006). Some shrubs are able to tolerate drought and remain photosynthetically active during periods of water and heat stress (Depuit and Caldwell 1975) or avoid severe drought stress by developing deep root systems that allow them to access deep soil water reserves throughout the summer (Franklin and Dyrness 1973). Schlaepfer et al. (2014) recently synthesized knowledge about natural big sagebrush regeneration. They noted that increases in temperature may not have a large direct influence on regeneration owing to the broad temperature optimum for regeneration. However, indirect effects could include selection for populations with less stringent seed dormancy. Drier conditions may have negative effects on germination and seedling survival and could also lead to lighter seeds, which lowers germination success further.

Moist US may be particularly vulnerable to future climate shifts, particularly at the lower ectone. Upland shrublands at low elevations may decrease in productivity in response to prolonged summer drought. Increased winter precipitation may allow soil recharge, but it is unclear if the capacity of these systems is adequate to avoid high stress and mortality in the summer. Paleoecological evidence suggests an increase in sagebrush with warming. Data from the Blue Mountains indicate that the shrub-steppe boundary was higher in elevation in the past during the warmer and drier early to mid-Holocene (fig. 6.3).

Species distribution models project nearly complete loss of habitat for big sagebrush, curl-leaf mountain-mahogany, and antelope bitterbrush (table 6.2). Models by others suggest that sagebrush distribution may decrease, with strong decreases in the southern part of the range and increases in the northern parts and at higher elevations (Schlaepfer et al. 2012). Simulation results for big sagebrush from a process model (under 2070–2099 CMIP5 climate scenarios) support expectations of increased probability of regeneration at the leading edge of the current big sagebrush range and decreased probability at the trailing edge compared to current levels. MC2 projects that although temperate (C3) shrublands will decline, xeromorphic (C4) shrublands (shrublands with an understory C4 grass component) will increase markedly. Currently, C4 grasses occur only in Hells Canyon National Recreation Area, typically within grassland communities. It is unclear if C4 grasses will actually increase in this region without adequate summer precipitation, but it is likely that more drought-tolerant species like sagebrush will increase in dominance within this PVG at the expense of species adapted to moister and cooler conditions. Thus, model output suggests that some shrubland species may decline, but that shrublands overall may increase markedly across the landscape.

Disturbance—

In productive mountain big sagebrush plant associations, such as those that include Idaho fescue, mean fire-return intervals ranged from 10 to 25 years, with large fires every 38 years (Miller et al. 2005). However, fire was much less frequent in the more arid plant associations such as Wyoming big sagebrush/Thurber's needlegrass (*Achnatherum thurberianum* [Piper] Barkworth) (50 to 70 years) and low sagebrush/Sandberg bluegrass (Miller and Rose 1999, Young and Evans 1981), where fire-free periods of 90 (Young and Evans 1981) and 138 years (Miller and Rose 1999) were reported in northern California and south-central Oregon, and fire-free periods probably exceeded 150 years for some sites. Baker (2006) argues that historical fire rotations were 70 to 200 years in mountain big sagebrush and longer in other types.

Long-term charcoal records suggest that fire regimes in these vegetation types are climate and fuel driven; sagebrush densities and fire frequencies increased during wet periods (decades to centuries) and declined during dry periods (Mensing et al. 2006).

Multiple shrub species are characteristic of these PVGs. Big sagebrush, antelope bitterbrush, and curl-leaf mountain-mahogany are fire sensitive and can be temporarily eliminated from a site by burning. Recovery of shrub canopy cover to predisturbance levels can require 10 to 50 years or more, with recruitment of new shrubs from soil seed banks being an important factor controlling recovery time (Ziegenhagen and Miller 2009). Short fire-return intervals can cause significant changes in species and productivity if shrub communities have not fully recovered between disturbances (Davies et al. 2012).

Shrublands, particularly Dry USs, are also prone to invasion by nonnative annual grasses (box 6.6). In some areas, introductions of invasive plant species such as cheatgrass and medusahead (*Taeniatherum caput-medusae* [L.] Nevski) have significantly altered fire regimes by producing sufficient fine fuels to carry wildfires (D'Antonio and Vitousek 1992). Other important disturbances and stressors in these systems include juniper expansion, livestock grazing, and land conversion owing to agriculture or development.

Synthesis—

Paleoecological data suggest sagebrush generally increased during warm periods in the past. However, in the early to mid-Holocene (warmer, drier) the shrub-steppe boundary was located higher in elevation, and results regarding shifts in sagebrush distribution and regeneration support this expectation. Less is known about other shrub species. However, species distribution models indicate highly reduced available climate habitat for many shrubs, but output from MC2 projects that xeromorphic shrublands will increase significantly. It is likely that increased warming would result in increased coverage of ecosystems better adapted to arid conditions ecosystems, such as shrublands. However, as wildfires and warmer conditions increase, there is a risk of conversion to nonnative annual grasslands in these systems, particularly for the drier PAGs.

Moist and Dry Upland Herbland

Moist UH (warm very moist, warm moist, hot moist, very hot moist) and Dry UH (hot dry and warm dry) are discussed together for this assessment. Dry UH is a more common PVG than Moist UH. Grasslands encompass numerous grass-dominated vegetation community types that differ according to the dominant grasses

Box 6.6—Nonnative annual grasses

Nonnative annual grasses such as cheatgrass, medusahead, and North Africa grass (*Ventenata dubia* [Leers] Coss.) alter fire regimes and are some of the most important ecosystem-altering species on the planet (Brooks et al. 2004). Cheatgrass is widely distributed in western North America (USDA and NRCS 2013) and is abundant and dominant in steppe communities (Mack 1981). Following disturbance, this species is capable of invading low-elevation forests (Keeley et al. 2003, Keeley and McGinnis 2007, Kerns et al. 2006) and creating surface fuel continuity between arid lowlands and forested uplands (fig. 6.21). Highly competitive traits enhance its ability to exploit soil resources after fire and to increase its status in the community (Melgoza and Nowak 1991, Melgoza et al. 1990), and dominance may be persistent (Zouhar 2003). After establishment, cheatgrass tends to increase the probability of subsequent wildfire occurrence (D'Antonio and Vitousek 1992) because the fine, continuous, and highly combustible fuels dry early in the season, increasing the length of the fire season in some ecosystems. Aside from the potential threat to biodiversity from this type of change, conversion of forests and woodlands to grasslands has important implications for carbon cycling and feedbacks between climate and the biosphere, because forests sequester large amounts of carbon (Bonan 2008).

A species distribution model that assumed lower (summer) precipitation in the future projected expansion of cheatgrass in a warmer climate (Bradley et al. 2009). However, when higher precipitation is included in the model, the area of reduced habitat for cheatgrass was reduced as much as 70 percent. Increases in productivity in response to elevated carbon dioxide has been documented, and elevated atmospheric carbon dioxide may have already contributed to cheatgrass productivity and fuel loading and associated effects on fire (Ziska et al. 2005). Because of the fire and

Becky Kerns

Figure 6.21—Cheatgrass invasion in a ponderosa pine stand after a prescribed fire that caused substantial overstory mortality (Malheur National Forest).

Box 6.6 (continued)

invasive grass cycle, changes in future fire regimes are important climate considerations for nonnative annual grasses. More area burned, more frequent large wildfires, larger extent of high-severity fire, longer wildfire durations, and longer wildfire seasons are expected in the future (Lutz et al. 2009, Miller et al. 2008, Nydick and Sydoriak 2014, Westerling et al. 2006), thus increasing the invasion risk of nonnative annual grass species.

The likelihood of forest, woodlands and shrublands being invaded by nonnative annual grasses in a warmer climate will increase because of more disturbance, effects of warming on species distributions, enhanced competitiveness of nonnative plants from elevated carbon dioxide, and increased stress to native species (Breshears et al. 2005, Dukes and Mooney 1999, Pauchard et al. 2009, Ziska and Dukes 2011) (fig. 6.8). Warming alone may increase the risk of nonnatives, because many invasive species have range limits set by cold temperatures, which has tended to limit their establishment in forests, particularly the higher elevation and continental western forests. Of particular concern in the Blue Mountains is the recent increase in North Africa grass and its apparent ability to outcompete and replace areas dominated by cheatgrass. The effectiveness of management actions to control invasive plants is decreasing in some areas as a result of reduced herbicide efficacy (Archambault et al. 2001, Ziska and Teasdale 2000), and biocontrol methods may not be as effective in a warmer climate (Hellmann et al. 2008).

and whose distribution largely reflects underlying gradients in annual precipitation and soil properties (Franklin and Dyrness 1973). The climate is classified as arid to semiarid with low precipitation, hot dry summers, and relatively cold winters. Most precipitation occurs as rain and snow in winter and spring. Moist upland grasslands are characterized largely by Idaho fescue and bluebunch wheatgrass (fig. 6.22). The absence of sagebrush suggests an improved moisture condition (Franklin and Dyrness 1973). Dry UHs are dominated by bluebunch wheatgrass and Sandberg's bluegrass.

Grasses can avoid summer drought stress by concentrating growth in the spring and early summer.

Potential future changes—

Soil moisture and winter temperatures are the major environmental limiting factors influencing grassland composition, with precipitation, temperature, soil texture, and soil depth being the primary abiotic determinants of soil moisture (Bates et al. 2006, Comstock and Ehleringer 1992, Schlaepfer et al. 2012). Grassland vegetation is generally well-adapted to both cold winter temperatures and summer drought. Grasses can avoid summer drought stress by concentrating growth in the spring and early summer, when soil water is still available and cooler temperatures promote high water-use efficiency (Comstock and Ehleringer 1992). Several studies also show that grasslands may be resistant to climate change effects (Dukes et al. 2005,

Mark Darrach

Figure 6.22—An example of a Dry Upland Herbland dominated by Idaho fescue, Umatilla National Forest. This site is considered to be in good condition.

Grime et al. 2008). Short-term changes in the interannual precipitation regime may not result in large changes in semiarid vegetation communities (Jankju 2008).

In a warmer climate, grasslands at lower elevation may shift in dominance towards more drought-tolerant species (e.g., less Idaho fescue) (Blinnikov et al. 2002) (fig. 6.5). There is essentially no model output available for individual grassland species, although Reeves et al. (2014) project that net primary productivity will increase in eastern Oregon grasslands in a warmer climate. MC2 projects that C3 grasslands will decline significantly, but that warm season (C4) grasslands will expand. However, the expansion of C4 grasses in the model is based on changes in temperature alone, and it is unlikely that C4 grasses will increase markedly in the future unless summer precipitation increases as well. Most GCMs project increased aridity in the summer. A potential shift from C3 to C4 grassland species is uncertain, although a shift in species composition to drought-tolerant species might be expected.

Disturbance—

Grasslands have been greatly affected by human activities, including livestock grazing, introduction of nonnative species, and agriculture (Humphrey 1943, Tisdale 1961). Grazing first became a major factor in Pacific Northwest steppe com-

munities with the introduction of cattle grazing in 1834 and sheep grazing in 1860. Settlers also introduced numerous nonnative grasses, including cheatgrass, which was well-adapted to climate within parts of the steppe region, and heavy grazing allowed nonnative grasses to invade native communities where they became highly persistent (Mack 1981). In addition to grazing, many grasslands have been cultivated for dryland agricultural crops like winter wheat or irrigated to produce summer fruits, vegetables, and grains. Homesteading on what is now public land was common in many tributaries of the Snake River and in Hells Canyon.

Wildfire occurrence is generally limited by lack of ignitions during the fire season or by lack of continuous fuels in particular on scabland communities on shallow soils. Cold season bunchgrass communities often have continuous fuels to carry fire. The extensive grasslands in Hells Canyon National Recreation Area have experienced numerous large grassfires, and much of the area has burned at least once since 1980. Many of the dominant bunchgrasses recover well from fires by resprouting from belowground organs and can achieve prefire abundance within 5 years. Fire-return intervals of less than 5 years have been documented only in isolated instances. However, in some areas, introductions of invasive plant species such as cheatgrass, medusahead, and North Africa grass have altered fire regimes by producing sufficient fine fuels to carry wildfires at higher frequencies than tolerated by perennial native grasses (D'Antonio and Vitousek 1992) (box 6.4).

Synthesis—

Grasslands may shift in dominance towards more drought-tolerant species, particularly at lower elevations and on more arid sites. Nonnative annual grasses may also increase in importance. In general, it is likely that with increased warming and fire occurrence, grasslands will become a bigger component of the landscape, particularly where shrublands and woodlands are no longer able to support woody species

Conclusions

Climate change is expected to alter vegetation structure and composition, terrestrial ecosystem processes, and the delivery of ecosystem services in the Blue Mountains. Climate influences the spatial distribution of major vegetation biomes, abundance of species and communities within biomes, biotic interactions, and geographic ranges of individual species. Climate also influences disturbance processes that shape vegetation structure and composition, which are often the catalysts for vegetation change. However, there is considerable uncertainty in what the actual effects on vegetation owing to climate change could be. Waring et al. (2011) report

that the Pacific Northwest has seen a significant decrease in the competitiveness of over 50 percent of the evergreen species in six ecoregions, with the highest concentration towards the northern and southern limits of their analysis area. However, these authors assigned the Blue Mountains a relatively low vulnerability score.

In a warmer, drier climate (especially in summer), the following may occur in the Blue Mountains by the end of the 21st century, although we note that there is considerable uncertainty about the future:

- The importance of pine and sagebrush species may increase.
- The forest-steppe ecotone may move north of its present position or up in elevation.
- Ponderosa pine may be found at higher elevations.
- Subalpine and alpine systems are potentially vulnerable, and subalpine tree species may be replaced by high-elevation grasslands, pine, or Douglas-fir.
- Juniper woodlands, which have been increasing in recent decades, may be reduced if longer and drier summers lead to more wildfire.
- Grasslands and shrublands at lower elevations may increase across the landscape but shift in dominance towards more drought-tolerant species.
- Nonnative species, including annual grasses, may increase in abundance and extent.

In general, species with life histories tolerant of frequent disturbance and highly altered environments will be more dominant because they can establish and persist in rapidly changing environments.

Tree growth in energy-limited portions of the landscape (high elevations, north aspects) may increase as the climate warms and snowpack decreases, whereas tree growth in water-limited portions of the landscape (low elevations, south aspects) will probably decrease. Some species may respond positively to higher concentrations of ambient carbon dioxide as a result of increased water-use efficiency, although this "fertilization" effect may diminish as other factors become limiting.

Ecological disturbance (e.g., fire, insect, and disease outbreaks), which is expected to increase in a warmer climate, will be extremely important in affecting species distribution, tree age, and forest structure, facilitating transitions to new combinations of species and vegetation patterns. Mountain pine beetle may be particularly important in lodgepole and ponderosa pine forests, and western spruce budworm and Douglas-fir tussock moth may also increase periodically. Annual area burned by wildfire is expected to increase substantially, and fire seasons will likely lengthen. In dry forest types where fire has not occurred for several decades, crown fires may result in high tree mortality. In addition, the interaction of multiple

disturbances and stressors will create or exacerbate stress complexes. For example, an extended warm and dry period may increase bark beetle activity, which would increase short-term fine fuels.

Considerable uncertainty exists about how climate change will affect species distribution, forest productivity, and ecological disturbance in the Blue Mountains. Simulation models provide science-based projections of how a warmer climate could modify the growth environment of species and broad patterns of ecological disturbance, supplemented by studies of the paleoecology of the region. However, because the future climate may differ considerably from what has been observed in the past, it is difficult to project vegetative response accurately for specific locations and time periods.

Despite the uncertainty associated with climate change, there are many proactive steps that land managers can take.

Adapting Vegetation Management to Climate Change in the Blue Mountains

Despite the uncertainty associated with climate change, there are many proactive steps that land managers can take that are likely to increase ecosystem resilience to climate change (e.g., Halofsky et al. 2011, Raymond et al. 2015). Based on the vulnerability assessment information presented in this chapter, and on documented adaptation principles (e.g., Millar et al. 2007, Peterson et al. 2011, Swanston and Janowiak 2012), workshop participants identified strategies, or general approaches, for adapting vegetation management to climate change (table 6.8). Participants also identified more specific on-the-ground tactics, or actions, associated with each adaptation strategy and considered the implementation of those tactics, specifically the timeframe for implementation, opportunities and barriers to implementation, and information needs (table 6.8). Adaptation strategies and tactics were focused on addressing potential increases in fire and invasive species (table 6.8 a, b, c), insect outbreaks (table 6.8 d), increases in temperatures and droughts (table 6.8 e, f), and effects of increasing temperatures on alpine and subalpine plant communities (table 6.8 g, h). These adaptation strategies and tactics are summarized below.

While the strategies and tactics described here do not include all potentially appropriate vegetation-related actions to take under a changing climate, they do include those that workshop participants thought were most important. However, not all of these strategies and tactics are appropriate in all places and in all situations, and they should be evaluated by managers on a case-by-case basis. Many of these ideas will require additional thought and analysis before they can be implemented.

Table 6.8a—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Timeframe	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Larger and more severe fires in forest ecosystems.				
Adaptation strategy: Manage forest vegetation to reduce severity and patch size; protect refugia (e.g., old trees).				
Map fire refugia	<10 years, opportunistic	Coordination with project planning	Lack of knowledge on current location and conditions of fire refugia	Compilation of locations of fire refugia with a consistent protocol
Use gaps and other methods in silvicultural prescriptions to reduce fuel continuity	<10 years, opportunistic	Coordination with project planning		
Identify processes and conditions that create fire refugia	<10 years, opportunistic	Coordination with project planning	Lack of knowledge on current location and conditions of fire refugia	Compilation of locations of fire refugia with a consistent protocol
Adaptation strategy: Manage forest landscapes to encourage fire to play a natural role				
Implement fuel breaks at strategic locations	<10 years	Coordination with roadside treatments Blue Mountains restoration strategy	Limited capacity for National Environmental Policy Act analysis	Maps of lightning- and human-caused ignitions
Identify areas where wildland fire use may help to meet management objectives	<10 years	Use and interpretation of the Wildland Fire Decision Support System	Risk aversion by fire managers	Incorporation of additional information in the Wildland Fire Decision Support System, especially on resources that benefit from fire
Implement strategic density management through forest thinning	<10 years			
Incorporate climate change in Wildland Fire Decision Support System	<10 years to >30 years			Public education on the benefits of fuel reduction and prescribed fire

Table 6.8b—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Timeframe	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Higher wildfire frequency will cause increased mortality of shrub species and increased dominance of invasive species.				
Adaptation strategy: Increase resilience of native sagebrush-grass ecosystems.				
Promote the occurrence and growth of early-season native species	<10 years	Collaboration with range permitees		Effects of a warmer climate on plant phenology
Reduce grazing in July and August to encourage perennial growth	10 to 30 years	Collaboration with range permitees	Opposition by some range permitees; economic issues	
Revise grazing policies, and review and evaluate grazing allotment plans	>30 years	Possible in some political environments	Entrenched policies for grazing; economic issues; effects on lifestyle	
Adaptation strategy: Maintain vigorous growth of native shrub, perennial grass, and other perennial species, while minimizing the spread of invasive species				
Remove encroaching conifers	<10 years	Collaboration with range permitees; coordination with fuel reduction projects; coordination with sagebrush restoration projects	Funding issues; potential challenges with invasive species	Synthesis of information on effects of conifer removal; prioritization of treatment areas
Plant seed of native species	<10 years	Postfire seeding; coordination with fuel reduction projects; coordination with sagebrush restoration projects	Low success of seeding	Improved acquisition of native seed
Monitor successional patterns of vegetative communities	<10 years	New LIDAR imaging	Cost of long-term monitoring; difficulty of on-the-ground location accuracy	Framework for timely and efficient monitoring
Adaptation strategy: Maintain reproducing populations of curl-leaf mountain-mahogany, so it can expand as needed				
Strategically protect mountain-mahogany populations to allow for natural reseeding after fire	<10 years to 30 years	After wildfire; during planning for grazing allotment management	Lack of resources; opposition by grazing permitees	Map of existing mountain-mahogany populations

Table 6.8c—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Timeframe	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Higher temperatures and increased fire frequency and intensity will reduce the dominance of native grasses and increase dominance of nonnative annual species.				
Adaptation strategy: Increase the resilience of native perennial grasses and other nontree vegetation.				
Apply prescribed burning in the spring	Ongoing	Integration in existing fuels reduction programs	Potential increase in invasive species	Information on range condition and high-priority areas for treatment
Adaptation strategy: Manage grazing by livestock and ungulates to reduce impacts on perennial grasses.				
Focus grazing on nonnative species in spring; do not graze natives in summer	<10 years to 30 years	Acceptance will be slow; collaboration with permitees to modify allotment plans	Opposition by permitees	Information on range condition and resilient locations
Find locations where late-season grazing has minimal impacts	<10 years	Acceptance will be slow; collaboration with permitees to modify allotment plans	Opposition by permitees	Information on range condition and resilient locations
Adaptation strategy: Manage fire to avoid increase in nonnative annual species.				
Apply prescribed burning in the spring	<10 years	Integration in existing fuels reduction programs	Potential increase in invasive species; small period of time for implementation	Identification of wildland-urban interface; seasonal fire forecasts
Adaptation strategy: Manage soil conditions to avoid increased runoff after wildfire.				
Maximize native vegetative ground cover	<10 years		Variability of local site conditions	Soil mapping; erosion modeling
Adaptation strategy: Determine potential resilience of different locations, and actively restore less resilient sites.				
Increase resilience of native species where intact or productive communities exist	<10 years	Monitoring of existing vegetative communities		Monitoring of areas that need protection
Decrease resilience of existing nonnative species with appropriate management practices	10 years to >30 years	Spring grazing		Improved communication with public
Identify and promote early successional natives that may be able to compete with nonnatives	10 years to >30 years	Coordination with other projects that have identified suitable early successional native species and reduction in nonnatives		Improved local knowledge and testing of native cultivars

Table 6.8d—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Timeframe	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Uncharacteristic size and severity of ecological disturbances in forest ecosystems: insects				
Adaptation strategy: Recognize natural role of insect disturbances, and identify areas at high risk.				
Allow for natural mortality within the historical range of variability for specific insects	<10 years, depending on insect cycle	National forest land management plan revision	Public opposition	Identify scale at which mortality is acceptable, relative to current objectives
In dry forest, restore low-severity fire to lower stand density and increase resilience to bark beetle outbreak	10 to >30 years	Coordination with project planning	Inadequate resources to conduct restoration at large spatial scales	
Adaptation strategy: In forest types where the risk of insect outbreaks is high, promote diversity of forest age and size classes.				
Diversify large contiguous areas of single-age and single-size classes		Coordination with project planning	Inadequate resources; public opposition; internal opposition	Historical/desired conditions for landscape pattern and patch size

Table 6.8e—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Timeframe	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: A warmer climate with more droughts will reduce growth in most forests and make regeneration more difficult for some species.				
Adaptation strategy: Protect genotypic and phenotypic diversity.				
Protect trees that exhibit adaptation to water stress, (e.g., trees with low leaf area/sapwood); collect seed for future regeneration	<10 years	Coordination with project planning	Identification of trees with low leaf/sapwood area by marking crews; Identification of successful adaptation by trees to stress	Better understanding of the phenotypic adaptive capacity of different tree species
Maintain variability in species and in tree architecture	10 to 30 years	Coordination with project planning		
Adaptation strategy: Maintain and enhance forest productivity regardless of tree species; focus on functional ecosystems and processes.				
Manage tree densities to maintain tree vigor and growth potential	<10 to >30 years	Coordination with project planning		
Prepare for species migration by managing for multiple species across large landscapes	10 to 30 years	Coordination with project planning		
Maintain soil productivity through appropriate silvicultural practices	>30 years	Coordination with project planning		
Adaptation strategy: Use judicious managed relocation of genotypes where appropriate.				
Push boundaries of seed zones and plant genotypes from warmer locations; use a variety of genotypes rather than just one	10 to 30 years, opportunistic	Coordination with forest regeneration projects		
Adaptation strategy: Use tree improvement programs to ensure availability of drought-tolerant tree species and genotypes.				
Develop seed orchards that contain a broader range of tree species and genotypes than in the past	10 to >30 years	Good foundation in existing seed orchards	Time required to develop new capacity	Data on tree genetics across broad landscapes to guide development of nursery stock and regeneration strategies

Table 6.8f—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Timeframe	Opportunities for implementation	Barriers .to implementation	Information needs
Sensitivity to climatic variability and change: Higher temperatures and increasing drought will stress some species in moist mixed-conifer forests, especially western larch.				
Adaptation strategy: Maintain vigorous existing western larch and encourage its regeneration.				
Create gaps in forests to reduce competition and increase larch vigor	<10 years, opportunistic	Tree mortality of other species, which creates gaps for larch regeneration (after wildfire and insect outbreaks)	Public opposition to large forest openings	Data on current distribution of larch; studies on gap sizes needed for regeneration
Regenerate larch with appropriate site preparation (e.g., prescribed burning, followed by planting)	<10 years, opportunistic	Following timber harvest, wild-fire, and insect outbreaks		

Table 6.8g—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Timeframe	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Higher temperatures may increase stress for some species in cold upland and subalpine forests.				
Adaptation strategy: Protect rare and disjunct tree species (Alaska cedar, limber pine, mountain hemlock, and whitebark pine).				
Plant and encourage regeneration of rare and disjunct species in appropriate locations	Opportunistic	Following wildfire		
Plant whitebark pine genotypes that are resistant to white pine blister rust	<10 years, opportunistic		Limited availability of rust-resistant nursery stock	

Table 6.8h—Adaptation responses for vegetation to climate change in the Blue Mountains

Adaptation tactic	Timeframe	Opportunities for implementation	Barriers to implementation	Information needs
Sensitivity to climatic variability and change: Higher temperatures may increase stress for some alpine plant species (including rare plants), especially in the Wallowa Mountains.				
Adaptation strategy: Improve our understanding of the effects of climatic variability and change on alpine plant species.				
Install GLORIA plots to monitor species distribution and abundance	<10 years			
Collaborate with other federal agencies to monitor alpine species	<10 years to 30 years			

GLORIA = Global Observation Research Initiative in Alpine Environments.

Responding to Increased Fire and Invasive Species Establishment

Increased temperatures with climate change will likely lead to increased wildfire area burned (Littell et al. 2010, McKenzie et al. 2004, Westerling et al. 2006). With increasing fire in forested ecosystems, managing vegetation to reduce fire severity and decrease fire patch size could help to protect fire refugia and maintain old trees (Peterson et al. 2011). For example, incorporating openings in silvicultural prescriptions decreases forest density and fuel continuity, which may reduce wildfire severity and protect old trees (Churchill et al. 2013, Stine et al. 2014) (table 6.8a). Management practices that help fire to play a more natural role in ecosystems, such as density management, prescribed fire, and wildland fire use, may also increase ecosystem resilience to wildfire under a changing climate (Peterson et al. 2011, Stephens et al. 2010, Stine et al. 2014) (table 6.8a).

In shrubland and grassland systems, increased area burned will likely lead to increased mortality of shrub species and native grasses, and increased abundance of nonnative species, annual grasses in particular (Creutzburg et al. 2014). Adaptation strategies and tactics to address these sensitivities include increasing the resilience of native ecosystems through grazing management (i.e., avoid grazing practices that promote invasive species establishment), active restoration of less resilient sites (e.g., plant natives on sites dominated by invasive species), and management of soil resources to maintain stability and productivity (e.g., establish native vegetation to stabilize eroded areas).

Increasing tree species diversity may also help to improve resilience to insect outbreaks, particularly in low-diversity stands.

Responding to Increased Insect Outbreaks

Native insect species have long played a role in Blue Mountain ecosystem dynamics (Oliver et al. 1994), and it will be important to recognize this role and accept that there will be insect-caused tree mortality under changing climate (table 6.8d). However, there are some management actions that may increase ecosystem resilience to native insect outbreaks, such as mountain pine beetle outbreaks. For example, restoring historical fire regimes in dry forests, and increasing diversity of forest structure and age and size classes may help to minimize the impacts of insect outbreaks (Churchill et al. 2013). Increasing tree species diversity may also help to improve resilience to insect outbreaks (Dymond et al. 2014), particularly in low-diversity stands (e.g., stands where ponderosa pine and western larch were removed and grand fir dominates).

Responding to Increasing Temperatures and Droughts

Increasing temperatures will likely decrease productivity in water-limited forests and inhibit regeneration of some species (Littell et al. 2010). Protecting trees that exhibit adaptation to water stress (e.g., trees with low leaf-area-to-sapwood ratios) and collecting seed from these individuals for future regeneration could help to increase resilience to water stress (table 6.8e). To ensure success of future revegetation efforts, seed-use plans should reflect seed needs based on projected climate change and disturbance trends. Managers may want to ensure that seed orchards contain tree species and genotypes that are well-adapted to drought and disturbance, and for some species, resistant to disease (e.g., white pine blister rust). Managers may also want to push the limits of seed zone boundaries and include seed from lower elevations in plantings.

With changing climate, it will be important to promote forest productivity and ecosystem function (table 6.8e). In some cases, managers may want to protect certain species that will be susceptible to increased drought stress, such as western larch in moist mixed-conifer forests (table 6.8f). Silvicultural practices that maintain densities to maximize tree growth and vigor and that protect soil productivity will likely help to maintain ecosystem function.

Responding to Effects of Increasing Temperatures on Alpine and Subalpine Plant Communities

Higher temperatures are likely to increase water stress for some species in cold upland and subalpine plant communities. Rare and disjunct populations (e.g., of Alaska cedar, limber pine, whitebark pine, and mountain hemlock) may require protection to ensure their continued survival under changing climate (table 6.8g). Planting in appropriate locations could help to prevent loss of these populations. For whitebark pine, planting of genotypes that are resistant to white pine blister rust will be critical. For alpine plant species, such as those in the Wallowa Mountains, monitoring will be necessary to improve understanding of how climate variability and change will affect them (table 6.8h).

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