5. The Potential Effects of Climate Change on Oregon’s Vegetation

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Summary and Knowledge Gaps

There is clear evidence from the paleoenvironmental record that Oregon’s vegetation has responded to climate changes over past millennia. Recent research indicates that plants have also responded to climate changes over the last century, with relatively large and more rapid changes occurring since the mid-1970s. Twentieth-century vegetation changes in Oregon that may be the result of changing climate include earlier flowering dates for some plant species and increased tree mortality in some old-growth forest stands. Evidence from studies in the California Sierra Mountains and elsewhere indicates that some tree species are shifting their distributions significantly upward in elevation, and this response would be expected to occur in mountainous regions of Oregon as well.

Some model simulations of future vegetation changes in Oregon indicate that high elevation areas of subalpine forest and alpine tundra as well as areas of shrubland in eastern Oregon will contract under projected future climate changes. These projected vegetation changes would reduce critical habitat for species of management concern, such as greater sage-grouse (*Centrocercus urophasianus*). The paleoenvironmental record indicates that plant species respond individualistically to changing climate. As species distributions change, the current associations of plant species in Oregon may be affected. Some model simulations indicate that the species composition of western Oregon forests may already be changing and that the rate of change will increase during the 21st-century.

Disturbance events will continue to play a critical role in the dynamics of Oregon’s vegetation. Recent forest dieback in the western United States, empirical studies, and model simulations indicate that the frequency and magnitude of some disturbance events, such as drought, wildfire, and insect outbreaks, may be changing. Model simulations indicate that more frequent, large fires could become increasingly common in western Oregon forests.

Oregon’s forests currently store substantial quantities of carbon and could store more under forest management practices that increase the time between harvests and/or reduce the amount of carbon that is harvested from forests. Carbon is also stored in wood products that are harvested from Oregon’s forests, but wood products are unlikely to provide for substantial increases in stored carbon under current manufacturing, use, and disposal practices. The more

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carbon that is stored in Oregon’s forest, the greater the potential carbon releases that may occur in the future in response to disturbances, such as fire. Management activities carried out in response to climate changes, such as thinning of forests to reduce moisture stress on the remaining trees, would also remove carbon from the forests.

Plant population genetics and life-cycle characteristics will play an important role in determining how plants respond to climate changes. Future climate changes are projected to occur relatively rapidly, making it difficult for plants to genetically adapt to climate changes or to disperse to areas of more suitable climate, although microclimate diversity in Oregon’s mountainous terrain may provide suitable climate conditions for some species over relatively short distances. Adaptive management strategies may assist plants in adapting to future climate changes, but will be challenged by the long life-cycles of many Oregon forest tree species.

There are a number of important knowledge gaps in our understanding of how plants may respond to climate change. Some species in Oregon, particularly important commercial species such as Douglas-fir (Pseudotsuga menziesii), have been well studied. However, we know relatively little about many plant species, their sensitivities to climate and climate change, and their interactions with other species, including insects and diseases. The need for additional information is particularly important with regard to invasive species in Oregon, including how invasive species will interact with native species and the ability of invasive species to alter disturbance regimes, such as fire. The potential for CO2-induced increases in plant water use efficiency are not well known, but could be critical for ameliorating plant drought-stress for some species. The roles of nutrient limitations in carbon sequestration and responses to disturbances also remain critical uncertainties.

Uncertainties in projections of future climate changes, particularly our ability to project changes in the amount and variability of precipitation, limit our ability to project how plant species will respond to climate change. The potential for future climate changes to produce spatially extensive, multi-year droughts is of particular concern. As projections of future climate change improve in quality and resolution, we will be better able to simulate the potential response of plant species to changes in climate. Similarly, improvements in vegetation and ecosystem models, including modeling efforts that integrate data from vegetation monitoring and vegetation spatial distribution modeling, as well as an explicit consideration of disturbances, will also improve vegetation simulations. The potential for rapid climate and ecosystem changes and associated uncertainties may require new adaptive management approaches.
5.1 Introduction

Oregon’s vegetation is remarkably diverse, ranging from the old-growth temperate rainforests west of the Cascade Range to the arid shrublands and grasslands of eastern Oregon. Vegetation directly contributes to the environmental, economic, and cultural well-being of the state. Plants provide habitat for wildlife, including threatened and endangered species such as the marbled murrelet (Brachyramphus marmoratus). Oregon’s forests support the state’s logging and wood products industries, its rangelands support ranching communities and livestock industries, and vegetation of all types contributes to the recreation and tourism activities in the state. Oregon’s forest and rangeland areas are the source of over 80% of the state’s water (Brown et al., 2008). The region’s vegetation is also culturally important. It provides many of the first foods (Ettwaii-lish, 2005) and sacred places used by local tribes, and is central to many of the iconic landscapes reflected in Oregon’s art and literature.

Potential future climate changes will affect vegetation across Oregon. As climate changes, the species composition, spatial distribution, and productivity of Oregon’s vegetation may change. In this chapter we provide a brief overview of some of the potential effects of future climate change on Oregon’s terrestrial vegetation. Additional information may be found in the references cited throughout the text as well as in the numerous national and regional climate change assessments that encompass Oregon. These resources include the Intergovernmental Panel on Climate Change’s Assessment Reports and Special Reports (e.g., IPCC, 2007), three Assessment Reports on climate change impacts for the U.S. (e.g., Karl et al., 2009), and a Synthesis Report on “Forests, Carbon, and Climate Change” produced as a joint effort of the Oregon Forest Resources Institute, Oregon State University, and the Oregon Department of Forestry (Cloughesy, 2006).

5.2 Vegetation Responses to Climate Change

5.2.1 Oregon’s current vegetation

Oregon’s vegetation is strongly influenced by climate and topography. Along the Oregon coast, the Pacific Ocean affects local climate conditions that support maritime species, such as Sitka spruce (Picea sitchensis) and shore pine (Pinus contorta var. contorta). Further inland, vegetation varies along the west slope of the Cascade Range from oak woodlands in the Willamette Valley to mixed conifer forests dominated by Douglas-fir (Pseudotsuga menziesii) at mid-elevations, to high elevation mountain hemlock (Tsuga mertensiana) and true fir (Abies spp.) stands. On the east side of the Cascade Range, ponderosa pine (Pinus ponderosa) and western juniper (Juniperus occidentalis) woodlands give way to steppe and shrubland vegetation in drier locations of eastern Oregon (Figures 5.1 and 5.2). These natural patterns in the distribution of vegetation are modified by human activities, including urbanization, agriculture, road building, logging, grazing, and fire suppression.
5.2.2 Past vegetation changes

Our understanding of how Oregon’s vegetation may respond to future climate changes is based, in part, on studies of how vegetation has responded to past climate changes. Many studies of paleovegetation changes have been carried out in Oregon, including reconstructions of vegetation patterns from pollen found in lake sediments (e.g., Worona and Whitlock, 1995), tree-ring analyses of the relationships between tree growth and climate (e.g., Pohl et al., 2002), and investigations of past fire occurrence using fire scars from trees (e.g., Weisberg and Swanson, 2003). These studies provide evidence of how vegetation has responded to past climate changes over time periods ranging from centuries to millennia (Whitlock and Bartlein, 1997). Worona and Whitlock (1995) analyzed pollen from the sediments of Little Lake in the Oregon Coast Range (near Blachly, Oregon), which provide a record of vegetation change going back ~42,000 years. This pollen record indicates that the current Douglas-fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), and western red cedar (*Thuja plicata*) forests of Oregon’s Coast Range developed in the last ~5600 years, possibly in response to cooling climate conditions during this time period. Charcoal from past fires found in the sediments of Little Lake (Long et al., 1998) and Taylor Lake (south of Warrenton, Oregon; Long and Whitlock, 2002)
indicate that fires in the region also became less frequent during this same time period, as would be expected under cooler climate conditions.

On shorter timescales, many research efforts are focused on identifying vegetation responses to recent climate changes that have occurred over the last century. Particularly important for understanding the potential effects of climate change on plants are changes in phenological events, such as the flowering dates for different plant species. Cayan et al. (2001) analyzed flowering data for honeysuckles (*Lonicera* spp.) and common purple lilacs (*Syringa vulgaris* f.
*purpurea* in the western U.S., including data from Oregon, and reported earlier first bloom dates in the 1980s and 1990s as compared with data from the 1960s and 1970s. They attributed these earlier bloom dates to increased spring temperatures across the region at the end of the 20th-century. In a more recent study, van Mantgem et al. (2009) identified an increase in tree mortality rates in the western U.S. since the 1950s that they attribute, in part, to increased drought stress on trees resulting from increased temperatures during this time.

Both the Cayan et al. (2001) and van Mantgem et al. (2009) studies used data from long-term vegetation monitoring efforts, which are critical for identifying vegetation changes over time in response to climate changes. A number of long-term vegetation monitoring efforts are under way in Oregon. These efforts include the U.S. Forest Service’s Forest Inventory and Analysis Program (http://fis.fs.fed.us/), the National Park Service’s Inventory and Monitoring Program (http://science.nature.nps.gov/im/index.cfm), and the U.S.A. National Phenology Network (http://www.usanpn.org/). Long-term vegetation data are also being collected at many individual research sites, such as the H.J. Andrews Experimental Forest near Blue River, Oregon (http://andrewsforest.oregonstate.edu/).

### Case Study 1. Mapping plant species distributions in Oregon

A recent effort to model and map existing species distributions across ownerships within Oregon, Washington, and parts of California is being conducted through the Landscape Ecology, Modeling, Mapping and Analysis (LEMMA) project team (U.S. Forest Service Pacific Northwest [PNW] Research Station and Oregon State University). The mapping is integrated with ongoing sample-based forest inventories conducted by the Forest Inventory and Analysis (FIA) program at the U.S. Forest Service PNW Research Station, the Current Vegetation Survey of Region 6 (Oregon and Washington), USDA Forest Service, and the U.S. Bureau of Land Management (BLM) in western Oregon. The project uses gradient imputation (Gradient Nearest Neighbor [GNN]; Ohmann and Gregory, 2002) to map detailed vegetation composition and structure for areas of forest and woodland. GNN uses multivariate gradient modeling to integrate data from FIA field plots with satellite imagery and mapped environmental data. A suite of environmental variables associated with each sampling plot is assigned to each pixel in a digital map based upon interpretations of satellite imagery and topographical information. From multivariate statistics using these variables, regional maps can be constructed for many of the same vegetation attributes available for FIA plots. The 2000-era species distribution maps produced with GNN (Figure 5.2) provide baseline information that will be compared with future inventory data to assess the rate and extent of future shifts in distributions of species.

### 5.3 Potential Future Vegetation Changes

Climate has long been identified as a primary control on the geographic distribution of plants (e.g., Forman, 1964; Box, 1981). Research from a variety of ecosystems and spatial scales describes the effects that climate has on plant species distributions and ecosystem type (Neilson and Wullstein, 1983; Davis and Botkin, 1985; Overpeck et al., 1990; Guisan and Zimmerman, 2000). The paleoenvironmental record provides clear evidence that species respond individualistically to climate change and supports the current scientific consensus that the
geographical distributions of plant species will change as climate changes (Huntley, 1991; Pitelka, 1997; Jackson and Overpeck, 2000; Bachelet et al., 2001; Hansen et al., 2001; Shafer et al., 2001; Walther et al., 2002; Higgins et al., 2003; Sans-Elorza et al., 2003; McLachlan et al., 2005; Neilson et al., 2005b; Wang et al., 2006; Jurasevski, 2007; McKenney et al., 2007; Xu et al., 2007; Thuiller et al., 2008). The ranges of many North American tree species in relatively flat terrain may have to expand at rates of 3-5 kilometers per year in order to adapt to the climate changes that are projected for this century (Davis and Shaw, 2001; Iverson and Prasad, 2002).

One approach for investigating how plants may respond to future climate changes is to use vegetation models to simulate potential future vegetation changes. These models range from simple conceptual models of how plants may respond to temperature and precipitation changes to more complex statistical and process-based numerical models. Statistical models include various correlation or regression-tree approaches to simulate species distributions or species mixtures as well as more complex statistical models that simulate stages of forest growth and succession (McKenney et al., 2007; Iverson and Prasad 2001; Weisz et al., 2009; Henderson 2008).

Although statistical models provide useful insights into vegetation responses to climate change, there are a number of important processes, such as many types of disturbance, which they often do not explicitly simulate. Furthermore, statistical models implicitly assume that observations of the past are also applicable under future conditions, which might not always be the case in a changing environment (see Williams and Jackson, 2007).

Mechanistic or process-based models are thus considered more robust models for simulating the complex and non-linear interactions of changes in temperature, humidity, precipitation and elevated atmospheric CO2 concentrations projected for the future (Rastetter et al., 2003; National Research Council, 2009). A number of process-based models have been run for Oregon, including 3-PG (Coops et al., 2005) and MC1 (Neilson et al., 2005a). Process-based models often incorporate the physiological effects of elevated atmospheric CO2 concentrations on plants, as well as the hydrological constraints on the distribution of vegetation density, which can be used to simulate the distributions of forests, savannas, shrublands and grasslands. Many of these process-based models also simulate the potential effects of disturbance on vegetation. It is particularly noteworthy that although these models differ in the various details of their construction, many produce qualitatively similar forecasts of the impacts of potential future climates on ecosystem distribution, function, and disturbances.

Figure 5.3 displays model output from MC1 (MAPSS-CENTURY, ver. 1, R. P. Neilson, U.S. Forest Service MAPSS Group; http://www.fs.fed.us/ccrc/video/skamania-scale.shtml), a dynamic general vegetation model that represents vegetation using plant functional types (e.g., needleleaf evergreen trees, shrubs, grass). Modern vegetation (Figure 5.3a) for Oregon was simulated by MC1 using PRISM historical monthly climate data for 1895-2002 (Daly et al., 2008). Future vegetation was simulated using climate data from two coupled atmosphere-ocean general circulation models (AOGCMs), CSIRO-Mk3.0 (Gordon et al., 2002) and UKMO-HadCM3 (Pope et al., 2000; UK Meteorological Office, Hadley Centre, 2006). These AOGCMs simulated future climates using the B1 and A2 future socio-economic scenarios (Nakicenovic et al. 2000) described in Chapter 1). The B1 scenario represents relatively low levels of greenhouse gas emissions and associated climate forcing for the end of the 21st-century while the A2
scenario represents relatively high levels of greenhouse gas emissions and associated climate forcing over the same time period.

Under the B1 emissions scenario, CSIRO-Mk3.0 simulates mean annual temperature increases of \(~1-2 \, ^\circ\text{C} (~2-4 \, ^\circ\text{F})\) and total annual precipitation increases of \(~0-10\%\) for the period 2070-2099 (30-year mean) as compared with 1961-1990 (30-year mean) data for Oregon. Under the A2 emissions scenario, UKMO-HadCM3 simulates mean annual temperature increases of \(~3-5 \, ^\circ\text{C} (~6-8 \, ^\circ\text{F})\) for the period 2070-2099 (30-year mean) as compared with 1961-1990 (30-year mean) data for Oregon. Total annual precipitation for 2070-2099 (30-year mean) is projected to decrease by \(~0-15\%\) in western and central Oregon and to increase by \(~3-6\%\) in easternmost Oregon.

In addition to responding to changes in temperature and precipitation, plants also may respond to changes in the concentration of atmospheric carbon dioxide (CO\(_2\)), which plants use for photosynthesis. Human activities have increased the concentrations of CO\(_2\) in the atmosphere and CO\(_2\) levels are projected to continue rising for the foreseeable future (IPCC, 2007). Particularly in areas where moisture for plants is limited, increased concentration of atmospheric CO\(_2\) may increase the efficiency with which plants are able to use water in the soil and may allow some plant species to expand their range in parts of the state, or to maintain their range under increasing drought stress (Morgan et al., 2004; Millar et al., 2006).

In future vegetation simulations produced by MC1, areas of subalpine forest and alpine tundra in Oregon are projected to decrease as temperatures increase at higher elevations (Figures 5.3b and 3c). Areas of shrubland in eastern Oregon are also projected to decrease, which could reduce sagebrush (\(\text{Artemisia}\) spp.) habitat (Neilson et al. 2005a). Sagebrush shrublands are considered important habitat for greater sage-grouse (\(\text{Centrocercus urophasianus}\)), a species of management concern in southeastern Oregon (Chapter 7). The future vegetation simulations project an expansion of forest and woodland into areas of eastern Oregon currently dominated by grassland and shrubland. This expansion occurs as the combined result of projected increases in precipitation, a longer growing season, and increased plant water-use efficiency produced by increased atmospheric CO\(_2\) concentrations. Areas of mixed evergreen and deciduous forest are projected to expand in the Oregon Coast Range. This vegetation type represents a major floral and faunal transition from temperate to subtropical species, including broadleaf vegetation (some evergreen), and its increase could represent expansion of maple species (\(\text{Acer}\) spp.), madrone (\(\text{Arbutus menziesii}\)), oak species (\(\text{Quercus}\) spp.), and various pine species (\(\text{Pinus}\) spp.) that currently occur in southwestern Oregon and northern California (McLaughlin 1989). The simulated vegetation changes produce a decrease in vegetation carbon in western Oregon by the end of the century (Figures 5.3d and 5.3e). This decrease is partly the result of changes in vegetation (Figures 5.3b and 5.3c) and partly the result of projected increases in the amount of biomass burned by wildfires, particularly in western Oregon (Figures 5.3f and 5.3g). Eastern Oregon is simulated to gain ecosystem carbon as a result of the simulated expansion of forest and woodland vegetation, while experiencing more and larger wildfires at the same time (Millar et al., 2006).
Figure 5.3 Vegetation types simulated by MC1 (MAPSS Group, contact: R.P. Neilson) on an 8-km grid for 1961-1990 using PRISM climate data (Daly et al., 2000) (A) and for 2070-2099 using climate data simulated by CSIRO-Mk3.0 under the B1 emissions scenario (B) and by UKMO-HadCM3 under the A2 emissions scenario (C). For each grid cell the vegetation type most frequently simulated during the model time period (i.e., the modal vegetation type) is mapped. Future changes in vegetation carbon calculated as 2070-2099 values minus 1961-1990 values (D, E) and future changes in biomass burned calculated as 2050-2099 values minus 1951-2000 values (F, G) were also simulated by MC1. Climate data from CSIRO-Mk3.0 and UKMO-HadCM3 were obtained from the World Climate Research Programme’s (WCRP’s) Coupled Model Intercomparison Project phase 3 (CMIP3) multi-model dataset and the British Atmospheric Data Centre. (Model output: MC1 [MAPSS-CENTURY, ver. 1], R.P. Neilson, U.S. Forest Service MAPSS Group [http://www.fs.fed.us/pnw/mdr/mapss/index.shtml; http://www.fs.fed.us/ccrc/video/skamania-scale.shtml]; Maps: R. Drapek, USFS)
5.4 Future Plant Species Distributions

Projections of future changes in Oregon’s forest, grassland, and shrubland vegetation, such as those presented in Figure 5.3, describe vegetation changes in terms of general vegetation types. In many cases, however, we are interested in how climate change may affect particular plant species in Oregon, such as species that are of particular conservation or economic concern. Rehfeldt et al. (2006) used statistical models to estimate the responses of western U.S. tree species to future climate changes. Their modeled projections indicate that future climate changes could increase the abundance of montane forest and grassland communities at the expense of subalpine, alpine tundra, and arid woodland communities. They also project extensive shifts in the spatial distributions of individual species, especially along elevation gradients. These results generally agree with the vegetation changes projected by the MC1 dynamic general vegetation model described above (Figure 5.3b and 5.3c), although MC1 simulates increased woody vegetation east of the Cascade Range. Statistical model simulations must be interpreted with some caution. Many statistical model projections are based on simple climate-species correlation models that do not incorporate a number of important processes that will affect future species distributions, such as the physiologic responses of plants to elevated atmospheric CO₂ concentration, range limitations from interactions with other species, and the ability for some species to persist for thousands of years, asexually, under climates unfavorable for seedling establishment (Neilson and Wullstein, 1983; Nogués-Bravo, 2009).

Process-based vegetation models are also used to simulate species distributions. Neilson et al. (2005a), using MC1, simulated the potential shifts in distribution for sagebrush (Artemisia spp.) in southeastern Oregon under projected future climate changes. In a different study, Busing et al. (2007), using a dynamic vegetation model accounting for individual tree interactions, simulated relatively small changes in forest composition and basal area for major tree species in the South Santiam River watershed under projected future climate changes for 2050. Their projections also indicated that some tree species, such as western hemlock (Tsuga heterophylla) and Pacific silver fir (Abies amabilis), could shift their distributions upward in elevation during this time (Busing et al. 2007).

Model projections of how species distributions may change in the future are a fundamental part of understanding potential ecosystem responses to climate change. Nonetheless, detecting and quantifying the actual changes in species distributions as they occur (e.g., Lenoir et al., 2008; Kelly and Goulden, 2008) is also a fundamental part of understanding ecosystem responses to climate change and the conditions to which humans and other species must adapt. Statistical modeling and mapping of individual species distributions based on field inventories is an essential component in both validating model projections of future species distribution changes as well as detecting and quantifying the actual rate and amount of change that occurs for individual species. An effective and reliable system of adaptation to the effects of climate change must be based on information and analysis conducted from large-scale, field-based monitoring systems that accurately quantify and delineate the actual status and trends of species distributions. These vegetation mapping efforts, such as the work described in Box 1, rest on the premise that vegetation patterns can be predicted from mapped environmental data (Franklin, 1995), using various hypotheses as to how environmental factors control the distribution of species and communities (Guisan and Zimmerman, 2000). In a number of
modeling studies, climatic gradients that include precipitation and temperature have been found to explain the distribution, composition and structure of forest vegetation in Oregon and the western U.S. using modeling approaches ranging from statistical to process-based models (Ohmann and Spies, 1998; Ohmann and Gregory, 2002; Rehfeldt et al., 2006; Coops et al., 2009). However, more experimental and observational information is still needed regarding how species are able to maintain their current distributions (at all spatial scales from micro-habitat to its full distribution) under modern climate, or to extend their range under a changing climate (e.g., Neilson and Wullstein, 1983).

5.5 Disturbance Regimes

Climate change can affect both the frequency and severity of disturbances that play a vital role in the natural dynamics of Oregon’s vegetation. Events such as wildfires, insect outbreaks, diseases, droughts, windstorms and landslides profoundly influence ecosystem dynamics in terms of ecosystem structure, composition, and functioning (e.g., Perry and Amaranthus, 1997; Franklin et al., 2002; Campbell et al., 2004; Law et al., 2004; Wilson, 2004; Busing et al., 2007; Spies, 2009; Meigs et al., 2009). One of the most important disturbance agents for Oregon’s vegetation is wildfire. An average of 317,300 acres burned per year over the last five years and 648,000 acres burned in 2007 (NIFC, 2010). The area of disturbance from insect pests and diseases is generally of the same order of magnitude as that of wildfires in western U.S. forests (Hicke et al., 2006). Mountain pine beetle (*Dendroctonus ponderosae*) is the most important insect pest in Oregon, affecting 348,400 acres per year on average from 2004 to 2008 (Nelson et al., 2009). A number of important disturbance agents are strongly climate sensitive (Dale et al., 2001) and interact dynamically (see Box 2). Consequently, changes in disturbance regimes are expected to constitute the most profound climate change impacts in forest ecosystems of the Pacific Northwest (Franklin et al., 1991; Littell et al., 2009b). The following sections focus on wildfire, insects and diseases, drought, invasive species, climate extremes, and the interactions of disturbances.

5.5.1 Wildfire

The frequency and extent of wildfires is strongly related to climatic factors. Analyses of fire history reveal a significant correlation of fire activity with decadal-scale (Pacific Decadal Oscillation), episodic (El Nino/Southern Oscillation) and interannual natural climate variation, with larger areas burned during warm and dry phases/years (Hessl et al. 2004; Pierce et al., 2004; Gedalof et al., 2005; Trouet et al., 2006; Kitzberger et al., 2007, Heyerdahl et al. 2008). A recent study found that both the frequency of large wildfires and the duration of the fire season increased sharply in the mid 1980s in the western U.S., an increase that could largely be explained by changed climatic drivers (Westerling et al., 2006). Critical climate-sensitive processes, however, differ by ecoregion and vegetation type. In mesic forest types (i.e., predominately west of the crest of the Cascade Range), dry and warm summers exert the strongest climatic influence on forest area burned, depleting fuel moisture and creating favorable conditions for fire spread (Littell et al., 2009a). In contrast, in drier forest types in eastern Oregon the main climatic influence on wildfire activity is via facilitation of vegetation growth in winter(s) prior to the fire (i.e., fuel availability is an important limiting factor for fires) (Littell et al., 2009a). These complex climate-vegetation interactions are important for assessing the potential impacts of climate change on wildfire activity, and they are replicated in the
process-based fire simulations produced by vegetation models, such as MC1 (Millar et al., 2006; Rogers, 2009).

Despite the different seasonal climate conditions influencing fire occurrence for different forest types, an increase in fire activity is expected for all major forest types in Oregon and the western U.S. under projected climate changes (Figures 5.3f and 5.3g; Bachelet et al., 2001; Whitlock et al., 2003; Keeton et al., 2007). A 78% increase in forest area burned by the middle of the 21st-century is estimated for the Pacific Northwest (Spracklen et al., 2009b). Increases of up to 6-fold in area burned are estimated for regions in the Pacific Northwest by the end of the century (McKenzie et al., 2004; Littell et al., 2009b). However, estimates of projected future changes in area burned vary significantly depending on the climate scenario and estimation method used. As Mote et al. (Chapter 1) note, estimates of future precipitation changes for the PNW vary, with some projections indicating wetter than present conditions and other projections indicating drier than present conditions. Whether the future climate is wetter or drier will significantly affect potential changes in area burned by fire, as will increases in interannual to interdecadal climate variability.

The actual occurrence of future fires, however, is not only driven by favorable climate conditions but also requires a source of ignition (usually lightning or human ignition sources) and a mechanism for rapid fire spread (strong winds and topography). These factors are strongly influenced by local conditions and, to date, are not fully represented in many climate change projections. However, growing evidence points towards increasing lightning activity over the western U.S. under climate change (Price and Rind, 1994; Del Genio et al., 2007).

5.5.2 Insect pests and diseases

The forest area in Oregon affected by mountain pine beetle has been increasing for the last eight years (Nelson et al., 2009; Figures 5.4 and 5.5). Increasing winter temperatures enhance winter survival probability of the bark beetle, particularly in high elevation areas (Regniere and Bentz, 2007; Bentz, 2008). In addition, host susceptibility may increase under climate change as a result of increased drought stress. The highest vulnerability occurs during extremely high temperature and extended drought (Littell, 2009b), weakening the ability of trees to repel beetle parasitism (see Raffa et al., 2008). However, in order to overwhelm a tree’s defense system, the mountain pine beetle depends on a synchronous emergence of a large number of adult beetles at an appropriate time of the year (a phenomenon referred to as “adaptive seasonality”). Rising temperatures will cause this adaptive seasonality to decrease in low elevation forests, which in some model simulations results in a distinct overall projected decrease in the area attacked by mountain pine beetle (Williams and Liebhold, 2002; Hicke et al., 2006; Littell et al., 2009b). High elevation forests, however, are likely to experience an increase in adaptive seasonality and hence of mountain pine beetle attacks in the coming decades, increasing the pressure on high elevation pine species such as whitebark pine (Pinus albicaulis). Later in the 21st century, adaptive seasonality is expected to also decline in high elevation forests, if warming continues as projected (Hicke et al., 2006).
Climate change also affects forest diseases, increasing the ability of pathogens to survive through the winter and shortening regeneration times of bacteria and fungi. In the Oregon Coast Range, Swiss needle cast (caused by the fungus *Phaeocryptopus gaeumannii*), a foliage disease of Douglas-fir, was found to be highly sensitive to winter temperature with an average predicted increase of 9.2% in infected needles per 1°C increase in temperature (Manter et al., 2005). Another important disease that is strongly climate sensitive is sudden oak death (caused by the non-native pathogen *Phytophthora ramorum*), which has been spreading northward from California into southwestern Oregon forests since 2001. In addition to mild and wet conditions (Anacker et al., 2008), heavy precipitation events facilitate infection of new trees (Rizzo and Garbelotto, 2003). Infected trees are more susceptible to mortality during droughts, which can lead to large diebacks of infected trees during extended drought periods (Frankel, 2008). More extreme weather conditions projected for the future could thus facilitate sudden oak death in Oregon. In general, warming is likely to encourage northward expansion of more southern insects and diseases, while longer growing seasons may allow more insect generations per year.
5.5.3 Drought

Drought is a critical predisposing factor that leads to both wildfires and major insect outbreaks. However, it is also an important disturbance agent in itself and, while most vegetation types in Oregon are well adapted to the state’s dry summers, projected future changes in the hydrological regime (Chapter 3) have the potential to cause large-scale tree mortality (Neilson, 1993; Bachelet et al., 2001; Bachelet et al., 2003; Mote et al., 2003; Whitlock et al., 2003; Lenihan et al., 2008). Projected increases in temperature will lengthen the growing season and increase evaporative demand, causing ecosystems to extract all available soil moisture before the end of the growing season. Recent increases in water deficit, for instance, are thought to be contributing to increased mortality rates in old-growth forests throughout the western U.S. (van Mantgem et al., 2009). A modeling study focusing on Ponderosa pine (*Pinus ponderosa* Doug. Ex Loud.), an important tree species in the dry forests of eastern Oregon, indicated that prolonged drought resembling the conditions of the 1920s and 1930s would surpass the species’ physiological thresholds on 18-30% of its current area (Coops et al., 2005). Empirical data from the pine-woodland transition zone in southern Oregon corroborate the sensitivity of these ecosystems to drought, particularly on sandy and rocky soils (Knutson and Pyke, 2008). For the forests of neighboring Washington State, severe water-limitation was recently projected to rise by 32% in the 2020s and an additional 24% by the 2080s, compared to conditions of the 20th century (Littell et al., 2009b), although these projections do not include potential increases in CO2-induced water-use efficiency. Allen et al. (2010) provide a global overview of drought-related tree mortality.

**Case Study 2: Disturbance interactions in Oregon’s forests**

The prevailing disturbance regime in Oregon’s forests is, in many cases, the result of a disturbance complex of interacting agents. Drought, for instance, can directly stress a tree’s physiology, but also increases the vulnerability of trees to insect attacks and increases the probability for fire (Figure 5.5). Well-documented positive relationships also exist between bark beetle outbreaks and fire: Trees weakened by fire have a higher vulnerability to insect attacks (Hood and Bentz, 2007; Breece et al., 2008; Youngblood et al., 2009). Similarly, insect outbreaks increase fuel loads and thus fire hazard and severity (Lundquist, 2007; Page and Jenkins, 2007; Jenkins et al., 2008). Considering these interactions, the impacts of climate change on the disturbance regime of Oregon’s forests will likely exceed the effect on any single agent or factor (Franklin et al., 1991).

The interactions between disturbances and vegetation need to be taken into account when evaluating expected changes in disturbance regimes. With regard to the projected range changes for mountain pine beetle, for instance, the simultaneous changes in the insect’s host species distribution also need to be considered. Simulation studies indicate that the climatically suitable habitat for the insect may shift at twice the rate of that of its host species, thus a net contraction of the mountain pine beetle outbreak range may occur (Williams and Liebhold, 2002). While climate-induced tree species migration is a relatively slow process, management measures influencing forest structure and composition have an immediate effect on disturbances (Hessburg et al., 2005; Ager et
al., 2007). In many cases management activities may be as important a driver of disturbance regimes as climate (Weisberg and Swanson, 2003; McCloskey et al., 2009). Fire suppression efforts over the past several decades resulted in smaller, low-severity fires despite indications of an increase in (mostly anthropogenic) ignitions (Weisberg and Swanson, 2003). The resulting increased stand density and decreased landscape heterogeneity, however, raise the likelihood of future large-scale, high severity wildfire events (Schoennagel et al., 2004; Keeton et al., 2007). In summary, the combined expectations regarding increases in water limitation, wildfire activity, high elevation adaptive mountain pine beetle seasonality, and insect host vulnerability, as well as their interactions, suggest that forest ecosystem functions and services of a large share of Oregon’s forests may be affected by altered disturbance regimes as the region’s climate changes.

![Figure 5.5](image.jpg)

**Figure 5.5** (Left) A stand burned by the B&B Complex Fires (2003, Mt. Jefferson Wilderness, Deschutes National Forest, Willamette National Forest) near Santiam Pass (Photo: R. Seidl, 2009). (Right) The effects of a mountain pine beetle outbreak around Little Three Creek Lake, Deschutes National Forest (Photo: E. Seidl, 2010).

5.5.4 Invasive species

As climate changes, plant species that are not native to Oregon may be able to expand into the state, or expand their ranges if they already occur in the state. These invasive species may be
native to other parts of North America or they may be exotic species that have been introduced to North America from other places. When exotic species are introduced into suitable environments they can often expand rapidly, outcompeting native species. In the rangelands of eastern Oregon a number of exotic plant species, such as yellow starthistle (*Centaurea solstitialis*), are outcompeting native bunchgrasses that provide important forage for livestock on rangelands (Roché and Thill, 2001).

Invasive plant species also can alter fire regimes, particularly in relatively dry ecosystems such as occur in many parts of eastern Oregon. In these regions, the fuel cover provided by native vegetation is relatively sparse, limiting the ability of fires to spread across the landscape. Invasive species can increase the fuel load and fuel connectivity in these systems allowing fires to spread more easily. For example, cheatgrass (*Bromus tectorum*), an exotic annual grass in eastern Oregon, is highly flammable and can alter fire regimes by increasing the frequency of fires (Brooks et al., 2004). Statistical models indicate that cheatgrass could continue to expand in eastern Oregon under some future climate projections (Bradley, 2009). More information about invasive species in Oregon is available on the Oregon Invasive Species Council website (http://www.oregon.gov/OISC/).

### 5.5.5 Climate extremes

Many of the potential changes in disturbance described above will be driven by potential future changes in climate extremes. The western US is projected to experience increased heat wave severity under projected future climate change (Meehl and Tebaldi, 2004), which could lead to increased drought-related plant mortality (e.g., van Mantgem et al., 2009) and fire occurrence. Oregon could also experience more intense precipitation events. Leung et al. (2004) describe regional climate model simulations of future increases in winter extreme daily precipitation events for the Oregon Cascade Range. Tebaldi et al. (2006), using global climate model simulations, also reported increased precipitation intensity for parts of the Pacific Northwest. Changing climatic extremes could also affect many other types of disturbances, such as damage from wind and snow events, landslides, and flooding. Meehl et al. (2007), CCSP (2008a) and Chapter 1 provide additional information about potential future changes in extreme climate events.

### 5.6 Carbon in Oregon’s Forests

Oregon’s forests have the potential to play a significant role in mitigating atmospheric CO₂ concentrations given the long-lived nature of many of the region’s tree species and the dead material they form, such as the coarse woody debris (e.g., dead branches, fallen trees, etc.) found on the forest floor (Krankina and Harmon, 2006). Potential carbon stores in the Pacific Northwest are among the highest for forests in the world (Smithwick et al., 2002; Homann et al. 2005). The current carbon stores of these forests, however, are substantially below this potential (Smithwick et al., 2002; Hudiburg et al., 2009), which suggests a change in management could result in a major carbon sink within the region for many decades. Either an increase of the interval between harvests or a reduction in the amount of carbon removed each harvest would lead to an increase in average carbon stores in forests (Harmon et al., 2009). Indeed, changes in management of national forests under the Northwest Forest Plan (Mouer et al. 2005), which essentially increased the interval between harvests, have already led to a substantial increase in
carbon stores in the Pacific Northwest and have likely changed the carbon balance of Oregon’s forests from adding carbon to the atmosphere to removing carbon from the atmosphere in less than a decade (Cohen et al., 1996; Turner et al., 2007; Hudiburg et al., 2009). While this trend helps mitigate atmospheric CO$_2$ emitted in Oregon, it also means that policy changes on national forests are unlikely to further increase forest carbon sinks. Some policies, such as thinning forests to increase climate adaptability or to alter fire behavior, would lead to decreases in forest carbon stores (Mitchell et al., 2009). To increase the uptake of carbon by Oregon’s forests further it will be necessary to increase carbon stores on state and private lands. Increasing carbon stores would require increasing the interval between harvests and/or reducing harvest amounts, which may be unrealistic without having funds available to offset losses in traditional harvest-based revenues (Alig et al., 2002).

An alternative to increasing carbon stores within the forest is to harvest wood and store some of this carbon within wood products (Perez-Garcia et al., 2005). Under current manufacturing, use, and disposal practices this alternative is unlikely to increase the overall carbon store of the forest sector, which includes the forest and wood products derived from the forest (Harmon et al., 2009). Manufacturing, use, and disposal of harvested wood all entail significant carbon losses that are either as large as or larger than those in the forest itself (Krankina and Harmon, 2007). Wood products carbon offsets associated with biofuels and substitution of wood for more energy intensive building materials, such as steel and concrete, can theoretically increase the carbon “stores” of wood products beyond that stored in the forest itself (Perez-Garcia et al., 2005; Lippke et al. 2010). However, several issues need to be recognized regarding these offsets. First, most analyses have presented theoretical maximum product substitution offsets and ignored the effects of additionality (i.e., degree to which practices differ from business as usual or statutory requirements), permanence and replacement of existing wood products, and end-user preferences for building materials. If these factors are included, then substitution effects are substantially lower than the theoretical maximum and unlikely to surpass carbon stores in forests for many centuries if at all. Second, depending on the starting condition of the forest, both product substitution and forest-related biofuels can create carbon debts that delay carbon benefits. For example, biofuels harvested from existing forests could offset fossil fuel releases of carbon, but recent studies have indicated that carbon debts associated with the energy used during biofuel harvests, decreased carbon stores in forests, and differences in carbon to energy ratios could persist for decades to centuries, implying a significant temporal lag in net carbon uptake (Fargione et al., 2008; Searchinger et al., 2009). Third, being offsets, the effectiveness of both biofuel and product substitution will vary with the duration of the offset; the longer the delay in releasing fossil fuel carbon, the more effective offsets become: An offset with a 1 year delay would have little impact on atmospheric CO$_2$ concentrations, whereas an offset of hundreds of years would have a much greater impact. Unfortunately, the duration of offsets is not well understood at this point, but it is unlikely to be infinite as tacitly assumed in many current analyses. Finally, while offsets are often counted as carbon stores, they are difficult to directly inventory because they are not physically in an identifiable location, whereas carbon stored in forests can be more directly inventoried and quantified.

Increasing the store of carbon in Oregon’s forests in the near term may increase carbon releases in the future. The degree to which this occurs will depend upon the degree, frequency, and severity of disturbance changes in the future (Smithwick et al., 2007). If disturbances increase in
frequency and/or severity, then carbon stores will decrease. For example, an increase in the frequency and severity of wildfire in Oregon will kill vegetation as well as combust live and dead plant matter, releasing carbon to the atmosphere (see Figure 5.3d-g). However, changes in disturbance frequency often need to be substantial before a major change in carbon stores occurs. In one analysis, a halving of average wildfire intervals from 200 to 100 years had relatively little impact on average carbon stores of forests in Yellowstone National Park (Kashian et al., 2006). The loss of carbon also depends on the degree that forests can reestablish after disturbances in the future. The effect of forest regeneration time is strongly influenced by the time interval between disturbances. However, unless disturbance intervals drop below 100 years, there appears to be little effect of regeneration rate on average carbon stores (Harmon and Marks, 2002). Still, if forests fail to regenerate, then substantial losses would occur as forests tree species generally store more carbon than grasslands or woody plants of lower stature. Regeneration failure of tree species is most likely on sites with low moisture and/or extreme temperatures. Some broadleaf tree species may be able to shift to asexual reproduction under these conditions, thus retaining some carbon sequestration capacity (e.g., Neilson and Wullstein, 1983).

![Figure 5.6](image)

**Figure 5.6** Average total system (ecosystem and forest products) carbon stores (Mg/ha) over a harvest rotation interval for different levels of removal (percent of live trees harvested) with an aggregated cutting pattern. An aggregated cutting pattern represents one contiguous harvest block. These data represent a system in which 75% of the harvested carbon is converted to long-term forest products with losses of 1% per year. (Figure from Harmon et al., 2009)

Many adaptation efforts are likely to decrease carbon stores in Oregon forests. Regenerating species or individuals of trees better adapted to future climate could release carbon if harvests are used to enhance regeneration, particularly if the time interval between harvests is shorter than current practices. This is because shortening the interval between harvests lowers the
average carbon store of the forest sector (Figure 5.6). Unfortunately, this constraint would likely slow efforts to adjust forests to future climates. Thinning to reduce water stress would also lower carbon stores in the short-term, but might assure more carbon is stored over the long-term if disturbance severity or frequency greatly increases or if forests disturbed in the future are unable to regenerate. It should be noted that due to the feedback between fire severity and fuel level, a period of more frequent, severe fires will not persist (more frequent removal of fuels means less fuel, which means lower severity fires). Removal of fuels to alter fire behavior and severity to reduce carbon emissions would, in most cases, lead to substantial losses of carbon stores as the amount of carbon needed to be removed to alter fire severity exceeds the amount released by fires at least 10-fold (Mitchell et al., 2009). Using removed fuels as a biofuels feedstock or for wood products narrows the carbon cost of fuel treatments, but it does not entirely eliminate the carbon debts created by these treatments. Assuming fuel removal continues to produce biofuels, it may take many decades to centuries to pay back the carbon debts incurred (Mitchell et al., 2009).

5.7 Genetic Changes

How plant species in Oregon respond to future climate change will be affected by the genetic variation in adaptive traits. Natural selection has resulted in current plant populations that are genetically adapted to their local climates, resulting in adaptive genetic variation within species that is structured across the landscape. As a result, the climatic tolerances of individual plant populations of a species are considerably narrower than the tolerances of the species as a whole. Populations are the primary biological unit of interest when evaluating adaptation to current and future environments.

When grown in a common environment, most forest tree species show geographic variation in such traits as growth rate, timing of bud flush or bud set, cold hardiness, germination rate, or biomass partitioning that correspond to gradients in temperature and moisture of source locations (Morgenstern, 1996; Howe et al., 2003; St.Clair et al., 2005; Savolainen et al., 2007). Maps of seed zones and seed transfer guidelines have been developed for forest trees in Oregon to manage this geographic genetic variation and ensure that planting stock used in reforestation is adapted to the environmental conditions where reforestation is occurring (Randall, 1996; Randall and Berrang, 2002; Johnson et al., 2004). More recently, seed zones in Oregon are being developed for native plants other than trees to ensure success in restoration projects (Erickson et al., 2004). These guidelines assume, however, that climates are static over long time periods, an assumption that we now know is unlikely. A study of Douglas-fir in western Oregon and Washington indicated that current practices of using local seed sources from within current seed zones for planting new stands would result in a high risk of poorly adapted Douglas-fir stands by the end of the century (St.Clair and Howe, 2007). The Douglas-fir populations expected to be adapted to future climates were located 500 to 1000 m lower in elevation and 2 to 5 degrees latitude (~200 km to ~540 km) further south.

With rapidly changing environments, plant populations face three possibilities (Aitken et al., 2008): (1) migration to new habitats in which they are adapted; (2) adaptation via natural
selection as climate changes; and (3) extinction. The persistence of plant populations by migration will depend upon the rate of migration via seed dispersal and establishment of new stands and upon the rate of gene flow via pollination from distant stands. Most studies of migration rates based on the establishment of new stands rely upon paleobotanical studies of range shifts over the last 25,000 years. Although estimates of historical horizontal migration rates vary widely from 10 km per century to exceptional examples of 150 km per century, all estimates are well below the 300 to 500 km per century that may be required to keep pace with current climate projections for the next century (Davis and Shaw, 2001; Aitken et al., 2008). Range shifts in mountainous areas may occur more quickly since locations of adapted populations could be relatively close, and, indeed, evidence of elevational range shifts associated with climate warming in the last century is being found (Millar et al., 2004; Lenoir et al., 2008). With respect to the potential for pollen flow to move adaptive genetic variation into populations, little is known for most species in native stands. Although pollen may be carried by wind over long distances, differences in the time of flowering between stands in different environments may limit the effectiveness of pollen from distant stands for promoting gene flow (Silen, 1963; White et al., 2007). Further research is needed to study effective pollen flow in native stands, particularly in highly heterogeneous environments such as the mountainous areas of Oregon.

Plant population persistence through adaptation via natural selection will depend upon a variety of factors, many for which we have insufficient knowledge from native populations to allow accurate predictions of the potential for evolution. Responses to natural selection within populations depend upon phenotypic variation (variation that is observable), genetic variation and the heritability of traits important for survival and reproduction, as well as the intensity of selection (Falconer and Mackay, 1996). High reproduction rates and large population sizes may allow for higher intensities of selection without population sizes being greatly reduced. Small populations may lead to loss of genetic variation through inbreeding and random processes called genetic drift.

Gene flow from adjacent populations may be an important contributor to genetic variation within populations of a species. The effectiveness of gene flow for contributing genetic variation important for adapting to a changing environment will depend on where the population is located within the species’ geographic range. Populations of a species that are on the periphery of the species’ range may have lower genetic variation as a result of their smaller population sizes and they may experience reduced gene flow as a result of their more isolated position on the edge of the species’ range (Lesica and Allendorf, 1995). Any gene flow from populations adapted to warmer climate conditions into peripheral populations adapted to cooler climate conditions (e.g., at higher elevations) could result in the introduction of genetic variation into these populations that could be favorable for adapting to future climate changes. Similarly, peripheral populations at the warmer edges of a species’ range that receive gene flow primarily from populations adapted to cooler climates could become increasingly maladapted to warmer climates (Davis and Shaw, 2001; Aitken et al., 2008). Such might be the case for species such as Douglas-fir, western redcedar, or Sitka spruce in southwest Oregon, which might see their southern ranges move north as environments become warmer and drier.
Finally, adaptation via natural selection requires generation turnover. Populations of long-lived species, such as Douglas-fir populations in Oregon, may persist for centuries before much of the existing population is replaced, during which time they may increasingly diverge from a population that is best adapted to the new environment. Generation turnover in short-lived species, on the other hand, allows for the opportunity to better keep pace with a changing environment given sufficient genetic variation and high intensities of selection (Lenoir et al., 2008).

The third possibility is that local populations may become extinct. Following from the previous discussion, populations that may be most vulnerable to climate change include small, fragmented and disjunct populations, particularly those at the low elevation and southern latitude edges of a species’ range. Although they may persist for a while, long-lived species may be at a greater threat from climate change than short-lived species. Rare species and populations already threatened by other factors such as habitat loss, fire, disease, and insects may be at an even greater risk of loss given the added impact of climate change.

Fortunately, there are a number of adaptive management options that may be used to improve the ability of plant populations to respond to future climate change. The selection and adaptation of plant populations in new environments may be facilitated by managing the genetic diversity inherent among and within plant populations by:

- reducing landscape fragmentation and maintaining corridors for migration and gene flow;
- planting species and populations in new locations in which they may be expected to be adapted in the future (a process referred to as assisted migration);
- establishing “genetic outposts,” genetically divergent planted stands that may facilitate gene flow for adaptive variation into adjacent native stands;
- using breeding programs to enhance adaptive traits such as drought hardiness, cold hardiness (for reasons of increased climatic variability and starting to move populations upslope), and pest resistance or tolerance;
- increasing genetic diversity within stands and across landscapes by planting mixtures of populations, and allowing for natural and human selection within diverse stands by planting at higher densities with the possibility of thinning;
- conserving genetic diversity by maintaining adaptive potential and reducing disturbance probability and intensity in native stands (in situ conservation) and by collection and storage of the most vulnerable populations (ex situ conservation).

5.8 Uncertainties in Projections of Future Vegetation Change

There are a number of uncertainties that are associated with our understanding of how Oregon’s vegetation may respond to potential future climate changes. Climate model simulations, such as those discussed in Chapter 1, provide estimates of future climate changes but they cannot predict with perfect accuracy how Oregon’s climate will change in the future. There is relatively good agreement among climate model simulations that air temperatures in Oregon will increase in the future. It is more difficult for the current generation of climate models to accurately simulate present and future precipitation changes, although the models
continue to be refined and improved. As climate model simulations become more accurate, simulations of future vegetation changes will improve as well. Giorgi (2005), Meehl et al. (2007), CCSP (2008b), and Wiens and Bachelet (2010) describe some of the uncertainties associated with future climates simulations and their use.

Models that simulate potential vegetation responses to climate change also have limitations that affect their ability to accurately simulate vegetation. Many models use complex hydrological, ecological, and physiological processes in order to simulate the effects of climate change on vegetation structure, composition, and function. These models require detailed information on the physiological and ecological responses of individual plant species to climate change. Some plant species in Oregon, particularly those with economic value such as Douglas-fir, have received significant research attention. For many other plant species, however, relatively little is known about how they may respond to future climate changes.

In some mechanistic vegetation models, species can be aggregated into plant functional types with similar physiological and ecological properties, thus considerably simplifying the data requirements for running the models. However, information is also lost by such aggregation and, as a result, ecosystem resilience may be underestimated. Novel future climate regimes may produce no-analog plant communities, thus rendering approaches where past plant functional types are extrapolated into the future more tenuous (Williams and Jackson, 2007; Hobbs et al., 2009). Species distribution models address potential changes at the level of individual species (e.g., McKenney et al., 2007). Some of their core assumptions, such as a species being in equilibrium with its environment, limited consideration of biotic interactions, and a simplified view of landscape level dispersal, also introduce uncertainties into their simulations of species distribution responses to climate change (Wiens et al., 2009). Recent efforts thus aim at increasingly incorporating process-based understanding into such models (Buckley et al., 2010; Franklin, 2010), since capturing fundamental ecological processes is widely seen as a key aspect of making these models more robust for simulating species responses to changing climate conditions.

This focus on improving model simulations is also being applied to understanding and projecting climate change impacts on disturbance regimes, where efforts to understand the complex interactions between climate, vegetation and disturbance agents (Dale et al., 2001; Turner, 2010) are increasingly fostering process-based models (Seidl et al. 2010). Models are essential tools to predict future impacts of climate change, but numerous empirical studies in Oregon and elsewhere are also instrumental in shaping our understanding of how plants will respond to future climate changes. The insights gained from these studies form the basis for improvements in vegetation simulations.

5.9 Summary and Conclusions
Projected future climate changes will affect vegetation in Oregon. Evidence from both the paleoenvironmental record and more recent empirical studies clearly indicates that Oregon’s vegetation has responded to past changes in climate. The ranges of plant species have shifted in latitude, longitude, and elevation in response to past climate changes and the distributions of different species have overlapped through time. More recently, earlier dates of first bloom for
some plant species (Cayan et al., 2001) and increased tree mortality in some locations in Oregon (van Mantgem et al., 2009) have been correlated with climate changes that have occurred over the last century. Past vegetation responses to climate change also have been affected by climate-driven changes in the frequency and magnitude of disturbance regimes, such as wildfire and droughts.

Vegetation modeling studies using estimates of future changes in climate and atmospheric CO₂ concentrations indicate that areas of subalpine forest and alpine tundra at high elevations in Oregon and shrubland areas in eastern Oregon will contract under projected future climate changes. Species that currently occur at lower elevations may expand upward in elevation over time. Plant species will respond individualistically to future climate change, and genetic variations within species will affect the ability of plant populations to adapt to changing climate conditions. Future climate change will also alter disturbance regimes in Oregon, such as fire, drought, and insect and disease outbreaks, as well as interactions between native species and invasive species in the region.

There are still many uncertainties associated with projections of future climate changes and the effects of these changes on vegetation. As research on climate change continues, data and models will improve, allowing for a better understanding of potential future climate changes and their effects. Finally, it has to be noted that this chapter provides only a brief overview of some of the potential effects of climate change on Oregon’s vegetation. The references cited in the text will provide the reader with additional sources of information.

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