Utah State University

DigitalCommons@USU

Aspen Bibliography

Aspen Research

2018

Effects of Climate Change on Forest Vegetation in the Northern Rockies Region

Robert E. Keane

Mary Frances Mahalovich

Follow this and additional works at: https://digitalcommons.usu.edu/aspen_bib

Part of the Agriculture Commons, Ecology and Evolutionary Biology Commons, Forest Sciences Commons, Genetics and Genomics Commons, and the Plant Sciences Commons

Recommended Citation

Keane, R. E., M. F. Mahalovich, B. L. Bollenbacher, M. E. Manning, R. A. Loehman, T. B. Jain, L. M. Holsinger, A. J. Larson, and M. M. Webster. 2018. Effects of Climate Change on Forest Vegetation in the Northern Rockies Region. Pages 128-173, In Climate change vulnerability and adaptation in the Northern Rocky Mountains-Part 1. USDA, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

This Contribution to Book is brought to you for free and open access by the Aspen Research at DigitalCommons@USU. It has been accepted for inclusion in Aspen Bibliography by an authorized administrator of DigitalCommons@USU. For more information, please contact digitalcommons@usu.edu.

UtahStateUniversity
MERRILL-CAZIER LIBRARY

Chapter 6: Effects of Climate Change on Forest Vegetation in the Northern Rockies Region

Robert E. Keane, Mary Frances Mahalovich, Barry L. Bollenbacher, Mary E. Manning, Rachel A. Loehman, Terrie B. Jain, Lisa M. Holsinger, Andrew J. Larson, and Meredith M. Webster

Introduction

The projected rapid changes in climate will affect the unique vegetation assemblages of the Northern Rockies region in myriad ways, both directly through shifts in vegetation growth, mortality, and regeneration, and indirectly through changes in disturbance regimes and interactions with changes in other ecosystem processes, such as hydrology, snow dynamics, and exotic invasions (Bonan 2008; Hansen and Phillips 2015; Hansen et al. 2001; Notaro et al. 2007). These impacts, taken collectively, could change the way vegetation is managed by public land agencies in this area. Some species may be in danger of rapid decreases in abundance, while others may undergo range expansion (Landhäusser et al. 2010). New vegetation communities may form, while historical vegetation complexes may simply shift to other areas of the landscape or become rare. Juxtaposed with climate change concerns are the consequences of other land management policies and past activities, such as fire exclusion, fuels treatments, and grazing. A thorough assessment of the responses of vegetation to projected climate change is needed, along with an evaluation of the vulnerability of important species, communities, and vegetation-related resources that may be influenced by the effects, both direct and indirect, of climate change. This assessment must also account for past management actions and current vegetation conditions and their interactions with future climates.

This chapter addresses the potential impacts of climate change on forest vegetation in the Forest Service, U.S. Department of Agriculture (USFS) Northern Region and the Greater Yellowstone Area (GYA), hereafter called the Northern Rockies region (fig. 6.1). Then, based on the

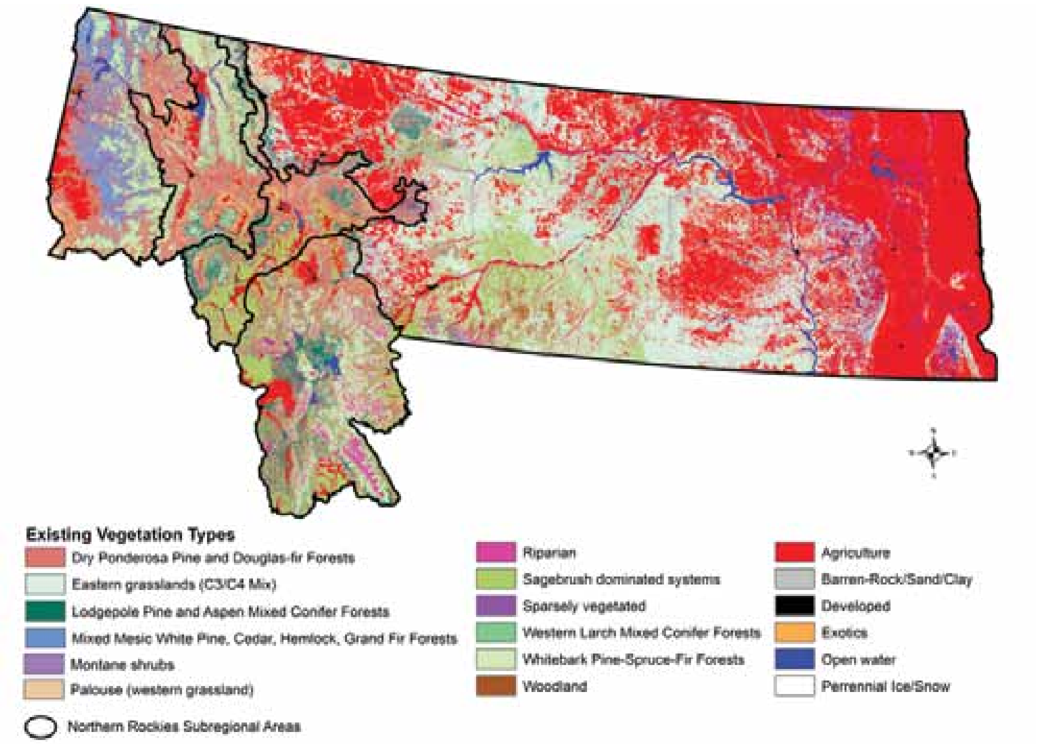

Figure 6.1—The Northern Rockies (NR) assessment area that includes the Northern Region of the U.S. Forest Service and the Greater Yellowstone Area (Yellowstone National Park and surrounding areas). Presented are existing vegetation types by the five geographic sub-areas used to stratify assessments in this report. This map was created from the LANDFIRE Existing Vegetation Type map by aggregating the National Vegetation Classification Standard vegetation types into a set of vegetation types that has some meaning across the NR at this coarse scale. This map is intended to convey current vegetation of the NR.

climate impacts assessment, we present an evaluation of the vulnerability of important tree species, vegetation types, and resources of concern to projected climate change effects. Last, we present various adaptation actions to address climate change vulnerabilities.

This chapter has six major sections. In the introductory section, we define terminology used throughout the chapter and provide background material on the details of the assessment including the scales, geographic areas, and pertinent information used to make our assessments. We discuss how to evaluate uncertainty in climate change projections and vegetation response. We also summarize the methods used to make projections of vegetation response to changing climate. The second section contains important ecological background information that was used to assess climate change impacts and projected climate change responses for 17 tree species, 5 forest vegetation types, and 3 resources of concern. The third section presents information on the tree species, types, or resources of concern that are important when evaluating climate change responses. In the fourth section, we rate the vulnerability of the species, vegetation types, and resources of concern to climate change using information from the third section. In the fifth section, we discuss adaptation strategies and management actions that can be used to address likely impacts of climate change. The final section is a concluding discussion.

This chapter uses the best available information about climate change effects on vegetation in the Northern Rockies. We have integrated broad-scale modeling results with a detailed synthesis of climate change literature for the region. This chapter was written to aid land managers in addressing climate change effects on forest vegetation in land management planning and development of management strategies. This chapter does *not* include the detail needed to address climate change effects at the project level, but it does include valuable information and syntheses that can be used in project planning and in addressing broad concerns at large spatial scales.

Terminology

Climate

Evaluations of climatic trends can be confusing, mostly because weather and climate vary at different spatial and temporal scales. To reduce this confusion, it is often helpful to clearly define the terms and explain the scales that distinguish weather, climate variability, and climate change. *Weather* is the hourly, daily, weekly, or monthly summaries in temperature, precipitation, wind, humidity, and other atmospheric conditions observed at a given place or across a large region. Weather changes at relatively small temporal scales (quickly) and it can change significantly as one moves north or south, east or west, or up and down in elevation. Weather is difficult to predict more than a few days in advance. *Climate* is a statistical characterization of the weather, averaged over many years. The World Meteorological Organization defines it as the average 30-year weather patterns of a region. *Climate variability* is the variation in weather statistics over relatively broad regions and long time periods. Climate variability can be caused by underlying climatic processes, such as changes in patterns of ocean temperatures. The El Niño-Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO), for example, are two sources of climate variability in western North America (Newman et al. 2003). ENSO oscillations occur over 2- to 7-year periods (Gershunov and Barnett 1998), while PDO oscillations occur on a longer cycle (20–50 years) (Heyerdahl et al. 2002). External forcings, such as changes in solar radiation, large volcanic eruptions, and increasing concentrations of greenhouse gases in the atmosphere, also influence climate variability. *Climate change* is a nonrandom change in climate that is measured over several decades or longer. It is technically defined as a statistically significant variation in either the mean state of the climate or in its variability, persisting for an extended period (decades or longer). Like climate variability, climate change may be due to natural internal processes or to external forcings. A *climate scenario* is a plausible and often simplified representation of one possible future climate, based on a consistent set of known principles about the climate system used as input to climate models.

Vegetation

Several general terms are used in vegetation ecology to describe how ecosystems respond to climate change (Intergovernmental Panel on Climate Change [IPCC] 2007). *Adaptive capacity* is the ability of a plant, species, or system to adjust to climate change (including climate variability and extremes) to moderate potential damages, to take advantage of opportunities, or to cope with the consequences. *Exposure* is the nature and degree to which a system is exposed to significant climate variations (Glick et al. 2010). *Sensitivity* is the degree to which a system is affected, either adversely or beneficially, by climate variability or change. The effect may be direct, such as crop yield decreases in response to a higher temperature, or indirect, such as damage caused by an increase in the frequency of coastal flooding due to sea-level rise. *Resilience* is the degree to which ecosystems can recover from one or more disturbances without a major shift in composition or function, whereas *resistance* is the ability of an organism, population, community, or ecosystem to withstand perturbations without significant loss of structure or function (i.e., remain unchanged) (Holling 1973; Seidl et al. 2016). From a management perspective, resistance includes (1) the degree to which communities are able to resist change, such as that from warming climates, and (2) the manipulation of the physical environment to counteract and resist physical and biological change (i.e., cutting, burning, harvest treatments). *Vulnerability* is the degree to which a system is susceptible to, and unable to cope with, the adverse effects of climate change, including associated climate variability and extremes. Vulnerability is a function of the character, magnitude, and rate of climate change and variation to which a system is exposed, its sensitivity, and its adaptive capacity.

Other terms describe how ecosystem processes that are affected by climate change will influence vegetation. In general, *stressors* are any physical, chemical, or biological entity that can induce an adverse ecosystem response. Stressors can arise from physical and biological alterations of natural disturbances, increased unmanaged demand for ecosystem services (such as recreation), alterations of the surrounding landscape, chemical alterations in regional air quality, or a legacy of past management actions (Joyce et al. 2008).

Management

Climate change *adaptation* is an adjustment in natural or human systems in response to actual or expected climatic stimuli or their effects, which is intended to moderate harm or exploit beneficial opportunities (Spittlehouse and Stewart 2004). Adaptation is often referred to as "preparedness," and is based on scientifically supported strategic and tactical activities that support sustainable resource management. Adaptation addresses specific aspects of the sensitivity of resources to an altered climate. An *adaptation tactic* is a specific action described in management and planning documents that supports adaptation strategies and is implemented on the ground (e.g., reducing stem density and surface fuels in a dry mixed-conifer forest, or increasing culvert size on roads along a stream that is expected to have higher flood volumes) (Joyce et al. 2008; Millar et al. 2007a; West et al. 2009). *Assisted migration* is deliberately planting species to colonize new habitats.

In an ideal sense, *ecological restoration* is defined as the practice of reestablishing historical plant and animal communities in a given area and the renewal of ecosystem and cultural functions necessary to maintain these communities now and into the future (Egan and Howell 2001). However, this ideal may be impossible to manage because: (1) little is known about historical conditions; (2) many key species may already be lost; (3) some efforts may be prohibitively expensive; and most importantly, (4) future climates will create novel ecosystems. As a result, The Society for Ecological Restoration has opted for a definition that states that ecological restoration is "the process of renewing and maintaining ecosystem health".

The USFS manual (FSM) direction contained in FSM 2020 includes objectives and a policy for restoration. The objectives of the USFS ecosystem restoration policy are to:

1. *Restore and maintain ecosystems* that have been damaged, degraded, or destroyed by reestablishing the composition, structure, pattern, and ecological processes.
2. *Manage for resilient ecosystems* that have a greater capacity to withstand stressors, absorb and recover from disturbances, and reorganize and renew themselves, especially under changing and uncertain environmental conditions.
3. *Achieve long-term ecological sustainability* and provide a broad range of ecosystem services to society.

The USFS emphasizes ecosystem restoration across all National Forest System lands with the goal of attaining resilient ecosystems. All strategic plans, including the USFS Strategic Plan, land and resource management plans, and area plans, must include goals and objectives to sustain the resilience and adaptive capacity of aquatic and terrestrial ecosystems by reestablishing, maintaining, or modifying their composition, structure, function, and connectivity. The goals and objectives must be established within this framework as defined by laws, Indian treaties and tribal values and desires, and regulations. The goals and objectives also must consider public values and desires, social concerns, economic sustainability, the historical range of variability, ecological integrity, current and likely future ecological capabilities, a range of climate and other environmental change projections, the best available scientific information, and technical and economic feasibility to achieve desired conditions for National Forest System lands. A primary element of an integrated approach is to identify and eliminate or reduce stressors that degrade or impair the ecosystem. Restoration activities should also take into account social and ecological influences at multiple scales and incorporate the concept of a dynamic system and ecological trajectory. Some ecosystems may have been altered to such an extent that reestablishing components of the historical range of variability may not be ecologically or economically possible. Therefore, goals and activities should focus on restoring the underlying processes that create functioning ecosystems.

Functional restoration, which is the restoration of abiotic and biotic processes in degraded ecosystems, focuses on the underlying processes that may be degraded, regardless of the structural condition of the ecosystem. Whereas ecological restoration tends to seek a historical reference condition, functional restoration focuses on the dynamic processes that drive structural and compositional patterns. Functional restoration aims to restore functions and improve structures with a long-term goal of restoring interactions between function and structure. It may be, however, that a functionally restored system will look very different from the historical reference condition in terms of structure and composition, and these disparities cannot be easily corrected because some threshold of degradation has been crossed or the environmental drivers, such as climate, that influenced structural and (especially) compositional development have changed.

Assessment Levels

This chapter uses three levels to assess the impacts of climate change on forest vegetation: species, vegetation types, and resource concerns. We selected these levels and their elements to ensure flexibility when considering the complex ecological concerns across the Northern Rockies. Not only did this structure facilitate consistent and comprehensive assessments for the major management concerns identified in this chapter, but it also allows for the addition of new elements that may be identified in the future.

Species

At the finest level of assessment, we address climate change effects at the species or species group level. We allowed for the use of species groups by aggregating species by genera, guilds, plant functional types, or lifeforms. In this chapter we had only one species group: all cottonwood species (*Populus trichocarpa*, *P. angustifolia, P. deltoides*). This allows us to address regional concerns about important individual species or species groups that might be adversely affected by climate change. All tree species cannot be addressed, so the list of species and species groups included here represents only those species that are identified by the government agencies in the Northern Rockies as critical for addressing both management and climate change concerns.

Vegetation Types

Vegetation type assessment addresses climate change concerns at a coarse vegetation community type level so that future evaulations can be spatially described using a map or geographic information system layer. Five forest vegetation types are assessed to summarize potential climate change impacts: dry ponderosa pine (*Pinus ponderosa*)/Douglas-fir (*Pseudotsuga menziesii*) forests; western larch (*Larix occidentalis*) mixed mesic forests; mixed mesic western white pine (*Pinus monticola*), western redcedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*), and grand fir (*Abies grandis*) forests; lodgepole pine (*Pinus contorta*) mixed subalpine forests; and whitebark pine (*Pinus albicaulis*) mixed upper subalpine forests. These types are shown in both an existing vegetation map (fig. 6.1) and a potential vegetation map (fig. 6.2). Both maps and resultant categories were derived from LANDFIRE data (Rollins 2009), which covered the entire Northern Rockies region. Many of the estimated effects of climate change were based on evaluations of MC2 model simulations (see MC2 section), and figure 6.3 portrays the MC2 vegetation types used to generate the model results in Appendix 6A. The potential vegetation type map and MC2 map can be used to estimate species assemblages in the absence of disturbance.

Resource Concerns

Three resource concerns related to forest vegetation are also addressed in this report. First, we considered the impact of climate change and vegetation response on *landscape heterogeneity*, defined as the diversity in landscape structure or patch characteristics. We assume that landscapes with high heterogeneity are more resilient to disturbance (Ahlqvist and Shortridge 2010; Oliver et al. 2010; Turner 1987). The second resource concern is *timber production,* as represented by timber volume. We attempted to address impacts of climate change on timber production solely via vegetation response, not from disturbance. Finally, we describe climate impacts on *carbon reserves* across the Northern Rockies. Resource specialists in the Northern Rockies region selected these resource concerns.

Geographic Stratification

The Northern Rockies region was stratified into five geographic subregions (fig. 6.1). The Western Rockies subregion includes northwestern Montana and northern

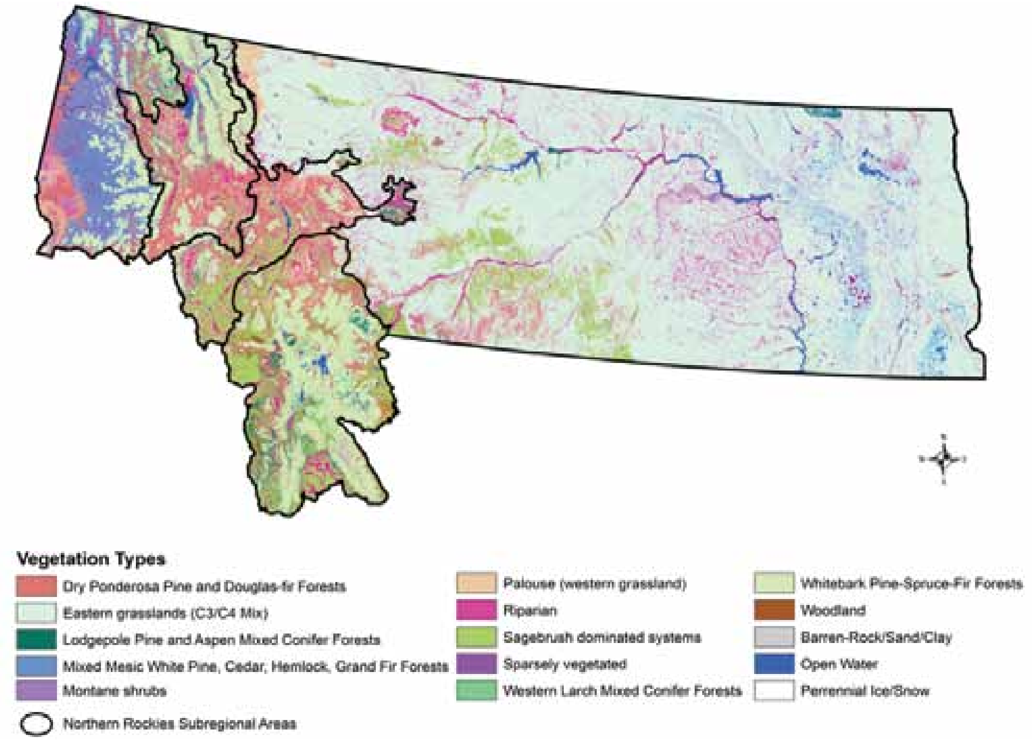

Figure 6.2—Potential vegetation types for the entire NRAP assessment area by the five geographic subregions. This map was created from the LANDFIRE Biophysical settings map by aggregating the National Vegetation Classification Standard vegetation types into a set of vegetation types that has some meaning at this coarse scale.

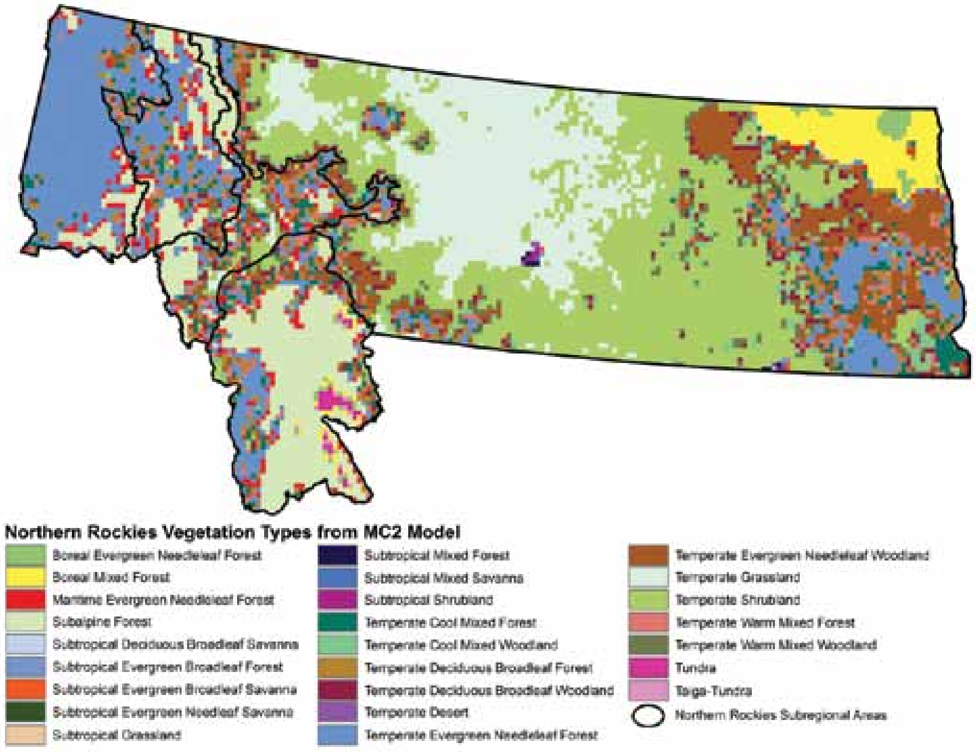

Figure 6.3—The MC2 vegetation types for the assessment area by the five geographic subregions. This map was created from an MC2 modeling effort (see appendix 6A).

and central Idaho. The Central Rockies subregion includes west-central Montana and all lands west of the Continental Divide and north to Canada. The Eastern Rockies subregion includes the Rocky Mountain Front and southwestern Montana. The GYA includes all lands that have been officially designated as part of this high-profile area, including Yellowstone National Park, Grand Teton National Park, the southern end of the Gallatin National Forest and the Beaverhead-Deerlodge National Forest, the western side of the Custer National Forest, and parts of the Shoshone, Bridger-Teton, Caribou, and Targhee National Forests. Last, the Grassland area includes all lands east of the Eastern Rockies subregion boundary to the eastern border of the USFS Northern Region.

The main purpose of dividing the assessment area into five subregions was to restrict climate change projections, impacts, and adaptation options to a specific part of the Northern Rockies region. The five subregions shown in figure 6.1 were included for all authors to standardize the spatial scales of their sections, but some authors of this chapter chose to evaluate climate change impacts at the national forest or finer scale. As a result, this chapter does not include formal sections for each subregion. Instead, the authors tuned their material to the subregion if the data and information allowed.

Uncertainty

Uncertainty is an expression of the degree to which something is unknown. Uncertainty can result from a lack of information or from a disagreement about what is known or even knowable. Uncertainty can also result from known and unknown errors. It may have many types of sources, from quantifiable errors in data to ambiguously defined concepts or terms, or uncertain projections of human behavior. Uncertainty can therefore be represented by quantitative measures, such as a range of values, or by qualitative statements, such as assessment of the judgment of a team of experts. Uncertainty differs from variability; variability is the actual range of a value or ecosystem variable.

All the climate models (global circulation models or GCMs) that predict rapidly warming climates have a high degree of uncertainty (IPCC 2007). Although there is little debate that atmospheric carbon dioxide (CO_2) is increasing and that this increase will cause major changes in climate (IPCC 2007), there is a great deal of uncertainty about the magnitude and rate of climate change (Roe and Baker 2007; Stainforth et al. 2005). This uncertainty will almost undoubtedly increase as climate projections are made at finer resolutions, for different geographic areas, and for longer time periods (Knutti and Sedlacek 2013). The range of possible projections of future climate from GCMs (an increase in global average annual temperature of anywhere from 2.9 to 14.4 ^{o}F) is much greater than the variability of climate over the past two or three centuries (Stainforth et al. 2005), and the variability across GCMs is greater than the variability in the climate projections of each model. Because it is impossible to know whether society will respond to climate change by employing technological innovations to minimize CO_2 emissions or to mitigate its effects, most GCMs also simulate a suite of scenarios that capture a range of possible strategies to deal with climate change, introducing yet another source of uncertainty. Moreover, it is the high variability of climate extremes, not the gradual change of average climate, that will drive most ecosystem responses to the climate-mitigated disturbance and plant dynamics, and these rare, extreme events are the most difficult to predict (Easterling et al. 2000).

Yet another source of uncertainty in attempting to predict ecosystem change is introduced when we try to predict how the Earth's vegetation and ecosystems will respond to highly uncertain climate change (Araujo et al. 2005). Mechanistic ecological simulation of climate, vegetation, and disturbance dynamics across landscapes is still in its infancy (Keane and Finney 2003; Sklar and Costanza 1991; Walker 1994). Many current ecosystem simulation models are missing the important direct interactions of disturbance, hydrology, and land use with climate that will surely dictate effects on plant distributions (Notaro et al. 2007). Little is known about the interactions among climate, vegetation, and disturbance, and interactions among different disturbance regimes (fire and beetles, for example) could create novel landscape behaviors. It is also difficult to determine how the critical plant and animal life cycle processes of reproduction, growth, and mortality will respond to changing climate (Gworek et al. 2007; Ibáñez et al. 2007; Keane et al. 2001; Lambrecht et al. 2007). These modeling uncertainties greatly increase as projections are made further into the future and at finer spatial scales (Xu et al. 2009).

Managers must account for these uncertainties when using the information in this report in any land management plan or analysis. Sometimes there is less uncertainty in implementing conventional restoration designs than in designing restoration or treatment plans that attempt to account for climate change impacts. For some areas or resources, such as the restoration of western larch ecosystems, addressing climate change in management plans may require only minor changes to current management practices. In other situations, major changes to current treatment designs may be needed, such as in ponderosa pine ecosystems. All climate effects will be manifested in different ways on different landscapes, and as a result, there is no magic "one size fits all" prescription that can be adopted everywhere. The decision to modify management actions to include climate change effects must always include an assessment of the uncertainty of that modification and, most importantly, local conditions.

Climate Change Assessment Techniques

Anticipating ongoing rapid climate change, ecologists are attempting to project the effects of those changes on myriad ecosystem processes across various scales (Clark et al. 2001; Joyce et al. 2014; Schumacher et al. 2006). Using traditional ecological field methods to explore climate change response may be difficult because of the complex interactions among ecological processes, disturbance, and climate at multiple temporal and spatial scales (McKenzie et al. 2014). It would simply be too costly and time-intensive to sample at the large spatial scales and long timespans need to quantify vegetation response (Keane and Finney 2003).

In general, there are four techniques to assess and project the effects of climate change on vegetation and other resource concerns. The first is *expert opinion,* and it involves having experts in the fields of climate change, ecology, and vegetation dynamics qualitatively assess what will happen to vegetation under various climate change scenarios. Most of the papers about climate change effects on vegetation used in this report were written by experts who have evaluated future climate projections and used their experience to deduce how vegetation will respond to different climates. Information from these papers was included in this report, but expert opinions were used only when there was no other information from the other assessment techniques.

The second technique is *field assessment*, where extensive field sampling or remote sensing projects monitor vegetation change in response to changing climate. Field sampling involves establishing plots in networks across the landscape, detecting change between plot measurements, and correlating these changes to climate data. Van Mantgem and Stephenson (2007), for example, related high tree mortalities to climate change using a network of monitoring plots. Demography studies track individuals over time, rather than using periodic plot-level inventories, to fully understand the role of climate relative to other risk factors such as competition, variation in physiology and function, and vulnerability to insects and pathogens. Such demography datasets are rare (Iverson and McKenzie 2013), but one study has tracked more than 27,000 individuals of 40 species for about a decade to address interactions over an area of the southeastern United States (Clark et al. 2011). The only demographic dataset available for the Northern Rockies region is the USFS Forest Inventory and Analysis (FIA) database. The extensive FIA dataset has been summarized to describe vegetation shifts due to climate change elsewhere (McNulty et al. 1994) but not in the Northern Rockies region. Although field assessment techniques are the most reliable and most useful, they are often intractable (see previous paragraphs) because of the large areas and long time periods needed to properly sample vegetation at the appropriate scales to detect changes as a result of climate.

The third method involves the use of statistical analysis to create empirical models that project climate change response. Most of the studies that project the habitat, range, or occupational shifts of Northern Rockies tree species from climate warming use species distribution models (SDMs) to project future geographic ranges (Hansen and Phillips 2015; Iverson and Prasad 2002; Warwell et al. 2007). SDMs, also called bioclimatic envelope models, niche models, and species envelope models, are developed by linking current climate with the current distribution of a species of interest by means of advanced statistical modeling (Guisan and Zimmermann 2000; Watling et al. 2012). Then, using the statistical model, a future species distribution is computed using projected future climate data as inputs rather than the past climate. However, SDMs are inherently flawed for projecting future species distributions in that they relate past species occurrence to climate, resulting in predictions of potential species habitat, not species distribution (Iverson and McKenzie 2013). The projected potential habitat is only reflective of the distribution of species today and does not relate climate to historical distributions. One of the biggest limitations to this approach is that many studies have now found that most species distributions are not in equilibrium with climate, thereby causing SDMs to miss those areas conducive to occupation by the species but where the species is currently absent. Moritz and Agudo (2013), for example, found many species in the fossil record existed over a wider range of climates than is recorded today.

Another limitation of empirical models is that the critical ecological processes, such as pollination, cone production, seed dispersal, seed germination, seedling establishment, tree growth, mycorrhizae influences, competitive interactions, disturbance, mutualism, and mortality, as well as the many disturbance processes, are not represented in SDMs, yet these are the main processes that control species abundance and presence (Iverson and McKenzie 2013; Watling et al. 2012). Dullinger et al. (2012), for example, found that range shifts predicted by SDMs retracted by more than 40 percent when seed dispersal was included in the prediction process. Girardin et al. (2008) found that process models were much better at projecting climate change effects on tree growth because they accounted for changes in soil moisture and growing season. Moreover, the climates used to develop SDMs represent a very small slice of time (50–100 years) relative to the long time periods that existing trees, such as the long-lived whitebark pine (>1,000 years of age), have survived on the landscape today, so SDMs cannot capture the climate for all stages in the life cycle of today's trees. Most mature trees used to evaluate species occurrence in statistical models may have lived for hundreds to thousands of years and continued to survive despite major changes in climate. Along those same lines, one of the major problems of SDM modeling is that there is no sense of how long it will take for a species to be eliminated from one site and effectively populate a new site; because migration is a slow process, the timing of SDM model results are often incompatible with management timeframes. In addition, SDMs assume that the current distribution of the species is a consequence of climate alone, yet we know that fire exclusion, exotic diseases, and management actions have also influenced species occurrence (Gustafson 2013; Iverson and McKenzie 2013). Therefore, it is difficult to have confidence in SDM projections for fine-scale applications; they are informative, but not prognostic, especially on the short time scales of decades and half-centuries required by land management. This is especially true when addressing the high uncertainty of the GCM-derived climate used by the SDMs.

The last and perhaps the most effective technique uses simulation modeling to assess climate-mediated vegetation responses (Gustafson 2013; Iverson and McKenzie 2013; McKenzie et al. 2014). Here, future projections of climate are used as inputs to simple-to-complex ecological models to simulate the climate change effects (Baker 1989; He et al. 2008; Merriam et al. 1992; Perry and Millington 2008). A variety of existing models simulate ecological change at broad (global, regional) and fine (ecosystem, stand, point) scales (Bugmann 2001; Cramer et al. 2001). However, landscape-scale (40–400 square miles) models are perhaps the most critical for predicting effects of climate change because this is a key scale at which most ecosystem processes and links are manifested and the scale at which most management decisions are made (Cushman et al. 2007; Littell et al. 2011; McKenzie et al. 2014). Finer-scale stand models cannot incorporate important exogenous disturbance regimes because of their limited spatial extent, and coarse-scale dynamic global vegetation models (DGVMs) are unable to simulate important plant-, species- and canopy-level competition and disturbance effects, such as successional shifts, community dynamics, and differential disturbance effects among species (McKenzie et al. 2014).

To realistically model species composition changes, a mechanistic, process-driven simulation approach might be preferable to emphasize those physical drivers of vegetation dynamics that are directly related to climate (Gustafson 2013). However, mechanistic model design is often overly complex and therefore currently intractable because of (1) overly detailed parameterization of life histories and physiologies for all species, (2) high complexity of many interacting disturbance factors, and (3) necessarily high-resolution modeling over large areas (Lawler et al. 2006). Dynamic global vegetation models, such as the MC2 model used in this report (see next subsection), operate at scales from regional (hundreds of miles) to global (degrees of latitude and longitude). Although DGVMs are valuable for projections of climate change across large domains, these models aggregate species into lifeforms or plant functional types (PFTs) using structural or functional attributes, which may be useless to local managers (Bachelet et al. 2003; Bonan 2008; Neilson et al. 2005). Most of these models project shifts to more drought-tolerant and disturbance-tolerant species or PFTs for future climates. This general shift in vegetation may be offset by physiological changes induced by CO_2 fertilization, as suggested by a DGVM (MC1) that links water use efficiency to CO_2-simulated expansion of forests into areas where the climate is currently too dry (Bachelet et al. 2003). This issue deserves further study to resolve the extent and duration of such mitigating effects of CO_2; projected effects could differ substantially depending on how relationships are modeled.

To be effective at realistically predicting climate change effects, ecosystem models must simulate disturbances, vegetation, and climate, and also their interactions across multiple scales (Purves and Pacala 2008). Yet few models simulate ecosystem processes with the mechanistic detail needed to realistically represent important interactions among landscape processes, vegetation dynamics, disturbance regimes, and climate (Keane et al. 2015b; Riggs et al. 2015). Direct interactions between climate and vegetation, for example, may be more realistically represented by simulating daily carbon (photosynthesis, respiration), water (evapotranspiration), and nutrient (nitrogen, phosphorus) dynamics at the plant level than by simulating vegetation development annually using state-and-transition modeling approaches (Keane et al. 2015a). A fully mechanistic approach, however, may be difficult for both conceptual and computational reasons, and some simulated processes may always require a stochastic or empirical approach (Falk et al. 2007; McKenzie et al. 2014).

We used output from the DGVM MC2 to standardize our evaluation of change and vegetation responses for the Northern Rockies region. Output from this model is presented in Appendix 6A, and this output was used by all authors in developing the material on future climate effects on vegetation and in the vulnerability assessment. We did not use MC2 simulated species projections in the following sections.

MC2 Model

MC2 is a new implementation of the MC1 DGVM, which was created to assess the impacts of global climate change on ecosystem structure and function at a wide range of spatial scales from landscape to global (Bachelet et al. 2001; Peterman et al. 2014). MC2 is short for "MC1 version 2." MC1 was produced by combining physiologically based biogeographic rules, originally defined in the Mapped Atmosphere-Plant-Soil System (MAPSS) model (Neilson 1995), with biogeochemical processes packaged in a modified version of CENTURY (Parton et al. 1987) and a new fire disturbance model, MCFIRE (Lenihan et al. 1998, 2003). The three linked modules simulate biogeography (lifeform interpreter and vegetation classifier), biogeochemistry, and fire. The main functions of the biogeographic module are to (1) predict lifeforms, that is, the composition of deciduous-evergreen tree and C3-C4 grass lifeform mixtures; and (2) classify those lifeforms and their associated biomass into different vegetation classes using a climatologic rule base. The biogeochemical module simulates monthly carbon and nutrient dynamics for a given ecosystem. Aboveground and belowground processes are modeled in detail and include plant production, soil organic matter decomposition, and water and nutrient cycling. Parameterization of this module is based on the lifeform composition of the ecosystems, which is updated annually by the biogeographic module. The fire module simulates the occurrence, behavior, and effects of severe fire. Allometric equations, keyed to the lifeform composition supplied by the biogeographic module, are used to convert aboveground biomass to fuel classes. Fire effects, specifically plant mortality and live and dead biomass consumption, are estimated as a function of simulated fire behavior (fire spread and fire line intensity) and vegetation structure. Fire effects feed back to the biogeochemical module to adjust levels of various carbon and nutrient pools to alter vegetation structure (e.g., leaf area index levels and woody vs. grass-dominated vegetation).

The MC2 model simulations were generated by Tim Sheehan (Conservation Biology Institute). Inputs to the MC2 model include soil (depth, texture, and bulk density), annual average atmospheric CO_2 concentration, and monthly average climate variables (monthly precipitation, mean vapor pressure, and means of daily maximum and minimum temperatures). Historical climate data (1895–2008) were obtained from the PRISM group (Daly et al. 2008) and were upscaled to 30-arc-second resolution (~0.23 square mile). Soils data were derived from STATSGO (Soil Conservation Service 1991) by Kern (1995, 2000) and were scaled to the resolution of the climate data. Future climate projections were available from various GCMs, and we chose the MIROC 3.2 medres (Hasumi and Emori 2004) based on its relatively high overall ranking according to Mote and Salathé (2010). GCM future projections were downscaled to 0.23 square mile using the delta or anomaly method (Fowler et al. 2007). Anomalies between future and mean monthly historical (1971–2000) values were calculated to project estimates for each climate variable and each future month across the study area. We evaluated model output based on two greenhouse gas emissions scenarios described in the IPCC Special Report on Emissions Scenarios (Nakićenović et al. 2000): A1B and A2. Future projections based on the most recent generation of emissions scenarios, the Representative Concentration Pathways, were not available across the entire Northern Rockies region, but the two generations of models are relatively similar in their estimates of global temperature change and spatial patterns of temperature and precipitation change (Knutti and Sedláček 2013).

To evaluate potential climate effects on vegetation assemblages and disturbance regimes and the interaction with land management, we evaluated a suite of vegetation-related and fire occurrence variables output by the MC2 model under historical (1971–2000) conditions and future projections for mid-century and end-of-century. Specifically, we compared past vegetation distributions across the Northern Rockies with fire suppression and without, and made similar comparisons for two future years, 2050 and 2100, under both the A1B and the A2 emissions scenario (Appendix 6A). Aboveground processes were examined by comparing the amount of carbon in live and dead biomass for three time periods (historical, 2030–2050, 2080–2100) and with and without fire suppression (Appendix 6A). Potential evapotranspiration was also evaluated to compare possible changes in aridity (over similar timespans and land management measures). Finally, projected changes in fire disturbance were examined by comparing estimated fire rotation and the percentage of the Northern Rockies burned by time period and suppression management (Appendix 6A).

Forest Vegetation Responses to Climate

In general, many scientists expect the effects of climate change on forest vegetation to be primarily driven by vegetation responses to shifts in disturbance regimes, and then secondarily, through direct effects of vegetation interactions with climate through shifts in regeneration, growth, and mortality processes at both individual plant and community scales (Dale et al. 2001; Flannigan et al. 2009; Temperli et al. 2013). Most of the expected climate changes are reduced precipitation and increased temperatures (see chapter 3), resulting in a reduction in water available to trees and understory plants. These effects will be highly variable across time, from year to year and day to day, and across space, as the footprint of the new climate is manifested at fine to coarse scales. Trees will respond to projected reduced water availability, higher temperatures, and changes in growing season in diverse manners, but because trees cannot pick up their roots and move, any changes in vegetation composition and structure will be the result of changes in both the life cycle processes and responses of a plant to disturbance. This section discusses some possible general responses of trees and forest vegetation to projected climates.

Individual Plant

The effects of climate on forest vegetation can occur as both direct and indirect effects. Direct effects are the immediate and long-term impacts of increased temperature and decreasing water availability on vegetation life cycle processes, as discussed in detail throughout this document. But indirect effects, such as changes to fire, insect, and disease regimes, may be more important and long-lasting than direct effects.

In short, there are several important modes of response of plants to changing climates (Joyce and Birdsey 2000). The first is changes in *productivity*; plant productivity may increase in the future because of increasing temperatures, longer growing seasons, more variable precipitation, and CO_2 fertilization (Aston 2010; Joyce 1995). Increases and decreases in productivity are related to changes in cone crops, tree vigor, and tree defenses. The window of successful seedling *establishment* will change (Ibáñez et al. 2007); increasing drought and high temperatures may narrow the time for effective regeneration in low- elevation Northern Rockies forests and widen the window in high elevation forests. Climate may directly cause tree *mortality* due to temperature or moisture stress on trees; there have been increases in tree mortality around the world from increasing temperatures and drought (Allen et al. 2010; Williams et al. 2010). This of course is related to productivity, but not entirely. Extreme climate events, such as late growing-season frosts and high winds causing blowdowns, may increase because of the predicted increases in climate variability (Notaro 2008), and these events may cause *mortality* events (Joyce et al. 2014). There will also be disruptions in *phenology* as climates change; many plants may sustain considerable damage or mortality as phenological cues and events are mistimed with new climates (e.g., flowering occurring during dry portions of the growing season) (Cayan et al. 2001). Another related mode is the *genetic* limitation of the species or tree to respond to climate change (Hamrick 2004); specialists may become maladapted to new climates (St. Clair and Howe 2007). Last, plants can respond to climate-mediated changes in *disturbance* in myriad ways (Aitken et al. 2008). This section deals only with those causal mechanisms that drive direct climate responses; the indirect climate-mediated disturbances and responses are detailed in a later section.

Direct effects of temperature on plant growth may increase both photosynthesis and respiration (Waring and Running 1998). Plant photosynthesis rates increase with temperature up to an optimum and then decline thereafter, with the optimum being species-dependent. If projected temperatures exceed the photosynthetic optima for Northern Rockies tree species, such as those in the lower elevation forests, then plant growth might suffer. However, there may be many portions of the Northern Rockies where temperature increases probably will not exceed optima, and there may be photosynthetic gains, such as in montane and subalpine areas. This, of course, depends on whether sufficient water is available to support increased photosynthesis. Respiration also increases with temperature; thus, photosynthetic gains may be lost through growth and maintenance respiration. Respiration occurs even when stomata are closed, so high temperatures coupled with low water availability may result in high respirational losses with few photosynthetic gains (Ryan et al. 1995).

Increased atmospheric CO_2 levels may also directly modify ecophysiological growth processes. Oxygen and CO_2 compete for active Rubisco (primary enzyme used in photosynthesis) sites. Higher atmospheric CO_2 concentrations may increase internal leaf CO_2 concentrations, thereby ensuring CO_2 reaches most of the Rubisco sites, which can result in photosynthetic increases of 2 to 250 percent depending on site and species (Ehleringer and Cerling 1995). Conifers may also have increased water use efficiency in future water-limited environments, and increased water use efficiency may compensate for decreases in water availability and increase growth rates in water-rich environments (Waring and Running 1998). Water use efficiency is the ratio of water used for plant metabolism (photosynthesis and respiration) to the water lost to transpiration. With higher CO_2 concentrations in the atmosphere, the plant would obtain more CO_2 during the time the stomata are open, resulting in less loss through transpiration. Leaf biomass is usually the first to increase as plants attempt to optimize photosynthesis by growing more photosynthetically active tissue (i.e., more leaf area). However, increases in leaf area index are often transitory and greatly dependent on available nitrogen and water. Increases in leaf area might also result in greater rainfall interception, higher snow collection, and greater canopy evaporation. Increased atmospheric CO_2 levels and increasing temperatures can also interact to increase growth. Photosynthesis has temperature optima that differ by tree species, and warmer temperatures might be closer to the new temperature optima, especially during the cooler early growing season, perhaps resulting in faster growth.

Another major direct effect of warming temperatures is longer growing seasons (Cayan et al. 2001; McKenzie et al. 2008); that is, increases in temperatures often lengthen growing seasons for forest plants. There are concerns that future climates are projected to be highly variable, and the coupling of highly variable daily weather with highly variable growing seasons may increase the chances that plants will be more susceptible to adverse weather during fragile phenological stages (Hanninen 1995). Warm conditions in the early spring, for example, might stimulate bud burst and early growth, only to have these expanding tissues frozen by subsequent frost events. Plant phenological cues may be disrupted or triggered inappropriately because of high weather variability, and while this variability might result in minor damage for mature individuals, it may be fatal for seedlings. This may be especially true in localized frost pockets and narrow valleys that accumulate cold air, resulting in frequent frost during the early growing season. Warmer temperatures may reduce and perhaps eliminate growing season frosts in mountain valleys, thereby allowing more frost-susceptible species, such as ponderosa pine and western larch, to exist in traditional lodgepole pine, subalpine fir (*Abies lasiocarpa*), and Engelmann spruce (*Picea engelmannii*) habitats. Chmura et al. (2011) note that increased temperatures may result in decreased winter chilling that could result in delayed bud burst, reduced flowering, and lower seed germination. Winter dormancy prevents trees from growth flushes during warm winter periods, and future climates may trigger changes in winter dormancy and subject trees to high mortality during those cold snaps after the winter warming.

Snowpack dynamics are also directly influenced by changes in temperature and precipitation and declining snowpacks are expected under future climates (Mote et al. 2005). Most of the water used by Northern Rockies trees usually comes from snowmelt (Waring and Running 1998), so the amount and duration of snowpack have the potential to influence regeneration and growth patterns of forest communities throughout the region. Warming temperatures may cause earlier snowmelt, leading to an earlier start of the growing season. However, earlier snowmelt could also result in longer periods of low soil water during the remaining part of the growing season, effectively shortening the growing season. Earlier snowmelt may also result in greater competition for water across the plants and species that make up the complex plant communities of the Northern Rockies. Plants with roots in the topmost layers of the soil might be able to more effectively capture the rainfall that used to occur as snowfall. This may favor grasses and some forbs over shrubs and trees (Daly et al. 2000). The lower snowpack may allow longer growing seasons in those subalpine and upper subalpine communities where cold and snowpack duration govern tree regeneration and growth; production and regeneration are likely to increase with decreasing snowpacks, especially in those high mountain environments where water is rarely limiting.

An indirect effect of climate change will be the shift in distribution of microsites that facilitate tree regeneration (Jones 2013). While effects of climate change at coarser scales mostly relate to tree growth and mortality, changes to microsite conditions will be likely to govern tree regeneration (Petrie et al. 2016). Microsites suitable for tree regeneration must be addressed in a spatial and temporal context. For example, conditions for tree establishment may be suitable all year on small microsites of up to 1.2 square yards, whereas larger microsites may be conducive to regeneration only during the spring. Researchers in Washington State found that even with major changes in climate, there probably will be ample microsites that are suitable for regeneration of trees (Little et al. 1994). In the Northern Rockies, however, projected climate changes will probably result in smaller and more ephemeral microsites for regeneration. The size, distribution, and duration of suitable microsites potentially will vary more each year and most regeneration might occur only during “wave” years (i.e., years with favorable weather conditions) where plentiful suitable microsites are widely available for long periods of time (see next subsection).

Climate change can also indirectly affect vegetation by altering mycorrhizae dynamics (Amaranthus et al. 1999). Nearly all Northern Rockies conifers depend on mycorrhizae for enhanced water use and nutrient absorption. Even whitebark pine, a species that lives in areas with the highest precipitation in the Northern Rockies, has a mutualistic relationship with several species of fungi (Mohatt et al. 2008). Many trees, particularly those in the seedling and sapling stages, need mycorrhizae to survive, especially in areas of periodically severe water shortage (Walker et al. 1995). The migration of Northern Rockies tree species to more favorable sites in future climates may be entirely governed by the ability of the mycorrhizae to also populate these areas to allow or facilitate tree species establishment (Lankau et al. 2015). Allison and Treseder (2008) found warming increased fungal interactions, but drying caused significant decreases. Without viable populations of mycorrhizae, tree species movement might be significantly slowed or stopped. New microsite conditions created by predicted future climates may be inhospitable to mycorrhizae, but so little is known about how these fungi shift with climate that it is difficult to evaluate how they will respond to climate change (Fitter et al. 2000). Mycorrhizae responses to climate change after increased fire may be more important; fire may reduce the fungi in some areas. This may be especially true if the large, severe fires projected for the future actually occur (Stephens et al. 2014). Severe fires may kill all trees in a large burn, thereby eliminating the host for the mycorrhizae, and perhaps eventually the mycorrhizae themselves. Establishment of trees into these burned areas can be delayed for long periods, decades or even centuries, as mycorrhizae and trees revegetate the area together (Schowalter et al. 1997).

Perhaps the most important future indirect ecological concern is the role of today's forest conditions and how they affect the ability of tree species to respond to future climate change. Ample research has shown that past and future human land use activities often result in ecological impacts that overwhelm any direct or indirect climate change effects (Moritz and Agudo 2013). Nearly a hundred years of fire suppression activities have resulted in increased tree regeneration and denser forest canopies, coupled with increases in duff, litter, and down dead woody fuels (Arno 1998; Ferry et al. 1995; Keane et al. 2002). Trees in these dense forests are in greater competition for the little water that is available for growth. As a result, trees in many fire-excluded stands are stressed, making them highly susceptible to mortality from secondary stressors, such as insect and disease outbreaks (Anderegg et al. 2012; Wikars and Schimmel 2001), drought (Allen et al. 2010), and fire (Hood et al. 2007). Increased tree densities may also foster increased severity of subsequent disturbances, resulting in more individuals dying and creating larger patches of mortality.

Another ecological concern closely related to fire exclusion is the current climate-mediated decline in forest communities that have recently become established as a result of fire exclusion. Some forests in the region, including the limber pine (*Pinus flexilis*) communities along the Rocky Mountain front, became established during the fire exclusion era but may now be declining in some portions of the Northern Rockies region because of increased drought and nonative disease (white pine blister rust). Other Northern Rockies forests, such as ponderosa pine, now have atypical forest compositions and structures due to the century of fire exclusion, and these now denser forests are stressed from both overcrowding and climate change (Millar et al. 2007b). Had fires been allowed to burn, there would probably be significantly less mortality, from either climate change or disturbance, compared to what we are experiencing today, and the mortality levels probably would not be projected to be as high in the future (Holsinger et al. 2014). Moreover, there are areas in the region where trees have encroached into dry grasslands (Arno and Gruell 1986), montane meadows, and subalpine forb fields (Butler 1986). Now, due to increased temperatures and decreased water availability, some of these recently established trees are dying. Examples include the limber pine expansion along the Rocky Mountain front (Taylor and Sturdevant 1998), Douglas-fir encroachment into dry prairie (Arno and Gruell 1986), and subalpine fir encroachment into GYA forb fields (Bigler et al. 2005). There is concern as to whether climate change represents a threat to these modern forest types, which were probably rare in the historical record. Does increased vulnerability in 100-year-old limber pine forests really constitute a management concern? Or is the increase in mortality expected because the forests established in wet periods of the fire exclusion era?

Many forest species in the Northern Rockies region may respond to direct climate warming and drying by expanding their range into new habitats. Migrating to a new site has historically been the primary response of plants to climate change (Huntley 1991). Migration requires that the species be able to quickly occupy newly desirable sites of the future before other species get there or to outcompete other species once they are there. Neilson et al. (2005) detail four basic components of migration (moving to a new site): fecundity, dispersal, establishment, and growth. To be successful in the future, a species must produce enough seeds or propagules (*fecundity*) that are easily dispersed to new sites (*dispersal*), and the seedlings that become established on these sites must be able to grow (*establishment*) so that they can also produce ample propagules that are then dispersed even further (*growth*). As Davis et al. (2005) note, however, the species also must have the genetic capacity to migrate to the new climate and survive into maturity. Adapting in situ can take many forms, such as reducing leaf area to minimize transpiration losses, or surviving perturbations in the new disturbance regimes created by climate change. Most Northern Rockies tree species are long-lived and genetically plastic so that they can survive the wide fluctuations of weather in the future, but the ability to handle both deep drought and modified disturbance regimes may be the most important factor dictating future distributions of forest species (Allen et al. 2010).

It is widely thought that warming climates will result in upward shifts in the elevational distribution of plant species. For example, Lenoir et al. (2008) found that some plant species have moved upward in elevation at a rate of 95 feet per decade. However, it is unclear whether such elevational shifts will drive long-term changes in forest communities, or if other predominating forces will outweigh these upward trends. For example, fire plays a dominant role in most Northern Rockies ecosystems, determining landscape structure and processes. Because even more wildfires are expected as climates change, effects of these large events may overwhelm any shifts in distributional ranges of forest species resulting from climate warming. Further, most plants in the region have slow migration rates, mostly because they are adapted to fire and as such rely more on regenerative organs (e.g., sprouting) than seed dispersal. Finally, implications of an upward elevational migration on forest communities need to be considered within a temporal and spatial context. That is, it may take a century or two for tree species to demonstrate significant elevational shifts due to long life cycles, old maturation ages, highly variable weather, and low dispersal potentials. The potential for tree species to migrate may be entirely different in each of the unique mountain ranges in the region, depending on a host of abiotic and biotic factors (e.g., precipitation levels, invasive species) and available colonizing species.

Most projections for the response of vegetation to climate shifts are for populations of species, not for communities. Little is known about how composition and abundance of biota will change at a community level in response to climate shifts. Will new plant communities be dominated by generalist species that can exist across a wide variety of biophysical settings? Or will future communities be similar to historical analogs where fire-adapted species dominate? Will future communities be composed of species collections that were historically rare? Answers to these questions have important implications for future land management in that there is a great deal of synergy between plants and species in historical communities, such as interacting via root-grafting, sharing mycorrhizae, and relying on common pollinators, and future community composition may not have as many interactions. Moreover, future communities may not be as diverse because they may be dominated by a limited suite of species.

Climate change can affect important phases of the life cycle processes of plants: reproduction, regeneration, growth, and mortality. Moreover, it can affect plants at various scales from the needle to the tree to the forest, and over seconds to days to years to centuries (Eamus and Jarvis 1989). The following subsections detail possible climate change effects by life cycle processes.

Reproduction

Cone and seed crops for many Northern Rockies trees could be both adversely and beneficially affected by climate change (Ibáñez et al. 2007; LaDeau and Clark 2001). Low-elevation xeric forests might have fewer and smaller cone crops because of increased stand density and water stress. Cone crops might also have a lower percentage of viable seed because of increased tree stress. The infrequency of cone crops coupled with low seed numbers may result in the lack of regeneration in recently burned areas, thereby causing a shift to nonforest vegetation.

The opposite might be true in higher, colder environments where increased temperatures will increase growing season length and thereby increase potential for more cone crops with greater number of seeds. Spruce-fir communities might produce so much seed that they may overwhelm regeneration of other conifers, especially after mixed-severity fires. Subalpine pine species such as whitebark and lodgepole pine have unique cone characteristics (whitebark pine cones facilitate seed dispersal by birds, whereas lodgepole pine cones may be serotinous and opened only by fire), so they may need to rely on disturbance for increased cone abundance.

An indirect result of the interaction of fire, vegetation, and climate is that as fire becomes frequent, some species, primarily trees, may be killed by fire before they reach reproductive maturity and may fail to set cones. Holsinger et al. (2014), for example, found that fires were projected to be so frequent in a western Montana watershed that lodgepole pine seedlings would be killed by fire before they were reproductively mature (around 15 years). Keane et al. (1990) found that ponderosa pine forests needed occasional interfire periods to be greater than 35 years to allow pine seedlings to grow above the lethal scorch height. If fire is too frequent, plants will not be able to grow to reproductive maturity or the reproductive organs might always be scorched by fire.

Climate warming and increased variability will also affect the phenology of cone crops, but effects may be minimal as plants adapt to new conditions. Some predict higher frost mortality of emerging cones due to an earlier onset of the growing season coupled with high daily temperature variability and lower flowering and seed germination because chilling requirements will not be met (Chmura et al. 2011). Others suggest that frequency and abundance of cone crops will be reduced in the future because of high tree stress from drought (Ibáñez et al. 2007). However, the increased productivity projected for many Northern Rockies forests may overwhelm minor losses from extreme weather events over the long run.

With changing climate, some tree species might be excluded from their current range because warmer temperatures may not allow chilling requirements for the seed (Shafer et al. 2001). The chilling requirement was a major evaluation factor in determining climate change vulnerability in Devine et al. (2012). Similarly, new climates may be asynchronous with the phenology of many tree species. Seed dispersal, for example, may occur at the driest and warmest times. Phenological keys may be out of sync in new climates, especially in a highly variable future, resulting in reduced flowering, growth, and reproduction. On the other hand, these phenological miscues may also occur in disturbance agents; highly variable weather may result in occasional deep frosts that kill beetle larvae, for example.

Regeneration

The life cycle phase in which most tree species are vulnerable to climate is regeneration (Solomon and West 1993). Most tree species in the region reproduce by producing seeds that fall to the ground to germinate and grow into seedlings that then become mature trees. Microsite conditions needed for successful establishment are so demanding that seed germination and survival, especially for seeds that are wind dispersed, are rarely successful (Anderson and Winterton 1996). The successfully germinated seed produces a fragile radicle (embryonic root) that must penetrate the litter, then duff, then soil to put down a root system that will eventually feed the growing aboveground tissue. This penetration process demands moist soil conditions or the radicle and associated cotyledon (developing leaves) and hypocotyl (stem) will dry and die. To become a seedling, the seed requires suitable moisture conditions for long periods of time. Because few seeds become seedlings, many tree species often rely on high seed production to overcome mesic site conditions to ensure successful regeneration; of the millions of seed produced, perhaps at least some will land on moist microsites suitable for establishment. For dry xeric forests, most of the successful regeneration occurs in those wet years when soils are moist for a suitable time and solar insulation does not kill developing leaves and stems. These moist years are often called wave years, and the pulses of regeneration that occur in these years results in even-aged patches. Projected climate change is likely to decrease the frequency of these wave years, and on the driest sites, the frequency of wave years may be so low that no regeneration may occur, depending on the species. Planting on these newly dry sites may also be ineffective because of the short window of high soil moisture.

At the highest elevations, where the depth and duration of snow cover often governs tree regeneration, warming may enhance regeneration. Most years are moist enough for regeneration at high elevations, but snow remains on sites too long for successful regeneration in many years. With warming temperatures, snow is likely to melt earlier, giving more time for seedlings to survive and grow. Previous warm wave years in upper subalpine ecosystems are often dated by using seedling and sapling ages. Recent observations of invasions of subalpine meadows and balds by subalpine fir, alpine larch (*Larix lyallii*), and Engelmann spruce attest to a high number of sequential warm years over the last decade, which have facilitated regeneration in the high-mountain landscape (Butler 1986). Therefore, climate warming is expected to enhance regeneration at the subalpine and upper subalpine forest ecosystems.

Future climates and their high variability may also affect the ability of forest species to successfully germinate. Seed chilling requirements may not be met during mild winters, thereby reducing germination, and germination could be delayed until the driest parts of the growing season. Nitschke and Innes (2008) found that the chilling requirements were not being met for most low-elevation tree species in British Columbia. Soil temperatures may be too high, causing greater mortality of both germinants and established seedlings (Rochefort et al. 1994).

Climate change may also affect the dispersal properties of the reproductive propagules. Rodents that disperse seeds of ponderosa and western white pine, for example, may migrate or decline because of warmer, drier habitat conditions. Whitebark pine is dispersed by the Clark's nutcracker (*Nucifraga columbiana*), which might shift habitats because of climate-mediated changes; nutcrackers usually nest in high elevation areas with ample snowpack (Tomback 1998), and these nesting habitats are predicted to decline in the future (Westerling et al. 2006). Longer and drier summers and falls also mean that seed dispersal may take place when the ground and litter are the driest and least hospitable for seed germination and establishment (Neilson et al. 2005). Human- and ungulate-mediated seed dispersal of exotic species could also be different in future climates; warmer, drier climates might reduce human and ungulate use to lower exotic seed dispersal. Changes in landscape spatial heterogeneity may also influence mechanisms of nonwind seed dispersal by shifting potential seed sources and changing patch sizes.

Growth and Mortality

Productivity potentially could increase in some Northern Rockies forests with warming climate, resulting in increased vigor and more resistance to stressors (Joyce 1995). Worldwide, Lin et al. (2010) compute increases in biomass of more than 12 percent (20 percent in forests) with climate warming. However, Chmura et al. (2011) note that even with increased productivity, most forests will undergo reduced growth and survival as the climate interacts with the entire tree species life cycle.

Climate can adversely influence growth and mortality in many ways (Bugmann and Cramer 1998; Keane et al. 2001). Projected decreases in water availability may result in shorter effective growing seasons and longer periods of continuous drought in the drier Northern Rockies forests (Williams et al. 2010). Longer drought might require Northern Rockies conifers to close stomata longer to conserve the little water available. Some xeric conifers, such as ponderosa pine and limber pine, have excellent stomatal control and are able to remain closed for long periods of time. Other conifers, such as Douglas-fir, have poor stomatal control, and this may drive leaf water potentials to extremely low values, which might result in intercellular cavitation, tissue damage, and perhaps plant mortality (Sala et al. 2005). The projected increased temperatures will increase both maintenance and growth respiration, especially when stomata are closed. Increased respiration will require additional photosynthetic gains to counterbalance respiration losses, thus demanding even more water in a drier future. If photosynthetic production cannot exceed respiration demands, then the plant becomes stressed, thereby increasing the probability of mortality and susceptibility to insects and disease.

In the most mesic and montane ecosystems, a warming climate is likely to enhance growth and decrease mortality. Wu et al. (2011) found increases in plant growth for many forest and rangeland ecosystems with warming worldwide. Earlier growing seasons with ample moisture, such as that predicted for mesic montane forests in the Northern Rockies, will probably lead to increased productivity and greater growth. Although this increased biomass could result in additional foliar material to increase canopy bulk density and therefore result in higher crown fire potential, it could also result in higher growth rates for timber production and forage. This will be especially true in the higher mountain environments where cold temperatures, not moisture, limit tree growth. Longer, warmer growing seasons might result in higher productivities and greater biomass. The increased biomass will also increase competitive interactions between species, thereby favoring the more shade-tolerant individuals in the absence of disturbance. However, increased biomass could foster more-intense fires, and maybe greater insect and disease outbreaks, such that the more disturbance-tolerant species might ultimately inherit the landscape.

Genetics Concerns

It is widely accepted that climate limits species distributions. Climate is also a major environmental factor affecting plant phenotypes and a critical agent of natural selection, molding among-population genetic variation. Plant adaptations to local environments have often developed a clinal or continuous response to abiotic and biotic factors such as temperature, frost-free periods, precipitation, fire, insects, and disease. More recently, ecotypic or a discontinuous response to environmental gradients is being recognized based on different soil or edaphic properties. The combination of clinal and ecotypic environmental gradients across the landscape enhances or limits plant survival and long-term persistence.

The hardiness of a plant is determined by its genetic background. Ecological genetics is a field of study investigating the genetic architecture, phenotypic plasticity (ability of an organism to change its phenotype in response to changes in the environment), and adaptive capacity of a species in the context of interactions among and between plant populations and environmental gradients. Ecological genetics and common garden studies are employed to study individual species. Well-designed common garden studies provide information on the adaptive strategy of a species (e.g., generalist, intermediate, or specialist; table 6.1) (Rehfeldt 1994). Processes that shape the genetic architecture of a species include natural selection, migration, genetic drift, and its mating system. Thus, the ability of plant populations to respond to climate change is influenced by the underlying patterns of genetic variation.

Molecular markers can reveal significant genetic diversity and divergence among populations associated with variation among populations (table 6.1). Past historical events affecting divergence among populations can be shaped by a variety of factors. Examples of abiotic factors are fire, glaciation (Hamrick 2004), and volcanic activity; for instance, range shifts east of the Cascades indicated ponderosa pine was replaced with lodgepole pine, and later repopulated by ponderosa pine after the Pleistocene (Hansen 1942, 1947, 1949). Other factors include abiotic and biotic seed dispersal agents (for whitebark pine, limber pine, and ponderosa pine) (Lorenz and Sullivan 2009) and pollinator history. Plants that are insect-pollinated or rely on animal-dispersed seed are more vulnerable to climate change because of the requirement for interaction with another organism.

Genetic diversity enables a species to adapt to changing environments, colonize new areas, occupy new ecological niches (USDA FS 2006), and produce substantial and robust progeny that persist in the long term (Ledig and Kitzmiller 1992). The entire species does not adapt to environmental change over time, but populations within a species do. Species and populations of plants most vulnerable to climate change are rare species, genetic specialists, species with limited phenotypic plasticity, species or populations with low genetic variation, populations with low dispersal or colonization potential, populations at the trailing edge of climate change, populations at the upper elevational limit of their distribution, and populations threatened by habitat loss, fire, disease, or insects (Spittlehouse and Stewart 2004; St. Clair and Howe 2011). The underlying assumption about forest and rangeland species is that as climate continues to change, populations will become poorly adapted to their local climates, thus becoming stressed. But the ability of a species to respond to environmental change is closely tied to its adaptive strategy and the mechanisms that shape its genetic structure; therefore, this assumption may be false. Some species such as Douglas-fir, juniper (*Juniperus* spp.), and sagebrush (*Artemisia* spp.) may show range expansion in the future (Hansen and Phillips 2015).

Historical gene flow (seed and pollen movement) creates patterns of genetic differentiation that may allow some populations to be more predisposed to respond to climate change than others. Fragmentation is a critical issue for plant populations because isolation and the occurrence of a relatively few number of individuals can lead to inbreeding and loss of genetic diversity (Broadhurst et al. 2008; Potter et al. 2015). This field of study also informs research and management of the adaptive capacity and vulnerability to climate change (i.e., its direction and magnitude) of each species. Gene flow from adjacent populations that are more typical of future climates has the ability to increase the rate of adaptation by introducing genetic variation that is pre-adapted to warmer or drier climates (Aitken et al. 2008). A practical application of this field of study facilitates evaluating options for responding to environmental gradients and climate change, for example, choice of the appropriate

Table 6.1—Comparison of attributes characterizing a species' adaptive strategy.[a]

	Adaptive strategy	
Attributes	**Specialist**	**Generalist**
Factor controlling phenotypic expression of adaptive traits	Genotype	Environment
Mechanisms for accommodating environmental heterogeneity	Genetic variation	Phenotypic plasticity
Range of environments where physiological processes function optimally	Small	Large
Slope of clines for adaptive traits	Steep	Flat
Partitioning of genetic variation in adaptive traits	Largely among populations	Largely within populations

[a] Modified after Rehfeldt (1994).

population or seed source to increase the likelihood of attaining a desired reforestation, restoration, or revegetation outcome.

Most species may not be able to adapt quickly enough to keep pace with projected migration rates of 328 to 3,280 feet per year with climate change (Davis 1989; Malcolm et al. 2002). Davis and Shaw (2001) and Davis et al. (2005) suggest plant adaptation may be a more important factor in response to climate change due to the slow rates of plant migration impeded by population fragmentation as a result of land use patterns. Although Hamrick et al. (1992) and Hamrick (2004) suggest that long-lived species with high levels of genetic variation are well positioned for climate change, Etterson and Shaw (2001), Jump and Peñuelas (2005), and Parmesan (2006) argue that the ability of forest trees to adapt or migrate and follow climatic shifts may be *restricted* by their long lifespans, long generation intervals, and long juvenile phases.

Long-lived species often maintain high levels of genetic variation and gene flow, which facilitates their ability to evolve in response to changing climates (Hamrick 2004; Hamrick et al. 1992). Whitebark pine is an example of a long-lived species with high levels of genetic variation (Mahalovich and Hipkins 2011) and extensive gene flow (Richardson et al. 2002) attributed both to long-distance seed caching by Clark's nutcracker and an outcrossed mating system involving wind pollination (Richardson et al. 2002). Because plant populations are genetically adapted to local climates, the climatic tolerance of individual populations is often considerably narrower than the tolerance of the entire species.

Knowledge of the adaptation of Northern Rockies plant species is well documented for conifers (Rehfeldt 1994) but incomplete or lacking for other native plants. A species does not necessarily have only one adaptive strategy, though most do. Differences in adaptive strategy can be characterized by differences in variety (e.g., *P. ponderosa* var. *ponderosa* or Rocky Mountain ponderosa pine [*P. ponderosa* var. *scopulorum*]), elevation, and geography. For example, *P. ponderosa* var. *ponderosa* is characterized as having an intermediate adaptive strategy; however, at high elevations (>5,000 feet), ponderosa pine has a specialist adaptive strategy. Rocky Mountain Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) is characterized as having a specialist adaptive strategy; that is, its genetic variation is organized into numerous local populations, finely tuned to site-specific gradients. At higher elevations east of the Continental Divide, however, Douglas-fir has a generalist adaptive strategy; its genetic variation is organized into one or a few populations capable of surviving, growing, and reproducing over a broad range of environments (Rehfeldt 1989). Species possessing a generalist adaptive strategy are proposed to fare better than their intermediate and specialist counterparts in changing climate.

Patterns of adaptive variation for other native plants (e.g., shrubs, forbs, grasses, and sedges) are more complex, being both clinal and ecotypic. These species differ in lifeform (e.g., annual, biennial, and perennial) and ploidy level (number of copies of DNA, such as 4X, 6X, or 8X). The base ploidy level is 2X, where one copy of DNA is inherited on both the maternal and paternal sides. Grasses are hypothesized to be largely generalists and less vulnerable to climate change; however, ecotypic variation can overlay the generalist adaptive strategy. Forbs, which are largely insect-pollinated, are more vulnerable to the changes in phenology and longer growing seasons expected with climate change.

Soil Responses

Each soil in the Northern Rockies region has an inherent ability to produce vegetation based on climate, parent material, topography, soil biology, and soil development (Armson 1977). Soil supports production of vegetation through interactions of nutrient cycling, soil hydrology, soil biology, physical support, and filtering (or buffering) (Attiwill and Leeper 1987). The quality and quantity of soil organic matter, the timing and amount of moisture, temperature, and acidity may all be altered by climate change, which will ultimately affect functional properties of soils and perhaps productivity (Bonan 2008).

Climate change affects the growth, mortality, and decomposition of vegetation, which in turn influence soil biology (Waring and Running 1998). Warmer temperatures, increased CO_2, and longer growing season contribute to higher vegetative growth. Warmer temperatures, increased drought, and greater susceptibility to insects and disease may lead to increased mortality. Although higher temperatures will increase decomposition rates, the moisture required for decomposition may increase or decrease, leading to variable changes in decomposition rates (Davidson and Janssens 2006). Decomposition will increase with a combination of warmer temperature and higher moisture, whereas decomposition will decrease if summer droughts extend later (Rustad et al. 2000). Increased fire frequency and severity would generally reduce soil organic matter across large landscapes (Dooley and Treseder 2012).

Higher air temperatures will directly increase soil temperature. Increased vegetative cover would provide dense shade, thus decreasing soil temperature, whereas decreased vegetative cover would result in more heating at the soil surface. Dry soil, which is expected to be more common during future drought, would have wider temperature fluctuations than wet soil, which is buffered by the high heat capacity of water. In addition, if snow cover is lower but extreme cold periods continue to occur, soils will have lower minimum temperatures (Davidson and Janssens 2006).

The vulnerability of soils to future climate change is summarized in table 6.2. Effects will differ greatly, depending on local soil characteristics, the magnitude and trend of climate change, and vegetation response.

Stressors—Biotic and Abiotic Disturbances

A warming climate will rarely be the direct agent of change for Northern Rockies tree species and communities. Most of the changes in vegetation are likely to result from

Table 6.2—Assessment for soils to apply to vegetation concerns in the Northern Rockies.

Broad-scale climate change effect	Ecosystem function	Current condition, existing stressors	Sensitivity to climatic variability and change	Expected effects of climate change	Risk assessment	
					Magnitude of effects	Likelihood of effects
Mainly changes in summer precipitation Higher summer temperatures	Available water (for vegetation)	Relative effective annual precipitation has been mapped for MT. Current condition has not been assessed for ID and ND. Existing stressors: extremes in water year (more frequent droughts), high stand densities.	Variable, depending on species, subregion and slope position.	In western Northern Rockies, decreased precipitation is projected at low elevations; increased summer drought may result. High elevations may have increased precipitation. In eastern Northern Rockies, the higher interannual variation of precipitation would lead to years of increased drought. No trend projected for annual or seasonal precipitation.	High	High
Increased atmospheric CO_2	Nutrient cycling for vegetation production	Variable. Nitrogen is the most common nutrient deficiency, but other nutrients (phosphorus, potassium, boron) are also limiting. Existing stressors: severe fires, erosion.	Unknown. Potassium is easily lost on some metasediments, during whole-tree yarding.	Nitrogen, phosphorus, potassium and boron will become less available if soils become more acidic. Increased acidity of leachate would contribute to faster parent material decomposition, which would release inherent mineral nutrients in soil.	Unknown	High
Soil temperature, moisture, and CO_2 levels	Soil biota	Most soils have healthy soil biota. Exceptions occur on abandoned roads and landings. Some stands have reduced biotic health from past management. Existing stressors: severe fires, erosion	Moderate	Alteration of soil biota (fungi, microbes, nematodes) as well as the communities.	Variable	Very high

Table 6.2(cont.)—Assessment for soils to apply to vegetation concerns in the Northern Rockies.

Broad-scale climate change effect	Ecosystem function	Current condition, existing stressors	Sensitivity to climatic variability and change	Expected effects of climate change	Risk assessment	
					Magnitude of effects	Likelihood of effects
Everything that contributes to increased fire and flood (e.g., more frequent high-intensity storms)	Soil stability	Little surface erosion occurs in the Northern Rockies Existing stressors: Most erosion occurs after severe wildfires or floods. Small amounts may occur in conjunction with roads, timber sales, grazing, and other land management activities.	High—existing stressors will become more prevalent.	Areas with decreased ground cover will be susceptible to increased surface erosion from wind and water.	Moderate	Very high
Changes in spring and autumn precipitation (rain rather than snow, period of saturation longer)	Slope stability	Landslide prone areas have been mapped. Existing stressor: long period of saturation in some years.	Variable	Size and frequency of landslides will increase.	Broad	Very high
Temperature, precipitation and CO_2 concentrations	Soil carbon storage and dynamics	Soil is currently a major carbon storage pool and will continue to be a carbon sink as long as young stands continue to mature. Current stressors: periodic wildfires and insect outbreaks.	High in areas where disturbances become more prevalent.	If wildfire increases in frequency and extent as much as projected, this will maintain forests in younger age classes with less potential for carbon storage but higher carbon uptake.	Potentially high in drier forests	High in dry forests

responses to climate change-induced disturbance or to some combination of other climate-exacerbated stressors (Keane et al. 2015a). Climate change has marginally to severely altered disturbance regimes in the western United States (Liu et al. 2011). As we consider past climate variability and then add the projections in temperature and precipitation, there may be significant changes occurring across Northern Rockies forests because of a changing water balance and the role of disturbances such as wildfires, insects, and diseases. Whether it is invasive species (e.g., white pine blister rust; causal agent: *Cronartium ribicola*), drought, uncharacteristic wildfires, elevated native insect and disease levels, loss of historically fire-adapted tree species, unusually high forest densities compared to historical conditions, or some other combination of disturbance agents that serves to stress trees and forest ecosystems, recent research suggests that climate change is likely to further exacerbate those stressors and "stress complexes" (Iverson and McKenzie 2013). The following subsections present a short summary on four major classes of stressors important in the region. More-detailed summaries of disturbance responses and their interactions to climate change are presented in Chapter 8 of this report.

Wildland Fire

Wildland fire is pervasive throughout Northern Rockies forest ecosystems and was historically the dominant landscape disturbance in the region (Baker 2009; Barrows et al. 1977; Wellner 1970). Fire exclusion since the 1920s has disrupted annual occurrence, spatial extent, and cumulative area burned by wildfires. Climate change impacts to fire regimes are overlaid on a century of ecological changes to forest vegetation and fuels; thus, observed differences between current fire patterns and historical ones are a product of management legacies as well as anthropogenic changes to climate.

Wildland fire regimes, defined by fire frequency, annual area burned, severity, and pattern, are greatly influenced by variability in landscape environmental conditions including vegetation distribution, climate, weather, and topography (McKenzie et al. 2011). Climate and fuels are the two most important factors controlling patterns of fire within forest ecosystems. Climate controls the frequency of weather conditions that promote fire, whereas the amount and arrangement of fuels influence fire intensity and spread. These wildland fuels—the live and dead biomass that burns in fires—lose moisture and become flammable in the region's typically warm and dry summers, during which there are ample sources of ignition from lightning strikes and humans. Therefore, the active fire season (period conducive to active burning) is in the summer, typically from late June through October, with shorter seasons at higher elevation sites where snowpack can persist well into July. Regionally, widespread fire years are correlated with drought (Heyerdahl et al. 2008). At large spatial scales, topography can influence the spatial pattern of fire spread. For example, in dissected mountainous areas, topographic features (e.g., barren slopes) can form barriers to fire spread (Grissino-Mayer et al. 2004), but where drainages are aligned with prevailing winds, topography can facilitate the spread of large fires (Sharples 2009).

Compositions and structures of forests in the Northern Rockies region are strongly determined by fire history. In general, fire regimes vary along environmental gradients, with fire frequency decreasing and fire severity increasing with elevation. For example, at the lowest and driest elevations, where forests are dominated by ponderosa pine, frequent surface fires historically consumed litter and dead wood and killed seedlings and smaller trees. Adaptive traits such as thick bark allowed mature ponderosa pines to survive many repeated fires over time and tree densities were kept low. Fire exclusion since the 1920s has increased surface fuel loads, tree densities, and ladder fuels, especially in low-elevation dry conifer forests (Schoennagel et al. 2004). As a result, fires at the lowest and driest elevations may be larger and more intense, and may cause higher rates of tree mortality, than historical fire. But in mid- and higher elevation forests, where fires were historically infrequent because of relatively cold, wet conditions, fire exclusion has not affected the fire regimes (Romme and Despain 1989; Schoennagel et al. 2004). However, earlier onset of snowmelt, predicted to occur with changing regional climate, will reduce fuel moisture during fire season, making mid- to high-elevation forested systems flammable for longer periods of time (Miller et al. 2009). As these forested systems are not fuel-limited, fire occurrence and extent are likely to increase in the future (Littell et al. 2009, 2010; Westerling et al. 2006).

Insect Outbreaks

Regional insect activity and outbreaks are highly correlated with climate drivers, and potential climate change-induced insect activity will be an important influence on future forest composition and structure. The mountain pine beetle (*Dendroctonus ponderosae*) is an integral component of forest ecosystem processes because of its role in stand thinning and redistribution of resources and nutrients important for tree regeneration. It is also recognized as an aggressive and economically important forest insect responsible for tree mortality across large areas (Logan et al. 2003). Both bark beetle populations and their host trees are being influenced by changing climate. Many bark beetle life history traits that influence population success are temperature-dependent (Bentz and Jönsson 2015), and warming temperatures associated with climate change have directly influenced bark beetle-caused tree mortality in some areas of western North America (Safranyik et al. 2010; Weed et al. 2015). Host tree distribution across the Northern Rockies region, and tree vigor, which influences susceptibility to bark beetle attack (Chapman et al. 2012; Hart et al. 2013), will also be influenced as climate continues to change. Future bark beetle-caused tree mortality will therefore depend not only on the spatial distribution of live host trees and heterogeneity of future landscapes, as described in this chapter, but also on the ability of beetle populations and their associates to adapt to changing conditions when existing phenotypic plasticity is surpassed.

Pathogens

Forest diseases are found in all forest ecosystems of the Northern Rockies region. They are one of three major disturbance groups that affect ecosystem development and change, yet impacts of forest diseases on various resources and services in the region are difficult to estimate. The major groups of forest diseases in the region that affect ecosystems and ecosystem services are fungi and rusts (fungi that infect needles and causes damage and mortality, the most important being white pine blister rust), dwarf mistletoes (*Arceuthobium* spp.; a group of parasitic seed plants that are widespread across the region and that mostly cause reduced tree growth and productivity), root diseases (a major cause of growth loss and mortality), needle casts and blights (diseases that cause crown thinning and loss of lower branches), and abiotic diseases (damage to trees resulting from impacts of adverse environmental factors on tree physiology or structure).

Effects of climate changes on forest diseases are difficult to predict. Climate change can alter pathogens through direct effects on the development and survival of the pathogen, physiological changes in tree defenses, or indirect effects on the abundance of natural enemies, mutualists, and competitors (Ayres and Lombardero 2000). These dynamics are not well captured by GCMs because the ecology and impacts of pathogens are based on local site and environmental conditions. Epidemics also depend on local conditions for spread and infection to occur. Although models usually generate mean climatic conditions, it is often the extremes that have the greatest influence on pest conditions (Hepting 1963), and these are also not well represented by GCMs. However, modeling efforts to date suggest that among the major Northern Rockies diseases, root disease is projected to cause the highest basal area loss as a percentage of total basal area in the region. Projected losses from root diseases ranged from zero percent on most national forests east of the Continental Divide to 15–19 percent on westside forests (Krist et al. 2014). Klopfenstein et al. (2009) used a subset of GCMs to predict how the geographic distribution of the climate envelope for Armillaria root rot (*Armillaria solidipes,* formerly *A. ostoyae*) and Douglas-fir could change in the interior northwestern United States. Their analysis suggests that Douglas-fir will have a considerably smaller geographic space that matches its current climate envelope and that this space will shift, while only minor changes are projected for *A. solidipes*. They suggest that areas where Douglas-fir is maladapted could increase, which could increase its susceptibility to Armillaria root rot. Climate-mediated changes to forest tree diseases will be dictated by disease and host responses to new climates, and their interactions (Sturrock et al. 2010); the interactions among biotic diseases, abiotic stressors, and host status will drive future pathogen outbreaks. Predicted increases in temperature and drought will probably serve to increase pathogen populations in the future (Chakraborty et al. 2008). The role of pathogens as important disturbance agents is likely to increase in the future because they are able to migrate to new environments at a faster rate than trees.

Drought

Soil type and depth, aspect, and elevation all contribute to effective moisture availability for tree establishment and growth, producing patterns of forests in the Northern Rockies region. Additionally, the impact of stand condition on overall water balance and the effect of site and soil conditions on moisture availability are important to consider. The Natural Resources Conservation Service (NRCS) and the Natural Resource Information System of the Montana State Library have mapped the relative effective annual precipitation (REAP) for the State of Montana (Montana State Library and NRCS n.d.). REAP is an indicator of the amount of moisture available at a location, taking into account precipitation, slope and aspect, and soil properties. For example, two sites that receive the same amount of precipitation may have different effective precipitation due to unique soil and landform factors at each site. Depending on the geographic location within Montana and degree of slope, the actual precipitation for southerly aspects may be adjusted downward while northerly aspects may be adjusted upward.

Future climate change models indicate that the Northern Rockies region will have longer, drier summers and warmer conditions. Pioneer (seral) species such as ponderosa pine have the unique ability to establish on bare soil surfaces where high surface temperatures (>149 °F) exclude other species. One of the adaptations of these seral species is a capability for deep rooting, which allows the tree to find an adequate water supply and avoid extensive competition with shallow- and fibrous-rooted grasses and forbs. As the shade from these species limits sun-loving grasses and forbs, shade-tolerant tree species establish and grow. Grass and forb species usually have a shallower rooting characteristic that allows them to gather soil water from the nutrient-rich soil surface; in contrast, the overall rooting structure of shade-tolerant tree species in essence becomes much more competitive as succession progresses. In addition, the overall leaf surface area that develops over time on a given site increases. Lands dominated by grasses/forbs or shrubs usually develop a maximum total leaf area of about 3.3 square feet per square foot of soil surface area. Forests can develop leaf areas in excess of 6.5 square feet per square foot of soil surface area. With increasing leaf area comes increased water transpiration, which can deplete the soil water storage capacity needed to keep trees hydrated throughout the summer. The additional canopy interception of rain and snow in dense forests, which directly evaporates into the atmosphere, further compounds this effect, reducing soil water recharge. The result is a water-stressed forest that not only becomes more susceptible to insects and disease, but also more prone to supporting severe wildfires because live fuel moisture is relatively low.

Climate Change Assessments

This section contains the information that was used to assess vulnerability for all tree species, vegetation types, and resources of concern. There are four subsections for each item (e.g., tree species) to detail the (1) ecology, (2) disturbance interactions, (3) current and historical conditions, and (4) potential climate change responses. The first subsection presents important ecological information needed to understand how a species, type, or resource of concern might respond to future changes in climates, such as its drought- and shade-tolerance. The subsection on disturbance interactions contains information on those agents that affect the species, type, or concern, and important projections of how those disturbance agents might change in the future. Historical and current conditions are included as a subsection because any climate change response is greatly dependent on current status and past actions. Last, the anticipated climate change responses for the species, types, and concerns are included in perhaps the most important subsection. This material was ultimately the basis for evaluations of vulnerability or development of potential adaptation actions.

Most of the material in this section was taken from the literature, but substantial amounts of anecdotal and observational information were also included for context. However, due to imperfect knowledge across the evaluated entities and the high uncertainty in climate predictions and ecosystem responses, we admit that many of our projected climate change responses and resultant vulnerability assessments are based on our own professional experience. Moreover, some climate change response material may appear uneven across species, types, and concerns because detailed information is not available for all of them; for example, more information is available for timber tree species than nontimber species.

Tree Species

Most of the background information used in this subsection was synthesized from three primary sources. The Bollenbacher (2012) report presents characteristics of the major tree species of the Northern Rockies region, adapted from the autecological synthesis developed by Minore (1979). The commonly used silviculture reference edited by Burns and Honkala (1990) was used throughout, and the climate change report compiled by Devine et al. (2012) for the Pacific Northwest was also used for genetics and autecological information. Table 6.3 provides a general summary of ecological and genetic characteristics by tree species that will be important under future climate change. In this subsection, we attempted to integrate the genetic, morphological, ecological, and disturbance response characteristics summarized in table 6.3 to predict how a tree species would respond under future climate warming. We also integrated any material available in the literature to aid and support our predictions.

Discussion on climate change responses was synthesized from information in the literature and the MC2 modeling results (Appendix 6A) to evaluate the effects of climate change on important species, vegetation types, and resource concerns. This material forms the foundation for our vulnerability assessments and the adaptation strategies and tactics. Many of these climate change responses are based on the species characteristics and current ecosystem condition presented in this section.

The most astonishing finding in this section is that the literature is inconsistent on the response of tree species to future climate change. Results from SDM modeling are often, but not always, different from most other sources that include gap modeling, mechanistic ecosystem simulation, and field data summaries. As a result, we put less emphasis on the SDM results in our vulnerability assessment evaluations in Appendix 6B. Another finding is that the amount of climate change really matters. Most climate change studies predict few species changes after moderate warming (e.g., the B1, B2, A1B, and RCP 4.5 scenarios), but major species shifts under the most extreme emissions scenarios (e.g., the A1 and RCP 8.5 scenarios). Third, the timeframe used in the climate change study is also important. Management timeframes of 10 to 50 years are not long enough to effectively evaluate changes in fire, beetles, and tree growth. Ecosystem response to disturbance takes time, often two to five times the disturbance return interval. Last, climate change study results and subsequent ecosystem responses depend tremendously on the choice of GCMs used to simulate and quantify climate change. Some GCMs predict minor warming for the Northern Rockies region, while others predict major changes.

Based on a thorough review of the literature, we propose three basic modes of response to climate change for the major tree species of the region: modification, contraction, and expansion. First, the species could increase or decrease in productivity in situ within its current range due to increasing temperatures and adequate precipitation (*acclimatization*); for example, the majority of information seems to support the inference that most lands in the Northern Rockies region will increase in productivity (Aston 2010). Next, the species could die in those parts of its range where conditions will change enough to become inhospitable to that species (Allen et al. 2010) (*contraction*). Last, the species could migrate to areas that are more conducive to establishment and growth (Johnstone and Chapin 2003) (*expansion*). Any species can have multiple modes of response to climate change, and most species will respond to future climates via all three modes.

Application of these three modes to determine future species dynamics demands a thorough integration of variability and scale. For example, the ebb and flow of species migration demands a relatively long temporal scope to properly evaluate species range shifts (Prentice et al. 1991). A tree species could become established in a "new" environment made suitable by climate change, such as subalpine tree expansion into snow glades, but the great variability in climate may result in 1 year of drought or high snow that kills all

Table 6.3—Summary of tree characteristics that indicate possible responses to future climate change. All fields are rated on a scale from one to five, with 1 generally indicating low levels and five generally indicating high levels of the variable (see specific notes below).

Species	Distribution[a]	Tolerances[b]			Genetics			Mycorrhizal dependence[f]	Insect and disease exposure[g]	Seed dissemination strategy[h]
		Shade	Drought	Fire	Diversity[c]	Geographic differentiation[d]	Strategy[e]			
Limber pine	1	1	5	2	3	3	I	4	4	5,4
Ponderosa pine–var. scopulorum	3	2	4	4	3	2	I	3	2	3,4
Ponderosa pine-var. ponderosa	3	2	4	5	3	2	I,S	3	4	3,4
Douglas-fir	4	4	4	4	2	2	S,G	3	3	3
Western larch	2	2	2	5	3	2	I	2	2	3
Western white pine	1	2	2	5	3	3	G	2	4	3,4
Aspen	2	1	4	1	5	2	G	2	3	3,2
Grand fir	2	4	2	2	2	2	G	2	4	3
Western redcedar	2	5	5	2	1	2	S	1	2	3
Western hemlock	2	5	5	2	2	2	S	2	2	3
Lodgepole pine	4	2	3	3	4	2	S	3	4	6
Whitebark pine	3	3	2	3	4	3	G	3	4	5
Subalpine fir	4	4	4	1	3	2	I	2	3	3
Engelmann spruce	3	4	4	2	3	2	I	2	2	3
Mountain hemlock	1	4	5	1	2	2	S	2	2	3
Alpine larch	1	1	5	1	2	3	S	3	1	3
Green ash	2	1	2	1	2	3	G	4	1	3
Cottonwood	2	1	4	3	3	3	G	3	2	1

[a] Distribution of 1 indicates narrow distribution in the Northern Rockies, while distribution of 5 indicates wide distribution.

[b] Shade, drought, and fire tolerance of 1 indicates high intolerance, while 5 indicates high tolerance.

[c] Genetic diversity of 1 indicates low diversity and 5 indicates high diversity.

[d] Geographic differentiation of 1 indicates low genetic differentiation (narrow seed zones) and 5 indicates wide differentiation (large seed zones).

[e] Genetic adaptive strategy of G indicates a generalist species (grows under wide range of conditions), I indicates an intermediate species (grows over a moderate range of conditions), and S indicates a specialist species (grows under a narrow range of conditions) (from Scott et al. 2013).

[f] Mycorrhizal dependence of 1 indicates that the species is not dependent on mycorrhizae, while 5 indicates that the species is highly dependent on mycorrhizae.

[g] Insect/disease exposure of 1 indicates that the species is not susceptible to many damaging agents, while 5 indicates that the species is highly susceptible to insects and disease.

[h] Seed dissemination is the primary means of seed dispersal where 1-water, 2-sprouts, 3-wind/gravity, 4-rodent, 5-bird, 6-serotinous.

established seedlings. Conversely, 1 year of drought could kill many individuals in the grassland-woodland ecotone, but several wet years in a row might facilitate reestablishment of tree species into the high mortality zone. Further, the rate of climate change shifts will be governed by disturbance, not competition, so disturbance adaptations will be more important than climatic niches. Management actions, such as fire exclusion, may facilitate species expansion into areas that will eventually burn, causing extensive mortality.

All of the climate change response evaluations in this chapter have a high level of uncertainty; they are essentially best guesses from a wide variety of resource specialists and a review of the literature. The following information may provide a starting place, a possible prioritization, or assistance in addressing climate change in forest plans, but it is in no way accurate enough to provide valid predictions of what will happen in the future.

Limber Pine

Autecology

Limber pine (*Pinus flexilis*) is a shade-intolerant, early seral to pioneer species in the Northern Rockies (Steele 1990). Its seeds are dispersed by rodents, but more importantly, by a bird (Clark's nutcracker) that will cache limber pine seed anywhere there is microsite pattern that it uses for finding the seed (Lanner 1980; Lanner and Vander Wall 1980). Limber pine has difficulty in competing with other encroaching species on more productive mesic sites and is often succeeded by Douglas-fir and subalpine fir. There is often little to no reproduction once tree densities are below 10 trees per acre, mostly because of the lack of an effective pollination cloud, and those seeds that are produced have increased likelihood of inbreeding. Moreover, a minimum of 10 cone-bearing trees per acre is needed for dispersal by Clark's nutcracker (McKinney et al. 2009). This tree species is very slow growing but long-lived, and some of the oldest trees in the region are limber pine.

Limber pine is a puzzling species in the context of ecosystem land management. It occupies xeric sites across a wide range of elevations (2,600 to 8,900 feet in elevation) in the Northern Rockies region that are often marginal for timber production (Jackson et al. 2010). Historically, it was often found on the margins between grasslands and forest ecosystems at the lower treeline on fire refugia (Steele 1990). Because limber pine is easily killed by fire, the species was mostly found in fire-protected cove sites where fire was rare and of low severity, such as rocky outcrops, barren areas, and moist north slopes (Steele 1990). In these lower treeline areas, limber pine is often associated with Douglas-fir, Rocky Mountain ponderosa pine, and quaking aspen (*Populus tremuloides*). On upland montane sites, it can often be found on limestone substrates and droughty soils, but in these areas it is associated with many other Northern Rockies conifers, especially lodgepole pine, subalpine fir, and Engelmann spruce (Langor 2007; Steele 1990). Limber pine seedlings are poor competitors with grass, but do well on rocky substrates and in shrub environments.

Limber pine is very tolerant of drought and can establish and grow in some of the most arid environments in the Northern Rockies region (Steele 1990) (table 6.3). It is associated with both ectomycorrhizae and arbuscular mycorrhizae that facilitate its ability to exist in extremely dry environments. Seedlings are very drought tolerant but have a low tolerance to competition, especially from herbaceous plants.

Genetically, limber pine has high outcrossing rates with average genetic diversity and average population differentiation (Devine et al. 2012). The fundamental and realized niche for limber pine is very broad in the region, indicating that this species has a generalist adaptive strategy with wide phenotypic plasticity.

Disturbance Interactions

As mentioned, the thin bark and low foliage of limber pine make the species highly susceptible to damage from wildland fire. Limber pine is also highly susceptible to white pine blister rust, and many communities suffer high mortality when the disease infects trees in a new region (Smith et al. 2013). Limber pine also facilitates the expansion of currant (*Ribes* spp.; an alternate host for the pathogen *Cronartium ribicola*) into traditional grasslands (Baumeister and Callaway 2006), thus increasing rust infections and mortality.

Other insects and pathogens are also impacting limber pine, but at a severity much lower than *C. ribicola*. Some researchers have detected mortality from mountain pine beetle in parts of the limber pine range (Jackson et al. 2010). Others have noted that limber pine stands on mesic sites may have severe dwarf mistletoe infections that could result in mortality levels similar to those observed from white pine blister rust. Porcupine (*Erethizon dorsatum*) damage is also prevalent east of the Continental Divide.

Historical and Current Conditions

With fire exclusion, limber pine has expanded its range from fire-protected cove sites into areas where it was historically restricted by frequent fires (Arno and Gruell 1983; Brown and Schoettle 2008). As a result of the diminished fire activity and active nutcracker caching, limber pine has expanded into grass and shrub rangelands, and this expansion has also allowed other species to inhabit historically nonforest areas (Jackson et al. 2010). Evidence suggests that limber pine can facilitate the establishment of other forest species, especially Douglas-fir, in rangeland settings (Baumeister and Callaway 2006). As a result, limber pine in the Northern Rockies region is currently occupying areas that were traditionally grasslands, and it is difficult to determine if this is inside or outside the range of variability of this ecosystem.

Ironically, the newly established limber pine forests throughout the Northern Rockies region are undergoing dramatic declines due to white pine blister rust, mountain pine beetle, and wind-caused red belt (Jackson et al. 2010; Langor 2007; Taylor and Sturdevant 1998). Increasing fires are also burning some of the stands that have become established since 1910. There is some white pine blister rust resistance in the species, but it is low, perhaps lower than 1 in 100 individuals (Steele 1990).

Climate Change Responses

Some anticipate that warming temperatures on the east side of the region, along with increasing but more-variable precipitation, especially during the growing season, and waning snowpack will result in increased growth in many limber pine communities (Aston 2010). Increases in vigor are usually accompanied by larger cone crops, higher seed viability, greater number of seeds per cone, wider seed dispersal, and greater resistance to disease. Increased seed dispersal includes denser caching by birds and mammals, and probably more distant caching by Clark's nutcracker. Increases in vigor might also extend to competitors of limber pine, so there could be increased competition from wind-dispersed conifers, especially on the more mesic portions of the limber pine range.

Warm temperatures, even with increased precipitation, could also result in drier conditions, especially for seed germination and seedling growth. Even if more seeds are cached by mammals and birds, the subsequent establishment of seedlings from the unclaimed caches might be low because of longer drought seasons and hotter ground temperatures. Any dispersal of limber pine seed to new areas, especially nonforested stands, might have limited regeneration success because of the lack of ectomycorrhizal associations and increased competition from grasses and dense shrubs (Coop and Schoettle 2009).

Disturbance interactions with warming climates are likely to be important to future limber pine dynamics. Increasing fire frequency and intensity may result in the burning of more limber pine stands, causing higher mortality (Coop and Schoettle 2009). Increased fire may stem the encroachment of limber pine into grasslands in areas where grazing is low. Warmer, drier conditions may also reduce blister rust infection by disrupting the blister rust cycle, especially during the late summer when *Ribes* species-to-pine infection occurs, and there may be fewer wave years where temperature and humidity are optimal for pine infection by white pine blister rust. Where precipitation is projected to increase, such as in the eastern portions of the Northern Rockies region, there may be higher rates of blister rust and dwarf mistletoe infection, which may cause higher limber pine mortality. Continued fire exclusion could enhance establishment of currant under mature limber pines and result in even greater white pine blister rust infection and mortality. Warmer temperatures also favor expansion of alternate host species such as currant, lousewort (*Pedicularis* spp.) and Indian paintbrush (*Castilleja* spp.) (Keane et al. 2015a).

Limber pine has an intermediate genetic adaptive strategy under changing climates largely driven by timing of pollen cloud dispersal (elevational effect) and seed dispersal by birds (Feldman et al. 1999). The species is highly adapted to populating the increasing burned areas projected for the future because of mammal- and corvid-mediated dispersal (Lanner and Vander Wall 1980). If future fires are larger and more severe, there may be less competition from other competing conifers, especially in the eastern portions of the Northern Rockies region along the timber-grassland ecotone. Limber pine has moderate genetic variation (capacity) in blister rust resistance, but major gene resistance to blister rust has not been identified in several studies of interior populations. There is probably little to no opportunity to hybridize with western white pine due to non-overlapping species distributions, and it will probably not hybridize with whitebark pine because the two species overlap only on limestone substrates. There is a high risk of loss of disjunct and isolated populations due to genetic drift, ineffective pollen cloud, and limited substrate availability.

Given all available information, limber pine responses to future climates may be minor and governed mostly by wildland fire and white pine blister rust. If fires increase, limber pine forests, some of which are already declining from rust, will suffer major declines, especially where they have encroached as a result of fire exclusion. Given its minor role in the Northern Rockies region prior to European settlement, we consider this species to be at most moderately vulnerable to climate change based on its high tolerance to drought and ability to populate severe environments, but high susceptibility to the introduced white pine blister rust and fire damage may put this species in peril.

Ponderosa Pine

Autecology

Ponderosa pine (*Pinus ponderosa*) shows distinct geographic variation over its range. The ponderosa variety (*P. ponderosa* var. *ponderosa*) ranges from the Fraser River drainage of southern British Columbia south through Washington and Oregon and into northern California (Oliver and Ryker 1990). In the Northern Rockies, it extends from the Canadian border to the central part of Montana on the west side of the Continental Divide. Rocky Mountain ponderosa pine (*P. ponderosa* var. *scopulorum*) extends east of the Continental Divide to North Dakota and South Dakota and south into Wyoming and farther. Within the wide range of both ponderosa pine variants, it is absent from several areas, including a large portion of southwestern Montana. This may be due to the lack of rainfall in the summer months, which prevents establishment except at higher elevations; however, it is also limited by the shorter growing season at these elevations.

In most of western Montana and Idaho, the upper elevational limit of the ponderosa variety is around 4,900 feet, depending on latitude (Pfister et al. 1977). Moisture is the factor most often limiting growth, especially in the summer. Seasonal rainfall deficiency is evident from July and August precipitation (Fowells and Kirk 1945; Tarrant 1953). The distribution of ponderosa pine on drier sites is closely related to supply of available soil moisture, which is closely related to soil texture and depth. Low temperatures, however, may dictate the success of ponderosa pine regeneration; seedlings of the species are highly susceptible to frost damage and the occurrence of frosts often excludes the pine from low valley settings, especially in frost pockets and cold air drainages (Shearer and Schmidt 1970).

Ponderosa pine is a shade-intolerant, drought-adapted species of the low-elevation dry forests of the Northern Rockies (Minore 1979) (table 6.3). It can be a climax species at the lower elevational limits of Northern Rockies coniferous forests, or a seral species in the higher elevation mesic forests, especially the Pacific variety. In dry climax forests, there is generally a mosaic of small even-aged groups. As a seral species, it is often associated with Douglas-fir, lodgepole pine, grand fir, and, in the northwestern Northern Rockies, western larch. Ponderosa pine is mostly intolerant of shade, but it is generally more tolerant than western larch and less tolerant than grand fir and western white pine. Although it reaches its greatest site indices on the mesic grand fir, western redcedar, and western hemlock sites (Cooper et al. 1991), it is rapidly replaced by a suite of more shade-tolerant competitors.

Ponderosa pine is a "drought avoider," meaning it tolerates dry soil conditions by efficiently closing stomata to avoid water loss and xylem cavitation and stay alive during deep droughts (Sala et al. 2005) (table 6.3). This allows the species to tolerate intense drought better than its associates, specifically Douglas-fir, which is a "drought tolerator" and able to obtain water at lower moisture conditions. Although drought tolerators may be able to obtain water at lower moisture conditions, they may attempt to draw groundwater at such low soil water potentials that they experience extreme xylem cavitation, which may cause death. Ponderosa pine has been associated with several species of ectomycorrhizae, giving it a high capacity to survive in dry environments.

Cone crop periodicity varies greatly with ponderosa pine; observations indicate it is a poor seeder west of the Continental Divide and a fair seeder east of the divide. Throughout the region, natural regeneration is sporadic; it is best when there is a heavy seed crop followed by favorable weather during the next growing season (Heidmann 1983; Shearer and Schmidt 1970). Potter et al. (2015) performed molecular work that indicates that Rocky Mountain ponderosa pine is one of the most inbred conifers in the Northern Rockies, and its vulnerability could be further compromised with limited gene flow between populations. With cone crop periodicity or masting events that occur only every 7 to 10 years, increasing natural regeneration problems may be developing on the east side of the Continental Divide. Soil texture, plant competition, and seedbed conditions have the greatest effect on seedling survival. Moisture stress reduces seed germination and limits seedling survival and growth. Competing vegetation deters seedlings. As mentioned, young seedlings (<36 days old) are susceptible to cold night temperatures and deep frosts, and occasionally the pine trees suffer winter desiccation in drying winds. Older seedlings (>110 days) can often withstand higher temperatures than Douglas-fir, grand fir, and Engelmann spruce, making it likely they will be more successful under future climates.

Ponderosa pine has a moderate potential for outcrossing with a high outcrossing rate. It has average genetic variation, but is weakly differentiated geographically. Although it has a strong population differentiation, it may be considered to be intermediate in adaptive strategy because both individuals and populations may be suited to diverse environments. There are steep clines (ecotypes or forms of species that exhibit gradual phenotypic and genetic differences over a geographic area as a result of environmental heterogeneity) in elevation, but gentle clines in latitude and longitude. There is high genetic variation between eastside and westside ponderosa pine in growth, survival, needle length, seasonal pattern of root growth, and ability to germinate under moisture stress (Oliver and Ryker 1990).

Disturbance Interactions

Fires have a profound effect on ponderosa pine where competing tree species are considerably less fire tolerant; this allows ponderosa pine to maintain dominance over large areas (Arno 1988; Steele et al. 1986). Fires historically allowed ponderosa pine to maintain its dominance across most of the low elevation savannas by killing competitors. Ponderosa pine has a great capacity to survive fire, better than nearly all of its competitors (Ryan and Reinhardt 1988).

There are about 108 species of insects that attack westside ponderosa pine and over 59 species that attack eastside ponderosa pine. The most damaging of the tree-killing insects are several species of *Dendroctonus* (Oliver and Ryker 1990). Among bark beetles, *Ips* species are second in destructiveness only to *Dendroctonus*. *Ips* are present naturally in all stands, where they usually breed in slash. Dwarf mistletoe is the most widespread disease on ponderosa pine but is rarely fatal in the region. Western pine shoot borer (*Eucosma sonomana*) is also a concern in the future.

Historical and Current Conditions

Ponderosa pine forests have been undergoing a severe decline due to the combination of logging and fire exclusion. Large pine trees in open pine savannas were harvested from nearly all but the most remote, inaccessible, or protected areas in the Northern Rockies region. Wildland fires have been excluded from remaining pine forests, causing advanced succession that was most rapid in the mesic habitat types (Arno 1988; Gruell et al. 1982). This has resulted in dense forests with overstories of stressed ponderosa pine and dense understories of its shade-tolerant competitors, most commonly Douglas-fir. There are often buildups of duff and litter, and an atypical accumulation of down dead woody fuels on the soil surface. The dense crowns, coupled with high surface fuel loadings, ensure that when these forests are burned by wildfires, the damage from the fire will be severe with high tree mortality, deep soil heating, high fuel consumption, and abundant smoke (Keane et al. 2002).

Climate Change Responses

We expect ponderosa pine in the Northern Rockies region to handle increasing temperatures and deeper, longer droughts with only moderate difficulty. Its ability as a "drought avoider" to close stomata when soil water potential is low makes it the only forest species besides juniper to maintain its presence in many low elevation settings (Stout and Sala 2003). Morales et al. (2015) projected an 11-percent increase in the range of ponderosa pine in the western United States, and Nitschke and Innes (2008), using a gap modeling approach, projected the replacement of dry Douglas-fir dominated communities of British Columbia with ponderosa pine. Hansen et al. (2001) projected an expansion of ponderosa pine across the western United States and specifically in the Pacific Northwest, when most other tree species ranges were retracting in area. Rocky Mountain ponderosa pine is more intermediate in adaptive strategy than the *ponderosa* variety; it therefore has a high phenotypic plasticity and is better adapted to drought (table 6.3).

However, declining precipitation and variable spatial and temporal rainfall patterns may cause declines in ponderosa pine regeneration and range contractions, except in the eastern portions of the Northern Rockies region, where precipitation is expected to increase. Crimmins et al. (2011) estimated that ponderosa pine environments may rise more than 2,300 feet in elevation by 2050 in its range. Similarly, Gray and Hamann (2013) estimated ponderosa pine might move more than 1,600 feet northward and almost 1,000 feet higher in elevation in the Northern Rockies by 2050. However, Franklin et al. (1991) projected future forests of ponderosa pine will cover about a third of its current range in landscapes of the eastern Cascades, and Bell et al. (2014) projected losses of more than 60 percent of its range by 2090.

Increases in mountain pine beetle outbreaks, advancing competition resulting from fire exclusion, western pine shoot borer occurrence, and increases in fire severity and intensity will dictate the future of ponderosa pine in the Northern Rockies. If fires are too frequent, established regeneration will never grow above the lethal scorch height, and mature individuals will not become established. Increasing fire severity and occurrence could also eliminate many of the Northern Rockies relict ponderosa pine trees that provide the critical seed sources for populating future burns.

Douglas-fir

Autecology

Douglas-fir (*Pseudotsuga menziesii*) has been a major component of forests of western North American since the mid-Pleistocene era (Hermann and Lavender 1990). Only Rocky Mountain Douglas-fir (*P. menziesii* var. *glauca*) is found in the Northern Rockies. The range of this variety extends from central British Columbia through the Rocky Mountains into central Mexico. The range is fairly continuous in northern Idaho, western Montana, and northwestern Wyoming, with several outlying areas in east-central Montana and Wyoming. In the Northern Rockies, Douglas-fir grows in areas with maritime influence and mild climate in all seasons except a dry period in July and August. In the central Rocky Mountains, the winters are long and severe, and summers are hot and in some parts very dry. West of the Continental Divide in the region, the rainfall may be evenly divided between winter and summer. The altitudinal distribution of Douglas-fir increases from north to south, due to the effect of climate on the distribution. The limiting factors are temperature in the northern part of the range and moisture to the south. Thus, Douglas-fir prefers southerly slopes in the northern part of its range, and northerly exposures in the southern part of its range (Pfister et al. 1977).

Douglas-fir in the Northern Rockies grows in pure stands on dry, cold sites, in both an even- and uneven-aged condition (Hermann and Lavender 1990). On other sites, the associated species are dependent on the climate, and by proxy, elevation and region. Montane low-elevation mesic Douglas-fir is often associated with western larch, western white pine, grand fir, western redcedar, and western hemlock, whereas on low-elevation xeric sites, Douglas-fir is associated with ponderosa pine, juniper, and quaking aspen. At upper elevational limits, the species is often found with lodgepole pine, subalpine fir, and Engelmann spruce. In rare cases it is found at the highest elevations associated with mountain hemlock (*Tsuga mertensiana*), whitebark pine, and alpine larch. Most of the Northern Rockies Douglas-fir forests are found on droughty sites, and the species is often associated with ponderosa pine; Douglas-fir is often the primary climax species whenever it is found with ponderosa pine (Keane 1985; Ryker and Losensky 1983; Steele and Geier-Hayers 1989). Again, proportion of other species growing with Douglas-fir varies widely depending on aspect, elevation, soil type, and history, particularly fire history, of the area.

Regeneration is most successful where Douglas-fir is seral, especially in the area of strong maritime influence in northern Idaho and western Montana, where it is associated with more montane species (e.g., grand fir, western redcedar, and western larch). Regeneration is poor where it has attained climax status in the cool, dry habitats (Ryker and Losensky 1983). Seedling growth the first year is relatively slow, limited generally by moisture, which triggers initiation of dormancy in midsummer. Competing vegetation may promote the establishment of a variety of seedlings by reducing temperature stress, but may inhibit seedling growth by competing strongly for moisture; this is most pronounced in the southern portion of the range. In the Rocky Mountains, it is a seral species in moist habitats and climax in the warmer, drier areas of its range.

In the interior portion of its Northern Rockies range, Douglas-fir ranks intermediate in shade tolerance, being more tolerant than western larch, ponderosa pine, lodgepole pine, and aspen (table 6.3). Old-growth Douglas-fir shows a wide range of age classes, indicating it became established over long periods after major fires. It is gradually replaced by more tolerant western hemlock, western redcedar, and true fir on mesic montane sites. Douglas-fir tolerates drought better than nearly all of its competitors except for ponderosa pine. The species is a "drought tolerator" in that it keeps stomata open to extract soil water at extremely low soil water potentials, thereby subjecting it to potential xylem cavitation and potential death (Sala et al. 2005; Stout and Sala 2003).

The species exhibits a great deal of genetic differentiation, which is strongly associated with geographic or topographic features (Rehfeldt 1978). The pattern of genetic variation in growth and phenological traits among clines has been observed along north-south, east-west, and elevational transects. There is evidence of low genetic variation within local regions. For example, in southern Oregon, seed collected on the more xeric southerly aspects grew slower, set bud earlier, and had larger roots compared to seedlings grown from north-facing slopes. Seedlings from seed sources on southerly aspects have adaptive characteristics for a shorter growing season and drier soils and may survive under drought stress better than seedlings from seed sources on northerly aspects.

Disturbance Interactions

Douglas-fir has a great capacity to survive fire because of its thick corky bark and its deep main roots. The capacity of the species to form adventitious roots is another adaptation that has enabled Douglas-fir to survive fire. However, young Douglas-fir have thin bark and low height to live crown, greatly increasing mortality from fire (Ryan and Reinhardt 1988). Ponderosa pine and western larch have better ability to survive fire across all life stages, so on sites with frequent fires where Douglas-fir is associated with other species, its cover is usually kept low by fire (Agee 1991). However, on cold, dry sites where the species is the indicated climax, frequent fire may create Douglas-fir savannas, especially east of the Continental Divide, such as in the high valleys of southwestern Montana.

Douglas-fir is subject to serious damage from a variety of agents that may increase under future climates (Hermann and Lavender 1990). Western spruce budworm (*Choristoneura occidentalis*) and Douglas-fir tussock moth (*Orgyia pseudotsugata*) are the most important insects affecting Douglas-fir. Both insects attack trees of all ages periodically throughout the range of interior Douglas-fir, often resulting in severe defoliation of stands. Many Douglas-fir stands in the central Northern Rockies are currently devastated by budworm and beetle. The Douglas-fir beetle (*Dendroctonus pseudotsugae*) is a destructive insect pest in old-growth stands of coastal and interior Douglas-fir. Armillaria and annosus (*Heterobasidion annosum*) root diseases may intensify in infection rate and widen in distribution to cause high tree mortality. Annosus root disease is particularly lethal in Douglas-fir (Hagle 2003). Of the many heart rot fungi (>300 species) attacking Douglas-fir, the most damaging and widespread is red ring rot (*Porodaedalea pini* Murrill, 1905). Knots and scars resulting from fire, lightning, and falling trees are the main paths of infection. Losses from this heart rot far exceed those from any other decay. Other important heart rot fungi in the Northern Rockies are *Fomitopsis officinalis, F. cajanderi*, and *Phaeolus schweinitzii*.

Historical and Current Conditions

Historical frequent wildland fires kept Douglas-fir from becoming established on those dry sites where it was associated with ponderosa pine as frequent fires favored ponderosa pine establishment. It often became established after long interfire periods, such as during the Little Ice Age, and easily attained dominance if fire frequency was decreased. However, in the more montane portions of the Northern Rockies range of the species, Douglas-fir was often one of the major dominants, as it was a major competitor under historical mixed-severity fire regimes (Arno et al. 2000).

Today, though, cumulative effects of the fire exclusion era coupled with logging have allowed Douglas-fir to become the dominant species across its range, especially where it successionally replaced the historically dominant ponderosa pine forests (Arno and Gruell 1983; Arno et al. 2000; Gruell et al. 1982). As a result, we have seen an expansion of Douglas-fir into areas where fire was frequent historically, but also an increase in the density of the forests where it is associated with more mesic species. This has created large, contiguous areas where canopy fuels have increased and become denser, and surface fuels that have been converted from grass and shrubs to heavy down dead woody fuels (Keane et al. 2002). These conditions predispose many Douglas-fir forests to severe future fires. Moreover, these dense stand conditions have contributed to decreased vigor that predisposes the species to western spruce budworm and Douglas-fir beetle outbreaks. Many Douglas-fir forests of southwestern and central Montana are currently experiencing high budworm and beetle mortality.

Climate Change Responses

Several studies suggest that Douglas-fir will respond positively with future changes in climate. Morales et al. (2015) projected a 7-percent increase in the range of the species in the western United States by 2060. Soulé and Knapp (2013) found almost doubled radial growth in Douglas-fir in the western portions of the Northern Rockies in the latter half of the 20th century, but they attributed some of this increase to other factors such as CO_2 fertilization. Rose and Burton (2009), using SDMs, projected that Douglas-fir forests in British Columbia will nearly triple in area by 2080, while Franklin et al. (1991) project no net loss of Douglas-fir habitat in the future in the Pacific Northwest. Using a gap model, Cumming and Burton (1996) also projected little change in the Douglas-fir zone in British Columbia.

However, it is likely that myriad factors will contribute to decline of Douglas-fir forests in some parts of the Northern Rockies region in the future. USFS Northern Region survey results from 2014 show significant increases in Douglas-fir 3-year seedling mortality (about 50 percent) due to increasing drought, high temperatures, and severe conditions, presumably related to climate change. In addition, Kemp (2015) found that natural postfire regeneration of Douglas-fir on sites that burned in 2000 and 2007 varied across gradients in elevation, aspect, and burn severity, and findings indicated that Douglas-fir regeneration was significantly reduced with increased heat loading (incoming solar radiation derived from site latitude, aspect, and slope). Specifically, the probability of successful Douglas-fir regeneration was lower at lower elevation sites and on sites with higher heat load (steep, southwest aspects). Likewise, Douglas-fir abundance was lower on sites at lower elevations and with higher heat loads.

On dry lower elevation southerly aspects in the southern Northern Rockies, ponderosa pine is likely to cope with moisture deficits better than Douglas-fir because it does not have the high potential for xylem cavitation (Stout and Sala 2003). In addition, Douglas-fir might not have the genetic potential to rapidly migrate to more-suitable sites (Aitken et al. 2008). More importantly, a suite of insects and diseases is increasing in Northern Rockies Douglas-fir forests and creating heavy mortality, especially in southwestern portions of the region. The spruce budworm is killing many Douglas-fir stands in southwestern Montana, while the Douglas-fir bark beetle is attacking stands in other parts of the Northern Region. Nitschke and Innes (2008) predict major losses of Douglas-fir from parts of British Columbia because of hot, dry conditions, while Shafer et al. (2001) predict major transitions in Douglas-fir in most of the U.S. Pacific Northwest, and raise some major concerns that the climate might be too warm to meet the chilling requirements of Douglas-fir seed. Using SDM approaches, Gray and Hamann (2013) projected that Douglas-fir will migrate more than 1,300 feet north and 560 feet upwards in elevation by 2050, and Bell et al. (2014) projected losses of more than 40 percent of its range in the Northern Rockies by 2090.

Increases in wildland fires, coupled with adverse effects of the fire exclusion era in Northern Rockies forests, could also present some problems for Douglas-fir. Increasing fire danger in Douglas-fir stands with high canopy and surface fuels may promote wildland fires that kill the majority of Douglas-fir, even the most mature individuals. If fires increase in the future, regardless of fire suppression efforts, they may be so frequent that Douglas-fir seedlings cannot become established and grow to maturity.

Douglas-fir might be one of the Northern Rockies tree species most limited in range expansion because of its limited genetic diversity and structure (St. Clair and Howe 2007). The species has a specialist genetic adaptive strategy at low-to-mid elevations and a more generalist strategy at higher elevations. With warming temperatures and a possible decrease in summer moisture conditions, Rocky Mountain Douglas-fir may contract from the driest portions of its range. Current natural regeneration failures may be exacerbated by reduced seed sources owing to large wildfires and hot and dry microclimate conditions, especially on southerly exposures at lower elevations. On moist sites (mixed mesic forest), mortality from root disease may increase because of increasing moisture stress.

Western Larch

Autecology

Western larch (*Larix occidentalis*) grows in the Upper Columbia River basin of northwestern Montana, and in northern and west-central Idaho (Schmidt and Shearer 1990). It grows in the relatively moist-cool climatic zone. Limiting factors to western larch are low temperatures at the upper elevations, and lack of moisture at the lower extremes (Habeck 1990). Western larch grows on a wide variety of soils; most soils suitable for growth are deep and well drained. It is commonly found on valley bottoms, benches, and northeast-facing mountain slopes (Schmidt et al. 1976).

Western larch is adapted to extreme environmental heterogeneity, from maritime climates in the west and northwest to more continental climates, as westerly air masses move across the Bitterroot and Cabinet Mountains (Rehfeldt 1982). At comparable elevations, the frost-free period in western Montana is 30 days shorter than in northern Idaho, and thus populations from western Montana are better adapted genetically to short frost-free growing seasons as compared to similar elevations in northern Idaho (Rehfeldt 1995a). Moreover, as elevation increases and frost-free periods decrease, growth potential decreases. Early fall cold snaps are a major temperature factor affecting seedling and sapling survival, before resting buds have had an opportunity to fully lignify (Rehfeldt 1995b). Drought is another major climatic factor affecting mid-to-late season survival (Schmidt and Shearer1995). It is most likely to affect seedlings under heavy shade because of the heavy moisture use by the overstory and other competing vegetation. Zhang and Marshall (1994) and Zhang et al. (1994) characterize western larch as having low water use efficiency, as compared to other conifers in the Northern Rockies. Plants that have low water use efficiency tend to be larger in stature and produce more biomass, which may be trait-limiting in future warmer and more arid or variable-precipitation climates. The lower water use efficiency of western larch may explain its absence on xeric sites (Gower et al. 1995).

Cone and seed production in western larch is most prolific at ages older than 30 to 50 years, with seed crops occurring every 14 years in Idaho and every 10 years in Montana (Owens 2008). Good cone crops may occur in successive years if conditions are favorable (Owens and Molder 1979). Spring frosts often reduce pollen, cone, and seed production in western larch, leading to sporadic seed years. Cone production is higher in stands that have larger crowns, such as stands that have been thinned (Shearer 1976). Cooler, wetter springs favor foliar diseases such as larch needle cast (*Meria larisis*); successive years of infection lead to reductions in available cone crops. Cone maturation follows elevation gradients; cones at lower elevations are generally mature in mid-August and seed dispersal occurs into September. As such, seed may be available for dispersal during the fire season.

Seed germinates best on seedbeds exposed by burning or mechanical scarification (Antos and Shearer 1980; Beaufait et al. 1977; Schmidt 1969; Shearer 1976). Western larch seedlings survive poorly on undisturbed litter, humus, or sod or with heavy root competition; seedlings germinated on duff do not often survive (Beaufait et al. 1977). High solar irradiation is the most important physical factor affecting seedling survival (Shearer 1976). Southerly and west exposures are generally too severe for western larch seedlings to establish, particularly in drier sites at the lower elevational limits of its range. In the middle and northern portion of its ranges, western larch grows well on all exposures. Young seedlings grow fast on desirable sites. Only lodgepole pine is similar to western larch in seedling growth; Douglas-fir grows at about half the rate, and spruce and subalpine fir at about one-quarter the rate, of western larch. Site productivity has the most effect on height growth on western larch sites (Shearer 1976).

Western larch is a long-lived early seral species. It is a fast-growing species with tall, open crowns, making the species easily able to outgrow all of its competitors on the more mesic sites (Milner 1992). It is also the most shade-intolerant conifer in the Northern Rockies (Minore 1979) (table 6.3); it can tolerate partial shading only in the seedling stage. Western larch is replaced through succession by all other conifers except for ponderosa pine. Western larch is moderately drought tolerant and can survive seasonal drought, but performs poorly when droughts last more than 1 or 2 years. Douglas-fir is the most common associate, but others include ponderosa pine on lower drier sites; western hemlock, western redcedar, and western white pine on moist sites; and Engelmann spruce, subalpine fir, lodgepole pine, and mountain hemlock on cool, moist subalpine sites (Schmidt and Shearer 1990). It has been associated with mycorrhizal fungi in many portions of the region (Harvey et al. 1978).

Western larch has average genetic diversity with a weak population differentiation. Its low levels of differentiation indicate that it is more a generalist than a specialist. The species has a moderate outcrossing rate, and the patterns of genetic variation are mostly dominated by latitude and longitude. Populations need to be separated by 1,640 feet in elevation before genetic differentiation is expected.

Disturbance Interactions

Wildland fire is essential to the maintenance of western larch populations. Western larch depends on the open-canopy high light environments, and mineral soil seedbeds created by fire for successful, widespread regeneration (Schmidt et al. 1976). Western larch has unique characteristics that allow it to survive intense fire, including the thickest bark (Ryan and Reinhardt 1988), high crowns with high moisture contents, deep roots, and epicormic branch production (Fiedler and Lloyd 1995; Harrington 2012; Schmidt and Shearer 1995; Schmidt et al. 1976). Western larch is one of the few Northern Rockies tree species that has adapted to survive mixed-severity to stand-replacement fires (Hopkins et al. 2013; Marcoux et al. 2015). Tall surviving western larch can produce copious seeds that are wind dispersed across large burns to land on mineral soil seedbeds and ensure continued western larch domination (Stoehr 2000). However, if serotinous mature lodgepole pine trees occur with western larch, regeneration may be dominated by both species (e.g., Hopkins et al. 2013). Because western larch grows quicker and taller, it often outcompetes lodgepole pine to attain dominance (Pfister et al. 1977).

Western dwarf mistletoe (*Arceuthobium campylopodum*) is perhaps the most damaging disease-causing parasite of western larch (Schmidt and Shearer 1990). It can infect seedlings as young as 3 to 7 years old and continue throughout the life of the tree. In addition to killing treetops, reducing seed viability, creating conditions suitable for entry of other diseases and insects, and causing burls, brashness, and some mortality, it decreases height and diameter growth. Three other important diseases are found in western larch: needlecast caused by *Hypodermella laricis*, brown trunk rot, and red ring rot. The exotic larch casebearer (*Coleophora laricella*) and native western spruce budworm are currently the two most serious insect pests of western larch (Schmidt and Fellin 1973). However, neither of these agents causes substantial mortality. Western larch is susceptible to defoliation as a result of the recent western spruce budworm outbreak (DeNitto 2013). Larch needle cast results in substantial needle damage in cooler, moister springs. Episodic outbreaks of larch casebearer and western spruce budworm can also cause defoliation severe enough to reduce the current year's tree growth (Schmidt et al. 1976) and disrupt cone production.

Historical and Current Conditions

The more mesic montane western portions of the region were often dominated by extensive western larch forests that had regenerated after major fires. The species dominated northwestern Montana and was the major timber species for most of the 1950s to 1970s. However, extensive logging as early as 1908 on USFS lands (Arno 2010) removed many of the large tall western larch that could have survived fire and cast seed across the landscape, and effective fire exclusion has removed the burned, mineral soil seedbeds where western larch can regenerate. Continued fire exclusion has served to increase forest density and surface fuel loads so that future fires may be more severe.

Climate Change Responses

Western larch is a species that is highly susceptible to climate warming. Most climate change studies predict major losses of western larch throughout the Northern Rockies. Morales et al. (2015) used an SDM approach to project a 41-percent loss of western larch in its range in the western United States, and Aston (2010) reports major expected declines in western larch habitat in the northern Rocky Mountains. Rehfeldt and Jaquish (2010) projected major shifts in western larch in the western portions of the Northern Rockies, with major losses in Montana and gains in Idaho. Nitschke and Innes (2008) used gap modeling approaches to simulate major losses in western larch in most of British Columbia. Coops and Waring (2011) suggest that western larch may invade many areas vacated by lodgepole pine in the future in some portions of the Pacific Northwest. In addition, considering western larch associates, competitive interactions among species may play a critical role in the current and projected distribution of tree species such as western larch (Thuiller et al. 2008). Although temperature-precipitation interactions tend to set the limits where species can successfully compete, temperature alone seems primarily responsible for adaptation of populations within those limits (Rehfeldt et al. 2014).

Western larch will probably migrate to more northerly and higher areas in the Northern Rockies, but not without surviving major fires. Gray and Hamann (2013) estimated western larch could migrate more than 500 miles northward and more than 1,100 feet higher in elevation in the region by 2050. Western larch has the ability to quickly take advantage of changes in productivity of colder sites, providing these areas burn and the western larch survives the fires to provide sufficient seed for colonization. Increasing fires may serve to return western larch to the Northern Rockies landscape, but managers may need to provide substantial assistance by planting western larch in burned areas before other species become established. Continued fire exclusion will probably result in major declines of western larch in the western portions of the region because increased competition will reduce vigor, making the trees more susceptible to damaging agents; surface and canopy fuel buildups will be so great that many relic western larch trees will die in uncharacteristically severe fires (Arno et al. 1997; Davis 1980; Norum 1974). Keane et al. (1996) simulated major declines in the future for western larch under fire exclusion and moderate climate change, but found it increased as more fire was allowed to burn in the Glacier National Park landscape.

Western larch exhibits an intermediate adaptive strategy, and geographic clines for most adaptive traits are relatively flat (Rehfeldt 1994, 1995b). Populations from northerly latitudes and higher elevations exhibit the lowest growth potential, least tolerance to larch needle cast, and the lowest survival. Using a common garden study of 143 populations, Rehfeldt (1995b) demonstrated that an increase of 9 °F would produce a mean annual temperature exceeding the current ecological distribution of the species. A molecular study using allozymes indicated low levels of genetic differentiation among populations from the inland Northwest (Fins and Steeb 1986). Evolutionary bottlenecks are commonly cited as a mechanism of reduced genetic differentiation among populations, and increased differentiation within populations.

Western larch may be highly susceptible to future changes in climate primarily because of its narrow geographic and elevational distribution in the region and its uncertain association with wildland fire. If wildland fires increase, western larch may have a distinct colonization advantage, providing fire mortality is low in those communities that have extensive fuel buildups from fire exclusion. However, if fires decrease and exclusion is continued, western larch may be outcompeted by its shade-tolerant competitors, and those seed-producing western larch that remain might be killed by severe fires created by abnormal fuel accumulations. If western larch is planted in those severely burned areas, the species will surely remain on the landscape in the future.

Western White Pine

Autecology

In the Interior West, western white pine (*Pinus monticola*) grows from near Quesnal Lake, British Columbia, south through the Selkirk Mountains of eastern Washington and northern Idaho and into the Bitterroot Mountains of western Montana (Graham 1990). Isolated populations are found as far east as Glacier National Park (Loehman et al. 2011a). The climate of the interior portion of western white pine range is influenced by the Pacific Ocean; summers are dry and most of the precipitation occurs in the fall and winter. Western white pine is limited by moisture at lower elevations and temperatures at upper elevations. The southern boundary is limited by a balance of precipitation and evaporation.

Western white pine grows on diverse of soil types in the Northern Rockies (Harvey et al. 2008), but it primarily grows in areas where the upper soil layers are composed of loess or loess-like material. In this region, it generally grows at elevations of between 1,600 and 5,900 feet and where the topography is steep with V-shaped and round-bottomed valleys. It grows on a variety of slopes, but is common along moist creek bottoms, lower benches, and northerly slopes. Western white pine grows in association with a variety of species, and in the western hemlock/bride's bonnet (*Clintonia uniflora*), western redcedar/bride's bonnet, and grand fir/bride's bonnet habitat types (Cooper et al. 1991).

Western white pine seeds require 20 to 120 days of cold, moist conditions before germination occurs. Seeds germinate in the spring when soil moisture is at field capacity from melting snow. Western white pine seedling establishment is favored by partial shade on severe to moderately severe sites (Graham 1990) but little to no shade on north slopes. Under full sun, germination begins earlier and ends earlier than in shaded conditions. Mineral soil surfaces are preferred over duff. Once established, western white pine grows best in full sunlight on all sites. Seedlings have low drought tolerance, and seedling mortality late in the first growing season is attributed to high surface temperatures on exposed sites, and drought in heavily shaded areas where root penetration is slow. Early root and shoot development is not rapid.

Western white pine is almost always a seral species and is classified as intermediate in shade tolerance (Minore 1979). It attains dominance in a stand only following wildfire or with silvicultural systems that favor it. It is tolerant of cold when it is dormant, and similar to lodgepole pine in cold tolerance.

Genetic variation of western white pine is high, with the greatest difference being among trees within a stand, although differences occur among stands and elevational zones. The adaptation of western white pine to different conditions (topographic, climatic, geographic, and edaphic) is governed more by phenotypic plasticity than by selective differences. The species has a high outcrossing rate and average genetic diversity with moderate genetic differentiation. It is a generalist species with broad climate and environmental tolerances (Devine et al. 2012).

Disturbance Interactions

Historically, western white pine forests mostly originated from wildfires, especially stand-replacement burns, but were also maintained by frequent low-severity fires (Barrett et al. 1991). The species, especially when mature, is more tolerant of heat and can better survive fire than nearly all of its shade-tolerant competitors. Its relatively thick bark and moderately flammable foliage make it intermediate in fire resistance among its conifer associates (Graham 1990). Native American burning was probably the primary source of fire that created the pure stands of western white pine in northern Idaho, but lightning was also important (Graham 1990).

The most prominent agent causing the severe declines in western white pine is white pine blister rust (Fins et al. 2002; Harvey et al. 2008). A combination of climate, extensive white pine blister rust, abundant alternate hosts, and susceptible western white pine caused significant losses in the recent past. Selection of naturally resistant trees as seed sources and planting of rust-resistant nursery stock can reduce losses. In the absence of blister rust, western white pine is long-lived, commonly surviving to 300 to 400 years old.

The foremost root disease of western white pine is Armillaria root rot, which causes fading foliage, growth reduction, root-collar exudation of resin, dead and rotten roots, and black rhizomorphs. Annosus root disease and laminated root rot (*Phellinus sulphurascens*) also cause some mortality of individuals and groups. In periods of drought, pole blight, a physiological disorder, can occur in stands of the 40- to 100-year age class, causing yellow foliage and dead resinous areas on the trunk. Later, the top dies and, after a few years, the tree dies. The disease apparently is not caused by a primary pathogen but results from rootlet deterioration in certain soils, which restricts the uptake of water. Bark beetles are the most important insects that attack western white pine, and the most important species is the mountain pine beetle, which kills groups of trees, primarily in mature forests. The mountain pine beetle often attacks trees weakened by blister rust. Likewise, the red turpentine beetle (*Dendroctonus valens*) sometimes attacks weakened trees.

Historical and Current Conditions

Western white pine stands were once extensive across northern Idaho and parts of Montana, with large, tall pine trees dominating the montane landscapes (Harvey et al. 2008). As a result of logging, fire suppression, and extensive white pine blister rust infection, western white pine forests are nearly gone, and the species occurs only as scattered individuals in mixed-conifer stands (Fins et al. 2002). This is truly an ecosystem in decline, and it may be doomed to extinction without active restoration.

Climate Change Responses

Western white pine presents a special challenge in forest management in the future. Recent studies have shown that western white pine might be the species best adapted to changes in climate in the northwestern portions of the Northern Rockies (Loehman et al. 2011a). Using SDMs, Gray and Hamann (2013) estimated western white pine could move almost 250 miles northward and 500 feet higher in elevation in the region by 2050. Its superior growth rates, ability to survive fire, and high timber value make it a species to promote in the creation of future forests with high resilience (Baumgartner et al. 1994; Graham 1990). It has the unique ability to disperse seeds into burned areas, which are likely to increase in the future, and the predicted increases in productivity where it occurs could mean that it might benefit more than any other tree species from changing climate, especially in the context of timber production. However, western white pine is currently ravaged by white pine blister rust, and it has not yet developed the genetic capacity to overcome the damaging effects of this exotic disease to populate future landscapes in the northwestern Northern Rockies (Fins et al. 2002). The species simply has not achieved a sufficient level of rust resistance to allow it to dominate future stands (Harvey et al. 2008). With white pine blister rust and its alternate hosts (e.g., currant) predicted to also increase in the future, creating a rust-resistant population of western white pine is critical to maintaining it in the mesic grand fir/western redcedar/western hemlock habitat types (Baumgartner et al. 1994). Without a comprehensive western white pine restoration program, there is little chance that this species will play an important role in the future.

Other issues may govern future western white pine dynamics. While wildland fire may increase growing space, thereby providing for high regeneration potential, there may be few western white pine trees to provide the seed source needed to regenerate these large burns. Further, in some portions of the Northern Rockies, the species may be highly dependent on ash cap soils (Graham 1990), which may prevent its migration to warmer sites. The species is also dependent on a unique assemblage of ectomycorrhizae whose availability in some areas might be reduced in the future; suitable substrate may not exist upslope so migration may not occur.

In summary, we deem western white pine to be highly predisposed to declines in the future due to the interacting effects of continued fire exclusion, low levels of white pine blister rust resistance in native populations, and rapid succession to more shade-tolerant conifer communities. While it may be a species of special interest for northwestern portions of the Northern Rockies, its populations and future under changing climates is especially precarious because of blister rust. Abundance of western white pine is currently low in isolated landscapes, and thus the magnitude of any decline may be large relative to current and past populations.

Quaking Aspen

Autecology

Quaking aspen (*Populus tremuloides*) is the most widely distributed native tree species in North America and is abundant in the mountains of western and southwestern Montana and northern Idaho (DeByle 1985; Perala 1990). Its habitat is limited primarily to areas of water surpluses (where annual precipitation exceeds evapotranspiration), and it is also limited by minimum or maximum growing season temperatures. Deterioration of aspen stands is often related to warmer summer temperatures (Perala 1983). Aspen grows on a variety of soils, but growth and development are strongly influenced by both physical and chemical properties of the soil. The best soils for aspen growth are usually well drained, loamy, and high in organic matter, calcium, magnesium, potassium, and nitrogen. Aspen has an important role in nutrient cycling because of its rapid growth and high nutrient demand. Aspen is limited by both shallow and deep water tables (>8.2 feet) because the roots need sufficient water and good aeration, especially during the growing season.

Compared to most conifer species, aspen is a short-lived, disturbance-maintained seral species (Mueggler 1985; Rogers 2002). It is shade-intolerant and aggressively sprouts following any disturbance (usually fire) that kills most of the live stems, thus stimulating vegetative reproduction (i.e., suckering) (Bartos 1978). Aspen reproduces primarily by asexual root sprouting. Parent trees (genets) produce stems/trees (ramets), resulting in a clone or stand of genetically identical aspen stems (trees). Damage to parent trees alters the growth hormones (auxins and cytokinins) and stimulates a sprouting response (Perala 1990). Soil temperature is the most critical abiotic factor affecting suckering. Light is not needed for suckering but is needed for secondary growth. Eventually, most of the original root connections are severed as the ramets develop their own root systems to support nutrient uptake (Rogers et al. 2007; Shepperd and Smith 1993). This reproductive strategy allows aspen to establish quickly on disturbed sites and outcompete conifers for soil moisture, nutrients, and light. In addition, the shared root system maintains overall stand vigor by allowing sharing of resources during the early stages of stand development (Mitton and Grant 1996; Romme et al. 1997).

Although there are a number of different aspen classifications (Mueggler 1988; Shepperd et al. 2001), it is widely recognized that aspen occurs as both stable climax communities and seral, disturbance-maintained communities (Mueggler 1985, 1988). Stable aspen communities occupy sites with both high soil moisture and solar radiation, which apparently preclude establishment of conifers for very long periods (or they never establish). Stable climax aspen communities do not require disturbance to maintain dominance. Seral aspen apparently occurs in two forms in the Northern Rockies. First, there are mesic stands in northwestern Montana and northern Idaho where aspen is a common seral component, but it rarely dominates stands unless there are several consecutive burns (Campbell and Bartos 2001; Cooper et al. 1991; DeByle 1985). The second seral type occurs in the drier forested areas in the region, such as east of the Continental Divide, especially in southwestern Montana. In these seral types, which are created by fire and sprouting, aspen occurs as the major stand component; these stands will eventually succeed to more shade-tolerant conifers (e.g., Douglas-fir, subalpine fir, and Engelmann spruce) in the absence of disturbance (DeByle 1985; Mueggler 1988).

Aspen has a high genetic diversity because it is essentially a transboreal broadleaf tree. It has weak genetic differentiation geographically, but strong differentiation at the population level. There is substantial phenotypic variation in the species, found both in the field and in genetic studies, as documented by varied leaf sizes, shapes, and phenologies.

Disturbance Interactions

Numerous factors other than competition will be important for quaking aspen under a changing climate. Perhaps the most important factor affecting aspen regeneration and distribution is browsing by ungulates, which frequently damage reproduction by browsing and by rubbing their antlers against the stems (Eisenberg et al. 2013; White et al. 1998). Elk (*Cervus elaphus*) and moose (*Alces alces*) can also damage pole- and saw log-size trees by “barking” them with their incisors. Such injuries often expose individuals to secondary attack by insects or pathogens. Heavy use by overwintering ungulates can greatly reduce the number of aspen trees in localized areas. Cattle and sheep browsing is a serious problem in many areas of the Northern Rockies as livestock are allowed to range through recent aspen clearcuts. Mature aspen stands adjacent to livestock concentrations (water holes, salt blocks, and isolated stands in large open areas) often have root damage, are declining, and have few if any suckers present.

Fire can kill aspen stands (Bartos 1998), but it also creates conditions conducive to aspen regeneration and suckering by eliminating shade-tolerant conifers, which compete for light and eventually overtop and shade out aspen, leading to aspen decline (Campbell and Bartos 2001; Shepperd et al. 2001). Mature aspen trees may not survive fire as well as the fire-adapted conifers of the region, but aspen is easily the most competitive after fire because it can aggressively regenerate from suckers (Shinneman et al. 2013). Aspen could do well in a warmer climate with increased fire frequency, but moisture will limit its success, with varying results (Anderegg et al. 2012; Hogg and Hurdle 1995; Kulakowski et al. 2013; Worrall et al. 2013).

Following disturbance, aspen normally dominates a site for 40 to 80 years. Natural thinning from disease, aging, and succession (shading) by competing conifers eventually reduces aspen abundance (Mueggler 1985; Rogers 2002). In central Utah, Shepperd et al. (2001) found that both regenerating and nonregenerating clones had stems of various age classes, which suggests that periodic sucker events occurred in these clones. In addition, they found that all of the nonregenerating clones had fewer roots than their regenerating neighbors, which indicates that root systems decline when clones are not periodically regenerating. Many aspen clones are known to be associated with ectomycorrhizae (Cripps and Miller 1993).

Aspen has low susceptibility to insect damage, except in urban plantings, but stem canker diseases have a significant impact on aspen ecosystems. Depending on the fungus, cankers may kill trees within a few years or persist for decades. Hypoxylon canker caused by *Hypoxylon mammatum* is probably the most serious aspen disease east of the Rocky Mountains, killing 1 to 2 percent of the aspen annually (Perala 1990). Young trees are killed by small rodents and other mammals, particularly large ungulates (Eisenberg et al. 2013).

Historical and Current Conditions

Since around 1970, aspen has been in a period of general decline that is thought to be the result of wildfire exclusion, which has allowed plant succession to proceed toward conditions that ordinarily exclude aspen (Campbell and Bartos 2001; Frey et al. 2004). Recent episodes of aspen dieback have been superimposed on this general decline. Dieback can be recognized by the suddenness of the impact, giving rise to the term “sudden aspen decline,” and by an epidemiology that begins with the death of branch tips, death of mature trees, and eventually death of entire clones (Frey et al. 2004). The dieback is suspected to be caused by drought.

Climate Change Responses

Aspen is a species that may experience both gains and losses under future climate, depending on local site conditions, particularly soil moisture. Seral aspen communities will respond differently from stable, climax aspen communities. Aspen communities on warmer, drier sites could have high mortality because of increasing water deficit. Ireland et al. (2014) found that drought was the major factor causing recent high mortality in southwestern aspen stands. In the boreal forests of western Canada, Hogg and Hurdle (1995) estimate that even with an 11-percent increase in precipitation, boreal forests in which aspen is a major component will decline due to drought stress. Sudden aspen decline has been associated with severe, prolonged drought, particularly in aspen stands that are on the fringe of aspen distribution (warmer and drier sites than those typically considered optimal for aspen persistence) (Frey et al. 2004). Recent research efforts have found that extreme weather events (e.g., drought, thaw-freeze events), insect defoliation, or pathogens, or a combination of these factors, have led to aspen mortality (Brandt et al. 2003; Candau et al. 2002; Hogg et al. 2002). Marchetti et al. (2011) found that aspen mortality from various insects and disease (e.g., Cytospora canker [*Cytospora*], bronze poplar borer [*Agrilus liragus*], and aspen bark beetles *Trypophloeus populi* and *Procryphalus mucronatus*) was greater in those stands that were drought-stressed and declining due to sudden aspen death. Further exacerbating the situation is that declining stands may have little or no regeneration because of intense ungulate herbivory, and those smaller stands that persist may be smaller and fewer with increased plant stress due to increased severity of summer droughts (Rogers et al. 2013). Growth may increase because photosynthetic rates appear to increase more in aspen than other tree species as atmospheric carbon increases, but this may be offset by increased atmospheric ozone, which reduces photosynthesis and may increase susceptibility to insects and disease.

Increased fire frequency, particularly on moist sites, is likely to favor aspen regeneration in the future by removing shading conifers, and younger stands (<40 years old) created by fire may be more resilient to drought. However, if future fires are severe, they may kill the shallow root systems and eliminate aspen. Increased herbivory on regenerating stands may occur as adjacent upland vegetation senesces and desiccates earlier in the growing season. Areas with mountain pine beetle-caused conifer mortality (especially in lodgepole pine) may release aspen, and it will regenerate once the conifer canopy is thinned or removed, again given sufficient soil moisture.

Grand Fir

Autecology

Grand fir (*Abies grandis*) is found on a wide variety of sites, including stream bottoms, and valley and mountain slopes of the northwestern United States and southern British Columbia (Foiles et al. 1990). Average precipitation in its range varies from 20 to 100 inches, but in northern Idaho the average is from 20 to 50 inches. The average growing season temperature is 57.2 to 66.2 °F. In the inland portion of its range, grand fir grows best on rich mineral soils of valley bottoms but also grows well on shallow exposed soils of mountain ridges if moisture is adequate (Antos 1972).

Grand fir is either a seral or climax species, depending on site moisture (Ferguson and Johnson 1996). On productive mesic sites, it grows rapidly to compete with other seral species in the overstory, but it is outcompeted by western redcedar and western hemlock. On drier sites where western redcedar and western hemlock are excluded because of drought, it is the most shade-tolerant species and can easily dominate the understory; it eventually assumes the dominant position in the climax condition. Grand fir is a major climax species in a variety of habitat types in Montana and northern Idaho, but it rarely grows in pure stands; one exception is on the Clearwater River drainage in north-central Idaho (Cooper et al. 1991; Pfister et al. 1977). In Montana and parts of Idaho, grand fir can also share dominance, even in the climax state, with subalpine fir, especially in narrow valley bottoms where subalpine fir can exert dominance in lower elevational zones (Antos 1972). In most of its range, grand fir is often associated with Douglas-fir, ponderosa pine, western larch, western white pine, and subalpine fir.

Grand fir has a high tolerance to shade but a low tolerance to drought, even though it can tolerate drought better than any of the conifers that may succeed it in the absence of disturbance (e.g., western redcedar and western hemlock). Grand fir forms associations with ectomycorrhizae and arbuscular mycorrhizae, which may allow it to outcompete some shade-tolerant conifers. It has a very low frost tolerance but can tolerate seasonally fluctuating water tables. It is monoecious and produces large, winged seed dispersed by wind. It has average levels of genetic diversity but weak geographic differentiation.

Disturbance Interactions

Grand fir is susceptible to fire damage in moist creek bottoms but is more resistant on dry hillsides where roots are deeper and bark is thicker (Ryan and Reinhardt 1988). Grand fir is less resistant to fire than western larch, ponderosa pine, and Douglas-fir but more resistant than subalpine fir, western hemlock, and Engelmann spruce. Most fires that burn grand fir sites are stand-replacement or mixed-severity, and these fires burn in fuels that generate sufficient heat to kill most grand fir trees (Arno 1980; Arno et al. 2000).

Grand fir is susceptible to heart rot and decay. Armillaria root rot and annosus root disease are common root diseases causing high tree mortality (Hagle et al. 2003). Numerous insects attack grand fir. The western spruce budworm and Douglas-fir tussock moth have caused widespread defoliation, topkill, and mortality in grand fir. The western balsam bark beetle (*Dryocoetes confusus*) and the fir engraver (*Scolytus ventralis*) are the principal bark beetles attacking grand fir (Foiles et al. 1990).

Historical and Current Conditions

Fire exclusion has increased grand fir on both dry and mesic sites, but increased tree densities have also stressed grand fir trees, contributing to increased fuel loadings, higher root rot, and greater insect damage and mortality. Historically, grand fir sites were probably dominated by western larch, western white pine, Douglas-fir, and ponderosa pine because of frequent fires, but these sites have since succeeded to the more shade-tolerant grand fir, and on the productive mesic sites, to western redcedar and western hemlock. Therefore, the condition of most grand fir stands depends on the last severe fire; if fire exclusion has caused grand fir to dominate in both the overstory and understory, then these stands are usually highly stressed because of increased root rot and insect agents. However, in earlier seral stands that have not yet experienced high grand fir regeneration, a rise in grand fir cover types is likely with continued fire exclusion.

Climate Change Responses

On xeric sites, increased drought and longer growing seasons will exacerbate grand fir stress from competition, resulting in high mortality mainly from insects and disease. Nitschke and Innes (2008), using a gap modeling approach, projected major declines in grand fir, and Coops and Waring (2011) used a mechanistic model to simulate a nearly 50-percent decrease in the range of grand fir compared to historical distributions. Franklin et al. (1991) projected that grand fir will nearly disappear from the east slope of the Cascades.

Yet projections of increased productivity suggest increased grand fir populations on moderate sites (Aston 2010). Urban et al. (1993) projected an expansion of grand fir forests into upland xeric sites of the Pacific Northwest. On mesic sites where grand fir is seral to western redcedar and western hemlock, the longer growing seasons coupled with higher temperatures may increase growth rates and regeneration success, thereby increasing tree density and competition and effectively reducing grand fir components. The opposite is true on those sites where grand fir is the indicated climax; grand fir will increase in both the overstory and understory in the absence of disturbance.

Disturbance, specifically fire, could be the major factor in the rearrangement of grand fir communities across the Northern Rockies landscape. Longer fire seasons and high fuel loadings from both fire exclusion and increased productivity will serve to foster large, severe fires that may reduce grand fir, especially on those sites where it is the indicated climax species (i.e., grand fir habitat types). Fire will reduce grand fir dominance at both landscape and stand scales.

In summary, although many grand fir forests are highly stressed from high tree densities, the species will probably tolerate changes in climate and remain on the landscape at levels that are closer to historical conditions rather than the high abundance observed now.

Western Redcedar

Autecology

The inland range of western redcedar (*Thuja plicata*) extends from the western slope of the Continental Divide in British Columbia south through the Selkirk Mountains into western Montana and northern Idaho (Minore 1990). The southern limit is Ravalli County and the eastern limit is near Lake McDonald in Glacier National Park. A few trees may exist east of the Continental Divide near St. Mary's Lake (Pfister et al. 1977). Western redcedar is abundant in many forested swamps as well as sites that are too dry for western hemlock; it has better root penetration than western hemlock

(Habeck 1978). Western redcedar dominates wet ravines and poorly drained depressions. Where there is sufficient precipitation, low temperatures limit the range of the species. It is not resistant to frost and can be damaged by freezing temperatures in late spring and early fall.

Western redcedar occurs only in pure stands where fire has been excluded for a long time, or where fire has been used to maintain western redcedar dominance (Barrett 1988; Barrett and Arno 1991). It is commonly associated with a wide array of tree species: grand fir, western white pine, western hemlock, western larch, and ponderosa pine. Only western hemlock in the Northern Rockies is more shade-tolerant than western redcedar, but western redcedar can be overtopped by Douglas-fir, grand fir, western hemlock, and western white pine (table 6.3). Its relative shade tolerance may be higher in warm than in cool areas, but western redcedar is very tolerant wherever it grows. Often present in all stages of forest succession, western redcedar can occupy pioneer, seral, and climax positions. In the Northern Rockies, however, most western redcedar stands are in the late seral-stages; it is usually considered a climax or near climax species. It has little tolerance to drought but can exist in seasonally wet areas, especially near riparian systems (Devine et al. 2012).

Western redcedar regenerates best on disturbed mineral soil, although scorched soil is not beneficial to its regeneration. Rotten wood that is in contact with the soil is a preferred seedbed in western redcedar groves. Western redcedar also propagates by clones, and clones tend to be more abundant than young trees established by seed. Establishing seedlings survive best in partial shade, as they are not tolerant of high soil temperatures or frost. Young branches are susceptible to sunscald. Roots of young seedlings grow more slowly than Douglas-fir roots but faster than western hemlock roots, and shoots have the longest growing period of any of the associated conifers.

Western redcedar has very low levels of genetic diversity, and this diversity is weakly distributed geographically and within populations. Clines are very gentle and seed zones narrow. This species cannot tolerate wide ranges of environmental conditions.

Disturbance Interactions

Relative to its associates, western redcedar is not as affected by damaging agents, but because it is long-lived, damaged trees are common (Minore 1990). Although western redcedar trees are somewhat wind-firm, especially on dry sites, the trees are often wind thrown in wetter environments. Western redcedar is less susceptible to fire damage than Engelmann spruce, western hemlock, grand fir, and subalpine fir in the Northern Rockies. Western redcedar is also less susceptible than other associated species to root pathogens. However, root disease still impacts western redcedars, and fungi eventually invade heartwood typically resistant to decay. In North America, the most important fungi attacking western redcedar are root, butt, and trunk rots, most importantly laminated root rot, honey fungus (*Armillaria mellea*), and stringy butt rot (*Perenniporia subacida*).

Historical and Current Conditions

Compared to historical distributions, there has not been a significant increase or decrease in western redcedar distribution in the Northern Rockies. However, western redcedar dominance has probably increased in those stands occupying mesic western redcedar sites due to fire exclusion.

Climate Change Responses

With warmer temperatures, mesic northern Rocky Mountain ecosystems may increase in productivity (Aston 2010), and western redcedar may expand into more upland communities. Hamann and Wang (2006) projected that the western hemlock/cedar forests of British Columbia would double in range by 2050, and Urban et al. (1993), using gap modeling, simulated an expansion of western redcedar into upland western Oregon sites. Devine et al. (2012) rated western redcedar as having moderate vulnerability in the Pacific Northwest Region. With increased western redcedar productivity could come increased cone production and seed dispersal into new areas that might be more conducive to long-term seedling establishment.

Although warmer conditions in the future may benefit western redcedar, drier conditions in the future are likely to result in retraction of western redcedar to the warmer, wettest Northern Rockies sites; upland western redcedar stands might have high mortality from declining productivity. Woods et al. (2010) noted recent declines in western redcedar in British Columbia and attributed the decline to increased drought that decreased vigor and increased insect attacks and disease in western redcedar. Warming may also result in a loss of chilling required for western redcedar (Nitschke and Innes 2008), and the narrow genetic potential of western redcedar (Devine et al. 2012) may limit its expansion into new habitats. Using SDMs, Gray and Hamann (2013) estimated western redcedar might move 400 miles northward and 1,000 feet higher in elevation in the Pacific Northwest by 2050. However, in some portions of the region, redcedar is mostly associated with ash cap soils, so despite the possible creation of new habitats by new climates, the potential of non-ash soils to sustain productive western redcedar may be limited.

It is uncertain how disturbance will affect western redcedar in the future. Fire can serve to maintain western redcedar communities if it burns at low severities and kills only seedlings and saplings. However, high-severity wildfires could eliminate seed sources. Continued fire exclusion may maintain current western redcedar distributions, but without proactive fuels treatments, wildfire that occurs after long periods of exclusion may burn with sufficient severity to cause extensive western redcedar mortality. Further, new warm-cold cycles may facilitate the injury called red belt and adversely affect young western redcedar, as evidenced by increased flagging during past dry seasons.

In summary, western redcedar may not be severely affected by future climate warming. The species may remain in its current range, and productivity may increase in some settings.

Western Hemlock

Autecology

The inland range of western hemlock (*Tsuga heterophylla*) includes the west side of the Continental Divide in Montana and Idaho, north to Prince George, British Columbia (Packee 1990). Western hemlock thrives in mild humid climates and in environments with abundant soil moisture throughout the growing season (Hann et al. 1994). Where the growing season is relatively dry, western hemlock is confined primarily to northerly aspects, moist stream bottoms, or seepage sites. Western hemlock grows on a variety of soil types, although it is a shallow-rooted species and does not develop a taproot. Abundant roots, especially fine roots, grow near the soil surface and are easily damaged by fire.

Western hemlock is considered very shade-tolerant and is perhaps the most shade-tolerant tree species in the Northern Rockies (table 6.3). It is a major climax or near-climax species in the region and is found with nearly all of the other conifer species, including western redcedar, grand fir, Douglas-fir, western larch, western white pine, lodgepole pine, and ponderosa pine. Seed germination and germinant survival occur when there is adequate moisture. Western hemlock can germinate on a variety of materials and in both organic and mineral seedbeds. Decaying logs and rotten wood are often favorable seedbeds; decayed logs have the added benefit of providing nutrients. Western hemlock is highly susceptible to drought and demands abundant water throughout the growing season (Baumgartner et al. 1994). It is associated with some ectomycorrhizae. Its seedlings are highly susceptible to frost.

Western hemlock has relatively low genetic diversity and low geographic differentiation. It has a high outcrossing rate and average heterozygosity (Devine et al. 2012). Growth rate is more related to soil conditions than to genetics.

Disturbance Interactions

A variety of root and bole pathogens cause significant damage and mortality in western hemlock. It is also very susceptible to fire damage because of its shallow roots and thin bark, and it is also susceptible to windthrow owing to shallow roots. On droughty sites, top dieback is common, and entire stands of western hemlock saplings have been killed in exceptionally dry years. Western hemlock is highly susceptible to annosus root disease and Indian paint fungus (*Echinodontium tinctorium*), but seems to have a high tolerance to Armillaria root rots (Packee 1990).

Historical and Current Conditions

The current distribution of western hemlock is similar to its historical distribution. However, most stands with western hemlock have become denser and the western hemlock component has increased in both the overstory and understory. Overly dense western hemlock stands may be declining in vigor, thereby becoming more susceptible to disease, insects, and abiotic perturbations (e.g., windthrow).

Climate Change Responses

In the past, western hemlock/western redcedar forests were associated with wetter conditions in the low elevation forests of the Northern Rockies, but this type declined as fires and drought increased (Gavin et al. 2007). Thus, increased drought and area burned may decrease western hemlock abundance and distribution. Several studies have projected contractions in western hemlock distribution. For example, Hansen et al. (2001) simulated major contractions in western hemlock range, and Franklin et al. (1991) project that western hemlock will occupy about half its current range on the western slopes of the Cascades. Shafer et al. (2001) noted that western hemlock may decrease in range because chilling requirements for the seeds will not be met. Using a mechanistic landscape model, Keane et al. (1996) simulated losses of western hemlock and western redcedar under moderate climate warming in Glacier National Park, mostly as a result of severe fires. Cumming and Burton (1996) projected minor changes in the western redcedar-western hemlock zone in British Columbia under moderate warming. On the other hand, Hamann and Wang (2006) predicted that western hemlock would increase its range by more than 200 percent in British Columbia, and using gap modeling, Urban et al. (1993) simulated an expansion of western hemlock into upland western Oregon sites.

It is possible that western hemlock will maintain its current range under a changing climate. It may not have the diversity in growth habit that will allow it to expand its range into the more upland sites as temperatures warm. Because the species is dependent on ash cap soils, any migration may be relegated to those wetter and warmer sites without ash cap soils.

Lodgepole Pine

Autecology

Lodgepole pine (*Pinus contorta*) has wide ecological amplitude, but only the inland form (*P. contorta* var. *latifolia*) is found in the Northern Rockies (Lotan and Critchfield 1990). Lodgepole pine has the widest range of environmental tolerance of any conifer in North America (Lotan and Critchfield 1990). It is relatively resistant to frost injury and can often survive in frost pockets where other species cannot (Pfister et al. 1977). In Montana, lodgepole pine does not grow on highly calcareous soils derived from dolomitic limestone (Lotan and Perry 1983). Lodgepole pine is primarily found on moist soils developed on colluviums from other types of limestone and calcareous glacial till. It grows well on gentle slopes and in basins, but it is also found on rough and rocky terrain, steep slopes and ridges, and bare gravel (Lotan and Critchfield 1990). Compared to other associated species, lodgepole pine is intermediate in its needs for water, requiring more than Douglas-fir or ponderosa pine but less than spruce and subalpine fir in the region.

Lodgepole pine is intolerant of shade but highly tolerant of frost and drought (table 6.3). Occasionally seedlings become established under a forest canopy, but these individuals rarely do well and remain in a stunted form for long periods of time (decades to centuries). In the absence of fire, lodgepole pine is usually succeeded by its more tolerant associates, such as Douglas-fir in xeric environments and Engelmann spruce and subalpine fir in subalpine environments. Succession proceeds at variable rates, however, and is particularly slow in some high elevation forests (Arno et al. 1993). Lodgepole pine grows both in pure stands and in association with many conifers, primarily subalpine fir, spruce, Douglas-fir, and western larch (Steele et al. 1983). Its successional role is dependent on environmental conditions and on competition. It is seral in most mesic Northern Rockies forest communities (Arno et al. 1986). However, on cool dry habitats, such as those found in the Greater Yellowstone Area and southeastern Idaho, it is dominant and tends to be persistent and form near-climax communities (Despain 1983). Its ability to remain on xeric landscapes is enhanced by its association with many types of mycorrhizae.

Lodgepole pine has a great ability to regenerate due to a combination of cone serotiny, high seed viability, early rapid growth, and ability to survive a wide variety of microsite and soil conditions (Hardy et al. 2000). The serotinous cone habit, where cones open only after being heated by wildland fire, is common in the Rocky Mountains, but in general, the highest serotiny is found in the northern parts of the region. Large quantities of stored seeds are available for regeneration after fire, and annual seedfall from nonserotinous cones helps in restocking in areas of relatively minor disturbance and maintaining lodgepole pine presence in mixed stands.

The best lodgepole germination occurs in full sunlight and on bare mineral soil or disturbed duff, with little competition. Adequate soil moisture is required for germination and survival, with the first few weeks being most critical. In southwestern Montana most of the season's total germination occurs during the 2 weeks following snowmelt in late June when soil is saturated and temperatures most favorable. Drought is a common cause of mortality in first-year seedlings. Freezing temperatures may kill seedlings, but seedlings vary in frost resistance based on seed source. Frost heaving also causes mortality. Height growth begins earlier than in other associated species, except for other pines and western larch.

There is moderate genetic variation in strains of lodgepole pine, resulting in some strains growing well in cold climate and on poor sites. Lodgepole pine has an average genetic diversity but a weak differentiation across its Northern Rockies range and strong differentiation among populations. The species is a prolific seed producer and has a good cone crop at about 1- to 3-year intervals. It is wind pollinated and its seeds are wind dispersed.

Disturbance Interactions

Fire plays a critical role in lodgepole pine forest succession (Lotan et al. 1984). Typically, many Northern Rockies lodgepole pine forests originated from stand-replacement fires, but extensive fire scars in Northern Rockies lodgepole pine forests indicate the existence of a low-severity, nonlethal fire regime component in these forests, especially in many areas east of the Continental Divide (Arno et al. 1993; Stewart and Arno 1997). Lodgepole pine is apparently able to survive low-intensity fires quite well even though it has thin bark (Ryan and Reinhardt 1988). However, most lodgepole pine forests in the region have a mixed-severity fire regime in space and time, where all fire severity types are possible depending on available fuels, antecedent drought, and wind conditions (Arno et al. 2000). Consequently, lodgepole as a species will be well adapted to the fires of the future. Repeated fires, however, can eliminate lodgepole pine seed sources if the fires occur before existing lodgepole has become reproductively mature (approximately 10 years). In most cases, lodgepole pine natural regeneration often overwhelms a burned site with abundant seed from serotinous cones and thereby excludes other species (Lotan and Perry 1983; Nyland 1998).

The mountain pine beetle is the most important insect pest and has played a significant role in the dynamics of lodgepole ecosystems (Roe and Amman 1970). Past research has tried to link fire, beetles, and lodgepole pine in a complex web of interactions (Geiszler et al. 1980). However, recent findings have shown that fire and beetles often act independently to influence lodgepole pine dynamics (Axelson et al. 2009; Moran and Corcoran 2012; Schoennagel et al. 2012).

Historical and Current Conditions

Advancing succession due to fire exclusion is contributing to replacement of lodgepole pine with subalpine fir in many areas of the Northern Rockies. Keane et al. (1994) found successional advancement of subalpine communities in the upper subalpine landscape of the Bob Marshall Wilderness increased from less than 8 percent of the landscape to more than 22 percent. Concurrent increases in burn areas are creating many new lodgepole stands and some may become dense thickets, but coupled with increased drought, these dense lodgepole stands may exacerbate stress from other factors, including competition, endemic insects and diseases, and wind. Warming temperatures have heightened bark beetle activity, resulting in more frequent and severe outbreaks that have devastated many mature lodgepole pine communities in the central Northern Rockies (Carroll et al. 2003).

Climate Change Responses

Longer drought periods and warmer temperatures in the lower, south-facing, drier lodgepole pine subalpine environments may cause decreased tree growth and regeneration potential, perhaps resulting in a transition to more-xeric trees species, such as Douglas-fir. Chhin et al. (2008) found that recent warming has decreased lodgepole pine growth rates in the foothill lodgepole pine communities of the low elevation forests in Alberta. Coops and Waring (2011) used process modeling to simulate minor declines with moderate warming in lodgepole pine in the Pacific Northwest (Oregon, Washington, and British Columbia), but major type conversions to other species with greater warming. Using an SDM approach, Hamann and Wang (2006) projected a net 50 percent loss of lodgepole pine in British Columbia under severe warming. Nigh (2014) projected that lodgepole pine heights may decrease by roughly 3 feet in moderate future warming, but the species has the genetic capacity to mitigate this height loss. In contrast, Rehfeldt et al. (1999) found substantial decline in lodgepole pine growth and height with minor changes in climate. Chhin et al. (2008) also found that decreases in lodgepole pine growth were correlated with high summertime temperatures, presumably related to summer drought. However, they also found that lodgepole pine growth increases with high fall temperatures. Gray and Hamann (2013) used SDM techniques and estimated lodgepole pine would move more than 250 miles northward and more than 650 feet higher in elevation in the Northern Rockies by 2050. Bell et al. (2014), using SDMs, projected losses of more than 70 percent of its Northern Rockies range by 2090. Given that lodgepole pine is a generalist that is capable of regenerating and growing in a wide range of environments, it is likely that the decline of lodgepole pine from drier sites will occur only under extreme warming scenarios (e.g., RCP 8.5 and A2) over longer time periods.

Another possibility is that, in the higher elevational areas of the Northern Rockies subalpine, where seasonal drought is not a problem, warming climates may actually increase lodgepole pine productivity because of high precipitation (Aston 2010). Johnstone and Chapin (2003) show that lodgepole pine is not in equilibrium with current climate; thus the response of the species to climate shifts will be difficult to predict using SDM approaches. However, they found that there are places where lodgepole pine will be positively affected by climate change. Wang et al. (2006) projected major increases in lodgepole pine productivity under future climates with moderate warming, but major decreases and perhaps local extinctions under extreme warming. Romme and Turner (1991) projected increases in the lodgepole pine zone in the GYA under moderate warming.

A third possibility is that lodgepole pine will migrate into areas where it is currently excluded by harsh, cold, windy conditions, such as the upper subalpine and treeline. This relocation process will likely be catalyzed by fire, especially in those areas with high serotiny. Clark et al. (2017) found that under moderate warming, lodgepole pine would remain on the GYA landscape, but it would also expand into higher elevation environments historically occupied by whitebark pine. Most studies have projected the migration of lodgepole pine into the whitebark pine zone (Hamann and Wang 2006; Romme and Turner 1991).

Lodgepole pine is well adapted to increases in fire occurrence, depending on level of serotiny (Turner et al. 1999). Smithwick et al. (2009) simulated some positive increases in GYA lodgepole pine after fire and under climate change. If fire is too frequent, however, lodgepole may be eliminated from sites where fires reburn stands before established seedlings and saplings become reproductively mature. Clark et al. (2017) simulated major and rapid decreases in GYA lodgepole pine under high climate warming due to both inhospitable environments and too frequent fire.

In mesic subalpine sites, continued fire exclusion coupled with higher productivities will certainly heighten competitive interactions and put more lodgepole pine trees into stress, thereby increasing mortality, vulnerability to insects and disease, and canopy and surface fuels, and accelerating succession toward subalpine fir (Smithwick et al. 2009). Severe fires that then occur in these advanced successional communities could convert communities back to lodgepole pine, providing there is not a loss of seed source. Fire exclusion, especially in areas of high serotiny, might delay the expansion of lodgepole pine.

Projected increases in climatic conditions that facilitate mountain pine beetle outbreaks could reduce lodgepole pine populations and forest extents (Creeden et al. 2014; Gillette et al. 2014). Lodgepole pine is highly susceptible to bark beetle mortality, especially on those landscapes where fire exclusion has resulted in an abundance of mature hosts (Temperli et al. 2013). Bark beetle outbreaks will favor the more shade-tolerant, nonhost tree species, thereby creating dense stands that may be subject to severe crown fires after 10 to 20 years. If beetle-killed stands burn, lodgepole pine can occupy the burned area only if viable seed sources remain. The varying levels of serotiny and beetle mortality will dictate future stand conditions in beetle-killed stands. Landscape heterogeneity is the only hedge against massive declines of lodgepole pine in the future (Logan and Powell 2001).

In summary, lodgepole pine is expected to both expand and contract in range, but as long as fire remains on the landscape, the species is likely to maintain its presence in the Northern Rockies at roughly the same proportions as during the last 100 years, albeit in different areas. The species is highly exposed to any climate changes because of its wide range and diverse growing environments. But Soulé and Knapp (2013) suggest that the steep clines associated with lodgepole pine may be driven more by density-dependent selection than by environmental selection, so this species may be well adapted to future changes in climate. In addition, although the magnitude of climate effects is likely to be great for tree growth, it may be only moderate for species survival compared to other species. The likelihood of these effects is highly uncertain, primarily because of the uncertainty about fire frequency and severity, which determine the extent to which fire will continue to play its role in the maintenance of lodgepole as a major component on future landscapes.

Whitebark Pine

Autecology

Whitebark pine (*Pinus albicaulis*) is an important component of high-elevation upper subalpine forests in the western United States and Canada (Arno and Hoff 1990). It is a keystone species because it supports unique community diversity, and it is a foundation species because of its roles in promoting community development and stability (Tomback and Achuff 2010; Tomback et al. 2001). More than 90 percent of whitebark pine forests occur on public lands in the United States and Canada, so maintaining whitebark pine communities requires a coordinated effort across Federal, State, and Provincial land management agencies (Keane et al. 2012).

Whitebark pine is a long-lived tree of moderate shade tolerance (Minore 1979) (table 6.3). It is common to find mature whitebark pine trees well over 400 years of age, especially on harsh growing sites; the oldest is more than 1,275 years (Luckman et al. 1984). Well-formed, thrifty individuals often have smooth gray bark, especially in the tree crowns, which may appear whitish in bright sunlight (Arno and Hoff 1990). Whitebark pine is slow growing in both height and diameter, and it rarely grows faster than most of its competitors, except on the most severe sites (Arno and Hoff 1990). In general, whitebark pine grows where summers are short and cool and where most precipitation comes in the form of snow and sleet, with rain only in June through September. Whitebark pine survives strong winds, thunderstorms, and severe blizzards, and is one of the few upper subalpine species that can tolerate long periods of drought (Callaway et al. 1998).

Whitebark pine is a major component of high elevation forests throughout the upper subalpine and treeline zones in the Northern Rockies (Arno and Hoff 1990). Whitebark pine forests occur in two high mountain biophysical settings. On productive upper subalpine sites, whitebark pine is the major seral species that is replaced by the more shade-tolerant subalpine fir, Engelmann spruce, and mountain hemlock, depending on geographic region (Arno 2001). These sites, referred to as "seral whitebark pine sites," support upright, closed-canopy forests in the upper subalpine lower transition to treeline, just above or overlapping with the elevational limit of the shade-intolerant lodgepole pine (Pfister et al. 1977); the two pine species can often share dominance. Other minor species found with whitebark pine on these sites are Douglas-fir, limber pine, and alpine larch (Keane et al. 2012). Sites where whitebark pine is the only tree species able to successfully dominate high elevation settings (called climax whitebark pine sites) occur in the upper subalpine forests and at treeline on relatively dry, cold slopes. Other species, such as subalpine fir, spruce, and lodgepole pine, can occur on these sites, but as scattered individuals with truncated growth forms. Whitebark pine can also occur as krummholz, elfin forests, clusters, groves, tree islands, and timber atolls in the alpine treeline ecotone (Tomback 1989) and as a minor seral in lower subalpine sites (Cooper et al. 1991; Pfister et al. 1977).

Whitebark pine is eventually replaced, in the absence of fire, by the shade-tolerant subalpine fir, spruce, and mountain hemlock on the productive, seral whitebark pine sites (Arno and Hoff 1990). It can take 50 to 250 years for subalpine fir to replace whitebark pine in the overstory, depending on the local environment and fire history (Keane 2001). Whitebark pine competes with lodgepole pine during early successional stages in the lower portions of its elevational range. Lodgepole pine usually has the competitive advantage over whitebark pine when it establishes from seed after a stand-replacing disturbance event because of its fast growth, serotiny, and copious seed production.

A bird (Clark's nutcracker) and whitebark pine have coevolved into a mutualistic relationship that ensures their continued presence on the landscape (Tomback 1982, 1983). Whitebark pine has evolved a nearly exclusive dependence on nutcrackers to disperse its large wingless seeds, and in turn, nutcrackers utilize the large whitebark pine seeds as an important food source. The key behavior that benefits the whitebark pine is the tendency of nutcrackers to bury thousands of whitebark pine seeds each year as food stores in small clusters or "seed caches" across diverse forest terrain (Keane et al. 2012). Nutcrackers retrieve these seed caches primarily in spring and summer as an important food source for themselves and their young. However, not all seed caches may be recovered, particularly following a large cone crop. Snowmelt, spring rains, and summer showers stimulate seed germination, leading to whitebark pine regeneration. Although whitebark pine depends nearly exclusively on nutcrackers, nutcrackers often harvest and cache seeds of other large-seeded pines.

Whitebark pine is a genetically diverse species because of its dependence on bird-mediated seed dispersal (Keane et al. 2012). As a result, the species is highly adapted to exist across many environments, and is limited only by competition, even at the lowest elevations. It has only six seed zones across its entire range, so it is easily able to migrate across local landscapes to rapidly take advantage of newly burned areas. Whitebark pine has weak geographic differentiation in the Northern Rockies, but a moderate level of inbreeding. One concern in the future is that the breeding of rust resistance in future whitebark pine seedlings may compromise other important traits; Mahalovich et al. (2006) found lower cold tolerance in highly rust-resistant seedlings grown in the nursery.

Disturbance Interactions

Whitebark pine fire regimes are complex and variable in space and time, but in general, all three types of fire severities describe whitebark pine fire dynamics: nonlethal, stand-replacing, and mixed-severity (Morgan et al. 1994b). Some whitebark pine stands may undergo fire events that burn in low-intensity nonlethal surface fires (sometimes called underburns or low-severity fires) because of sparse surface and canopy fuel loadings and unique topographical settings. However, most fires in the upper subalpine burn in mixed-severity patterns that best facilitate continued existence of whitebark pine (Keane et al. 1994).

Mountain pine beetle is by far the most damaging insect in mature stands of whitebark pine. Much of the mature whitebark pine in the Northern Rockies was killed by this insect between 1909 and 1940. Epidemics evidently spread upward into the whitebark pine forest after the beetle became established in the lodgepole pine forests below. The GYA whitebark pine ecosystems have recently suffered one of the most severe mountain pine beetle mortality events in recorded history.

The principal disease of whitebark pine is the introduced white pine blister rust (Schwandt 2006). Blister rust is particularly destructive where the ranges of whitebark pine and blister rust coincide with currant, the alternate host of the rust. Where there is a source of inoculum from lowland forests, the spores that infect pine can be carried by wind to the trees, but cool, moist conditions are needed for infection in whitebark pine. Blister rust damage is severe and prevents tree development in many upper subalpine settings of northern Idaho and northwestern Montana. Whitebark pine has some resistance to the disease, and efforts at developing rust-resistant seed for regenerating burned and treated areas have been very successful.

Historical and Current Conditions

Whitebark pine has been declining since the early 20th century from the combined effects of native mountain pine beetle outbreaks, contemporary fire exclusion policies, and the spread of the exotic white pine blister rust (Schwandt 2006; Tomback and Achuff 2010). Losses of whitebark pine in some areas of the Northern Rockies exceed 80 percent (Keane et al. 2012). Whitebark pine is listed as endangered in Alberta, it is a candidate species for listing under the U.S. Endangered Species Act (USFWS 2011), and it is listed as endangered in Canada under the Federal Species at Risk Act. Within the last decade, major outbreaks of pine beetle and increasing damage and mortality from blister rust have resulted in cumulative whitebark pine losses that have altered high-elevation community composition and ecosystem processes in many regions of the United States and Canada.

Climate Change Responses

There is much disagreement in the research and management communities about the fate of whitebark pine as climates slowly warm. Some maintain that projected warmer conditions will severely reduce whitebark pine habitat and push whitebark pine “off the tops of mountains” (Lenoir et al. 2008) or restrict the species to north of the Canadian border (Koteen 1999; Schrag et al. 2007; Warwell et al. 2007). This assumes that less hardy, shade-tolerant conifer species would establish in those higher elevation stands where whitebark pine currently dominates, and whitebark pine would “migrate” upslope to the limited areas above its current elevational range (Romme and Turner 1991). Bell et al. (2014), using SDMs, project minor losses (10–20 percent) in whitebark pine range in the Northern Rockies by 2090. Others hold that climate-mediated changes in the disturbance regimes will serve to keep whitebark pine within its current range, albeit at lower levels (Loehman et al. 2011b). The fate of whitebark pine is uncertain because of high uncertainty in regional climate change predictions, the high genetic diversity and resilience of the species, and the localized changes in disturbance regimes and their interactions (Keane et al. 2015a).

Climate change has the potential to significantly impact whitebark pine ecosystems (Bartlein et al. 1997). Devine et al. (2012) rated whitebark pine the most vulnerable of all the Pacific Northwest tree species, primarily because of restricted range and white pine blister rust infections. Taking a historical perspective, however, we can see that whitebark pine was able to persist through many major climatic cycles in the past. Historical analogs of warmer climates in the paleoecological record indicate whitebark pine was maintained and even increased in some places under past warmer and drier climates in parts of its range (Whitlock and Bartlein 1993; Whitlock et al. 2003). Whitebark pine can grow within a broad upper-elevation zone in the West; it just happens to grow best at high elevations where there is little competition from other tree species. For example, Arno et al. (1995) found that the elevational range of whitebark pine in the Bitterroot Mountains of Montana extended more than 500 feet below its current lower elevation limits. Nitschke and Innes (2008) suggested that temperature alone would exclude whitebark pine from British Columbia landscapes. However, whitebark pine occupies the largest range of any five-needle pine in the United States and Canada—about 18° of latitude and 21° of longitude—indicating a great deal of tolerance to different climates (Tomback and Achuff 2010). Because it is bird dispersed, it is planted and grows in many environments and dies only from competition (Arno and Hoff 1990). Moreover, its longevity provides potential buffering against changing climates (Morris et al. 2008).

The same three responses of tree species to climatic change will occur for whitebark pine: Ranges will decline, stay the same, or expand. SDM studies have projected dramatic decreases in whitebark pine habitat over the next 50 years (McDermid and Smith 2008; Warwell et al. 2007). Hamann and Wang (2006) projected a 100 percent decline in whitebark pine in British Columbia with high levels of warming. These models also predict that whitebark pine will probably make a transition to treeline environments that are above the current elevational range, but these transitional areas are much smaller than the traditional range of whitebark pine, thereby resulting in a net loss of the species. Climate can adversely affect growth and mortality of whitebark pine in many ways (Bugmann and Cramer 1998; Keane et al. 2001). Projected decreases in water availability may result in less water being available for some droughty sites. Longer drought might cause whitebark pine to shut their stomata longer to conserve the little water available, resulting in slow growth.

However, many whitebark pine stands may have positive responses to warming climates. Anecdotal evidence shows that some whitebark pine forests are exhibiting abnormally high growth and more frequent cone crops with warmer summers and longer growing seasons. These observations are consistent with some region-based scenarios using computer modeling (Loehman et al. 2011b). Recent modeling efforts have shown that whitebark pine might be maintained on the landscape in the future, provided that projected increases in large, stand-replacement fires do occur and create large, competition-free burned areas (Clark et al. 2017; Loehman et al. 2011b). If tree dispersal enables range shifts to occur, this will lead to a new northern distributional range of whitebark pine (Hamann and Wang 2006; McKenney et al. 2007). Moreover, whitebark pine shows promise for being maintained in the Northern Rockies because of high levels of genetic diversity (Mahalovich and Hipkins 2011; Richardson et al. 2002), moderate to high heritabilities in key adaptive traits, demonstrated blister rust resistance (Hoff et al. 2001; Mahalovich et al. 2006), minimal inbreeding (Bower and Aitken 2007; Mahalovich and Hipkins 2011), and generalist adaptive strategies.

Future climates may enhance growth of whitebark pine diameter and height and decrease mortality, especially in mesic seral whitebark pine forests. Earlier growing seasons with ample moisture, such as those projected for the upper subalpine forests, will result in increased productivity and greater growth. Longer, warmer growing seasons may also result in higher productivities and greater biomass, especially considering the high amounts of precipitation that currently fall in upper subalpine forests. The abundant moisture may enable longer growing seasons at high elevations. Increased biomass could result in higher growth rates for timber production and forage, especially in the widespread higher mountain areas where cold, not moisture, limits tree growth, creating potential for the inclusion of whitebark pine in the timber base. Increased biomass could also foster more intense, severe fires, and maybe insect and disease outbreaks, but more importantly, the increased biomass will probably increase cone crop abundance and frequency. However, this increased production may also heighten competitive interactions between whitebark pine and its associated species, thereby favoring the more shade-tolerant individuals in the absence of disturbance. If disturbances increase, however, the more fire-tolerant whitebark pine might inherit the landscape.

Whitebark pine cone and seed crops could be both adversely and beneficially affected by climate change. In high-elevation, historically cold environments, increased temperatures may increase growing seasons and thereby increase potential for more frequent and more abundant cone crops with greater numbers of seed. This is important because decreases in species abundance and associated cone production may be offset by climate-driven increases in cone crops. Warming and variability in climate will also affect the phenology of cone crops, but these impacts may be minimal as plants adapt to the new conditions. Some predict higher frost mortality of emerging cones due to earlier onset of the growing season, coupled with high daily temperature variability (Chmura et al. 2011). Others suggest that cone crops will be reduced in the future because of high tree stress from drought, resulting in less frequent and abundant cone crops. Many expect that changes in climate variability and timing will have low impact on species reproduction because whitebark pine is both drought-tolerant and cold-tolerant.

Perhaps the life stage most critical for whitebark pine is regeneration, where most species, but especially trees, are most susceptible to shifts in climate (Solomon and West 1993). The microsite conditions needed for successful regeneration are so demanding that seed germination, especially from seeds that are wind dispersed, is rarely successful (McCaughey and Tomback 2001). Bunn et al. (2003) emphasized the importance of accounting for microsite variability in assessing climate change response; high-elevation microsite changes, coupled with increased fire activity, could increase whitebark pine regeneration and growth as climates change. The depth and duration of snow cover often governs high-elevation tree regeneration. Most years are moist enough for regeneration, but snow remains on sites for a long time, thereby limiting the number of days that a seedling can actually photosynthesize and grow. If temperatures increase, then snow might melt earlier, giving more time for seedlings to survive and grow. Warm years often result in waves of regeneration and can be dated in upper subalpine ecosystems by using seedling and sapling tree ages (Little et al. 1994; Rochefort et al. 1994). Recent observations of invasions of subalpine meadows and balds by subalpine fir, alpine larch, and spruce are a testament to a high number of sequential warm years over the last decade, which have facilitated regeneration in the high mountain landscape. Moreover, there is often abundant precipitation in upper subalpine settings, and projections for the future indicate roughly the same amount, so seedling mortality from drought might continue to be minimal.

Many climate change studies consistently project drier conditions in the range of whitebark pine, which would result in large increases in the annual number and severity of wildfires and area burned (Flannigan et al. 2009; Krawchuk et al. 2009; Marlon et al. 2009). With increased fire, whitebark pine will have a unique opportunity to maintain its range or even increase in distribution in the future because it has bird-mediated seed dispersal mechanisms that can disseminate seed great distances into large, severe burns, well before wind can disperse the seeds of its competitors (Tomback 1977, 1982, 1989; Lorenz et al. 2008). Whitebark pine also has morphology that enables it to survive low- to moderate-severity fires (Ryan and Reinhardt 1988). Therefore, whitebark pine is uniquely positioned as a species that can increase under the more frequent fire regimes that result from warming climates. Further, nutcrackers may be harvesting seeds from trees that have survived blister rust, so there is some chance that seeds from unclaimed nutcracker caches may become blister rust-resistant trees. It is entirely possible that as long as wildland fire creates areas where birds will cache seeds and resultant seedlings can grow without competition, whitebark pine will continue to thrive throughout its range.

Current mountain pine beetle outbreaks are killing more whitebark pine than historical records indicate, and these outbreaks are probably a result of warmer winter temperatures that facilitate expansion of and establishment of beetle populations in the higher elevation whitebark pine zone (Logan and Powell 2001; Logan et al. 2003). A warmer climate may also accelerate the spread of blister rust (Koteen 1999).

In summary, whitebark pine is not expected to do well under future climates, not because it is poorly adapted to shifts in climate regimes, but rather because it is currently undergoing major declines from the exotic disease white pine blister rust that preclude its immediate regeneration in future burned areas. Moreover, the declines from white pine blister rust and mountain pine beetle have served to reduce whitebark pine populations to severely low levels, and now the nutcracker is acting more as a seed predator than a seed disperser (Keane and Parsons 2010). Climate shifts will only exacerbate this decline and complicate restoration efforts. Whitebark pine will be highly exposed to any climate changes because of its (1) confined distribution to the upper subalpine environments, (2) severely depressed populations, and (3) lack of ability to regenerate when populations are low because of nutcracker predation. The species has the genetic capacity to overcome both white pine blister rust and new climates to thrive over the next century, but only with extensive restoration efforts.

Subalpine Fir

Autecology

Subalpine fir (*Abies lasiocarpa*) grows in the coolest and wettest forest areas of the western continental United States (Alexander et al. 1990). Although widely distributed, it grows within a narrow range of mean temperatures (25 to 40 °F); however, January temperatures average 5 to 25 °F. In contrast with other subalpine species, cool summers, cold winters, and deep winter snowpack are more important than precipitation in determining where subalpine fir grows.

Subalpine fir occupies the lower valleys to upper subalpine zone in the Northern Rockies. In the lower valley bottoms and footslopes, it is often associated with grand fir, western larch, Douglas-fir, western redcedar, and western white pine (Pfister et al. 1977). At the mid-subalpine, it is often associated with lodgepole pine, Douglas-fir, and Engelmann spruce; at the upper subalpine, it is associated with whitebark pine, alpine larch, mountain hemlock, and Engelmann spruce (Arno 2001). In the Rocky Mountains, subalpine fir is commonly found with Engelmann spruce, and the two together are often called spruce-fir forests (Moran-Palma and McTague 1997). The subalpine fir habitat types are probably the most common forest habitat types in the Northern Rockies (Pfister et al. 1977).

Subalpine fir is shade-tolerant, and is often the most shade-tolerant of all its associates, except for grand fir and mountain hemlock in isolated cases (Alexander et al. 1990; Minore 1979) (table 6.3). Although subalpine fir can grow under nearly all light conditions, seedling establishment and early survival are usually favored by partial shade (Knapp and Smith 1982). In the absence of grand fir and mountain hemlock, subalpine fir will survive under closed-forest conditions with less light than Engelmann spruce. It may not compete well with the spruces, lodgepole pine, or interior Douglas-fir in the lower subalpine when light intensity exceeds 50 percent of full shade. Subalpine fir is quite intolerant of drought, and many seedlings can be killed if droughts are overly long or deep. The species is highly tolerant of frosts and can remain alive in seasonally wet conditions. Subalpine fir is usually the climax tree species in most subalpine areas of the Northern Rockies, although it sometimes shares climax status with spruce.

Subalpine fir is a prolific seed producer, often having large cone crops every 2 or 3 years (Alexander et al. 1990). Seeds usually drop in late fall, over snow in most places. The species is restricted to cold, humid habitats because of the low tolerance of seedlings to high temperatures and dry conditions (Knapp and Smith 1982); newly germinated subalpine fir seedlings rarely tolerate high solar radiation, and they are susceptible to heat girdling and drought (Little 1992). Seedlings are also killed or damaged by spring frosts, competing vegetation, frost heaving, damping off, snowmold, birds, rodents, and trampling and browsing by large animals, but losses are not different than for any of the common associates of the species (Alexander et al. 1990). However, the abundant seedfall of fir, coupled with cool conditions in the subalpine, often create dense mats of seedlings in stands that contain partial shade and overwhelm seedling establishment of all other species. Subalpine fir has average genetic diversity for a Northern Rockies tree species and weak geographic differentiation. There are strong clinal variations in phenological and morphological characteristics.

Disturbance Interactions

Subalpine fir is highly susceptible to fire damage because of thin bark, low-hanging dense foliage, and shallow roots (Ryan and Reinhardt 1988). Even the lowest severity fire can cause high mortality in subalpine fir. Frequent fires often eliminate subalpine fir from both the overstory and understory, thereby maintaining the more fire-adapted species of lodgepole pine, whitebark pine, western larch, and western white pine (Little et al. 1994; Murray et al. 1997; Wadleigh and Jenkins 1996). Invariably, some fir trees survive even the most severe fires in refugia, thereby providing a seed source for future stands (Murray et al. 1998; Veblen et al. 1994).

In spruce-fir forests, the most important insect pests are the western spruce budworm and western balsam bark beetle. The silver fir beetle (*Pseudohylesinus sericeus*) and the fir engraver may at times be destructive, but only in local situations in the Northern Rockies. Fir broom rust (*Melampsorella caryophyllacearum*) and wood-rotting fungi are responsible for most disease losses, but root and butt rots may be important locally. Broom rust and wood rots weaken affected trees and predispose them to windthrow and windbreak. Decades of intense competition, coupled with a period of moderate to severe drought, often cause extensive mortality in subalpine fir stands. These high mortality events are often attributed to a complex of disease, insects, and other agents, but the underlying cause is low vigor in existing trees from overcompetition.

Historical and Current Conditions

Effects of 100 years of fire exclusion have not yet become manifest in most subalpine fir ecosystems because of historically infrequent fire and slow successional advancement. However, abundance of subalpine fir cover types has increased in many subalpine and upper subalpine landscapes (Keane et al. 1994), and many stands that had low subalpine fir components now have fir dominating the understory and encroaching in the overstory. Increased stand density has resulted in many stands becoming stressed from competitive interactions, heightening susceptibility to disturbances. Recent dry, droughty conditions have led to high-elevation subalpine fir mortality of undetermined origin called subalpine fir die-off, usually attributed to a complex of causal mechanisms such as drought, greater competition, higher temperatures, and increasing diseases. Therefore, as fire is progressively kept off the subalpine landscape, the subalpine fir stands that replaced the pine communities will progressively decline in vigor and be more susceptible to fire, insects, and diseases. A concern is that if these overly dense, unhealthy stands continue to escape fire, the seed sources of the fire-adapted pines will be eliminated, and high elevation sites may be converted to grass and shrublands (Keane 2001). Another concern is that as fire is excluded from these dense forests, canopy and surface fuels will accumulate to such levels that, when they are burned, fires will be of extremely high severities (Keane 2001; Morgan et al. 1994b).

Climate Change Responses

It is challenging to predict responses of subalpine fir to future climate change. It is a species that is highly adapted to moist growing conditions, so it is likely to respond poorly to increasing temperatures and drought (Alexander et al. 1990). On the other hand, it is a fierce competitor that can outcompete all subalpine tree species for shade, and it is a species that has a diverse range throughout the Northern Rockies. Subalpine fir could expand its range into the treeline, become more or less productive in its current range, and decline in productivity and occurrence in those areas that become inhospitable for the species, presumably the warmer, drier portions of its current range. Most paleo-reconstructions over the Holocene show subalpine fir dominated during periods of cold, moist conditions but declined in extent as climates warmed (Brunelle et al. 2005; Whitlock 1993, 2004). Hamann and Wang (2006) projected that future losses of subalpine fir from drought will exceed gains from range expansion, resulting in a 97-percent decrease in the range of the species in British Columbia. Romme and Turner (1991) estimated major to minor losses in subalpine fir in the future in the GYA, depending on degree of warming, and Bell et al. (2014) modeled little loss of subalpine fir in the Northern Rockies. Using SDM techniques, Crimmins et al. (2011) estimated that the subalpine fir/spruce zone would move upward in elevation by 300 feet by 2050. Woodward et al. (1994) suggested that subalpine fir will produce less frequent and lower cone crops in the future. However, seedling establishment may be the bottleneck for subalpine fir in the future; the species needs long periods of high moisture for seeds to germinate and seedlings to thrive (Urban et al. 1993), and years that meet these conditions may be less frequent in the future in the lower subalpine.

In those areas with abundant precipitation, longer growing seasons and reduced snowpacks may increase regenerative success of subalpine fir, especially in subalpine areas where snow pack historically controlled regenerative success (Means 1990; Urban et al. 1993). These areas constitute most of the range of fir in the subalpine to treeline. Little and Peterson (1991) found that most fir regeneration occurred in low snow years because there were more suitable microsites. Villalba et al. (1994) found subalpine fir growth was positively correlated with increasing summertime temperature. Higher productivity in these subalpine forests may also increase cone crops, tree growth, and species densities. Denser stands may eventually result in high competitive stress, making these fir stands even more vulnerable to high mortality from insects, disease, and abiotic factors, and therefore less resilient. However, declines in whitebark pine and lodgepole pine from beetle- and rust-caused mortality may facilitate fir regeneration and growth, resulting in more fir trees and more stands dominated by subalpine fir.

Those Northern Rockies areas in high-elevation (treeline) settings may have an increase in subalpine fir as the heavy snow and cold conditions that precluded fir regeneration at treeline become less frequent (Cayan et al. 2001; Rochefort et al. 1994). However, the decline of whitebark pine trees that act as nurse crops to facilitate subalpine fir establishment may contribute to the inability of the fir to establish in the high elevation settings. Establishment of populations at the upper subalpine and treeline may be possible only when fire is absent long enough to allow enough subalpine fir to gain reproductive maturity; then sufficient seed can be dispersed over enough years to ensure suitable environmental conditions for seedling establishment (Little and Peterson 1991; Little et al. 1994).

Increases in wildland fire would decrease subalpine fir throughout the Northern Rockies, and those decreases would be much more extensive and steeper than any resulting from direct climate change effects. Frequent fires would favor nearly all of the associates of subalpine fir because it is the least adapted to survive fire (Arno and Hoff 1990; Keane

2001). Little et al. (1994) found limited subalpine fir regeneration over 30 years after a fire at Mount Rainier, but those regeneration events that did occur happened after low snow years. Heusser (1998) found that subalpine fir increased in growth with warmer temperatures during the spring of the growth year, but growth was negatively correlated with high summer temperatures the previous years.

In summary, the future of subalpine fir will depend on both the future level of fire and the degree of climatic warming. Subalpine fir is likely to be a species that shifts across the high mountain landscape, with gains in expansion balancing losses of contraction (caused directly by changes in climate). However, future increases in fire, disease, and insects may limit its abundance. Fir is an aggressive competitor, so gains in the species through advanced succession in the upper subalpine will probably be balanced by or exceed losses from the drier, lower subalpine caused by fire, drought, and pathogens.

Engelmann Spruce

Autecology

Engelmann spruce (*Picea engelmannii*) is widely distributed in the western United States and is a major component of the high-elevation Northern Rockies forests (Alexander and Shepperd 1990). It grows in humid climates with long, cold winters and short, cool summers, and occupies one of the highest and coldest environments of the western United States. The range of mean annual temperature in which this species occurs is narrow considering its wide distribution. Engelmann spruce grows best on moderately deep, well-drained, loamy sands and silts and clay loam soils from a variety of volcanic and sedimentary materials. It also grows well on glacial and alluvial soils where the water table is accessible.

Engelmann spruce is rated as shade tolerant, but it is less shade tolerant than its major associate, subalpine fir (Minore 1979) (table 6.3). It is more shade enduring than interior Douglas-fir, western white pine, lodgepole pine, quaking aspen, western larch, or ponderosa pine (Alexander and Shepperd 1990). The species does not tolerate drought well and it is perhaps the least adapted to drought conditions of the subalpine tree species (Alexander and Shepperd 1990). It is highly frost tolerant and one of the few upper subalpine species that can tolerate seasonal standing water. It is associated with mycorrhizae but does not seem to be dependent on the fungi for survival. It is wind pollinated and its seeds are wind dispersed.

In the Northern Rockies, Engelmann spruce is mostly associated with subalpine fir but grows with many other tree species, including mountain hemlock, whitebark pine, western larch, Douglas-fir, quaking aspen, lodgepole pine, limber pine, and western hemlock. In most upland subalpine Northern Rockies sites, Engelmann spruce is a minor species associated with subalpine fir in later seral communities. It is often found in the upper subalpine as scattered individuals with the greatest height and diameter (Arno 2001). Pure Engelmann spruce communities are found in wet areas and riparian settings, and in severe frost pockets where all frost-sensitive tree species are excluded.

Spruce seeds germinate in a variety of substrates, including duff, litter, and decomposed humus, and seedlings have best initial survival on duff seedbeds, rather than on mineral soil. Engelmann spruce has low tolerance to high temperatures and drought, especially in the first 5 years of establishment. Due to its slow initial root penetration and sensitivity to heat in the succulent stage, drought and heat girdling kill many first-year spruce seedlings. Drought losses can continue to be significant during the first 5 years of seedling development, especially during prolonged summer dry periods (Alexander and Shepperd 1990). After establishment, adequate soil moisture, cool temperatures, and shade favor survival.

Engelmann spruce is similar to subalpine fir in that it has an average genetic diversity with weak geographic differentiation. It is considered intermediate in its adaptive strategy, being neither a generalist nor a specialist. Populations show habitat specificity. Engelmann spruce has a high outcrossing rate and possesses the ability to cross with white spruce (*Picea glauca*).

Disturbance Interactions

Engelmann spruce is highly susceptible to fire injury and death, but some spruce survive severe burns because of their large size (Bigler et al. 2005; Wadleigh and Jenkins 1996). The species can survive fire better than its primary associate, subalpine fir (Ryan and Reinhardt 1988). Surviving large spruce trees can provide abundant seed in burned areas, but rarely do these seeds germinate to create forests dominated by Engelmann spruce; these spruce-dominated forests occur only in seasonally wet habitats (Pfister et al. 1977).

Engelmann spruce is susceptible to windthrow, especially after any cutting in old-growth forests. The spruce beetle (*Dendroctonus rufipennis*) is the most serious insect pest of Engelmann spruce. It is restricted largely to mature and overmature spruce, and epidemics have occurred throughout recorded history. The western spruce budworm also attacks Engelmann spruce.

Historical and Current Conditions

It is difficult to determine recent trends in Engelmann spruce forests across the Northern Rockies because the species is rarely the dominant component in a stand. One would expect that advancing succession under a century of fire exclusion has increased spruce in the subalpine and upper subalpine. However, logging and fire have reduced spruce in lower elevation areas, where it occurs in seasonally wet areas and frost pockets.

Climate Change Responses

Similar to subalpine fir, some losses of Engelmann spruce are likely in the drier portions of its range, especially in those seasonal moist sites that will be mostly dry in the future. Liang et al. (2015) found that major mortality events have occurred in Engelmann spruce over the last 20 years because of increased drought, presumably related to changing climate. Alberto et al. (2013) found that this species had lower growth the year following warm, dry years in the U.S. Cascades. Using SDMs, Morales et al. (2015) projected a 47 percent contraction in Engelmann spruce in the northern Rocky Mountains by 2060, and Hamann and Wang (2006) projected a 27-percent decrease in the range of Engelmann spruce in British Columbia by 2050. Gray and Hamann (2013) estimated Engelmann spruce would move 230 miles northward and 550 feet higher in elevation in the Northern Rockies by 2050. Using SDMs for the southwestern United States, Notaro et al. (2012) projected that Engelmann spruce would disappear by 2050. Using gap modeling, Burton and Cumming (1995) projected a precipitous collapse of Engelmann spruce in the mesic forests of British Columbia. Coops and Waring (2011) used mechanistic modeling to simulate a retraction in spruce range of more than 50 percent.

Engelmann spruce may be the first species to become established in high elevation areas where snow precluded conifer regeneration historically, but where there now may be a seasonal wetland or subalpine wet meadow (Schauer et al. 1998). Jump and Peñuelas (2005) note that Engelmann spruce has the genetic capacity to adapt to large swings in climate in situ by taking advantage of shifts in microsites. Due to the great seed dispersal ability and tall stature of the species, it is able to disseminate into previously unforested areas, such as glades, meadows, and balds, to expand its range. Whitlock (2000) found increased spruce regeneration during the warmest periods of the past several centuries, and Luckman et al. (1984) found Engelmann spruce growth positively correlated with increasing summertime temperature. Various SDM approaches project minor changes in the spruce-fir subalpine zone (Bell et al. 2014; Crimmins et al. 2011).

Engelmann spruce is poorly adapted to fire, and thus major declines are expected with the projected increases in area burned. But these declines may be offset by increased regeneration on burned areas with mineral soil substrates. Continued fire suppression activities may maintain spruce on the landscape, but this species may persist at lower abundance because of increased drought. Fire suppression may also ensure the demise of Engelmann spruce in that, when fires eventually burn, the severities may be so great that they kill all spruce seed sources. In addition, Bentz et al. (2010) noted that future climates are likely to be more conducive to the spruce beetle, and Stout and Sala (2003) suggested that future climates may foster more spruce budworm events, leading perhaps to further declines in spruce.

In summary, Engelmann spruce is a species that is highly sensitive to climate but likely to persist in future Northern Rockies landscapes because of the superior ability of the species to seed into new areas, especially burned areas, and ability to remain in the high mountain landscape. Projected increases in subalpine productivity will also serve to keep spruce on the Northern Rockies landscape.

Mountain Hemlock

Autecology

Mountain hemlock (*Tsuga mertensiana*) is usually found on cold, wet, snowy upper subalpine sites, where it grows slowly and sometimes lives to be more than 800 years old. The species is apparently limited by late snowmelt, short growing seasons, and cool summer temperatures throughout much of its range in the Pacific Northwest. Earlier snowmelt, higher summer temperatures, and lower summer precipitation in the lower portions of its range produce conditions under which growth is limited (West et al. 2009). Areas occupied by mountain hemlock generally have a cool to cold maritime climate that includes mild to cold winters, a short, warm to cool growing season, and moderate to high precipitation. The presence of mountain hemlock in the Rocky Mountains is closely correlated with the eastward penetration of moist maritime air masses (Woodward et al. 1994). Mountain hemlock occurs in mixed upper subalpine stands in the western portions of the Northern Rockies, often relegated to the moist north slopes.

Mountain hemlock is considered highly tolerant of shade and other forms of competition, and it is probably more tolerant than any of its Northern Rockies associates, even subalpine fir in some places (Minore 1979). Mountain hemlock is considered a minor climax species in most of its limited habitats; mountain hemlock often succeeds lodgepole pine or subalpine fir (Means 1990). The species has a low tolerance to drought but a high tolerance to frost and standing water. It is wind pollinated, and the seeds are wind dispersed.

Mountain hemlock has large cone crops about every 3 years. It reaches reproductive maturity quickly by around 20 years of age, and most of the seedfall occurs during the fall months. It has average genetic diversity and weak geographic differentiation, but moderate population differentiation.

Disturbance Interactions

Mountain hemlock is considered susceptible to fire because it often retains branches almost to the ground, grows in clusters, and often has shallow roots spread throughout well-developed forest floors that dry out in the summer (Dickman and Cook 1989). It has thick bark and can withstand some low-intensity fire, but overall, it will often succumb to fire damage over time. Fire is a rare visitor to these mesic, cold stands, so any increase in fire frequency will reduce mountain hemlock populations.

The most striking damage to mountain hemlock is probably that caused by laminated root rot. This fungus spreads from centers of infection along tree roots so that all trees are killed in circular areas that expand radially. Mountain hemlock is the species most susceptible to root rot in subalpine forests (Means 1990).

Historical and Current Conditions

There have been few evaluations of mountain hemlock distributions in the Northern Rockies, and thus it is difficult to gauge trends in this species over the last century. The fire exclusion era has advanced succession in those subalpine sites where mountain hemlock can be found, thereby increasing the numbers and density of the species. These dense forests are probably not currently stressed because of high productivity in these areas. Mountain hemlock has a limited range in the region, so any significant warming and drying could drive the species to local extinction.

Climate Change Responses

With warming climate in western North America, existing mountain hemlock forests will probably increase in productivity, especially in the upper and lower elevational boundaries of the mountain hemlock zone (Means 1990). Near Mount Baker, Washington, ring width of mountain hemlock increased with increasing monthly temperatures in the preceding 12 months, decreasing winter precipitation, and decreasing snow depth, implying that productivity should increase with predicted temperature increases (Woodward et al. 1994). Graumlich et al. (1989) estimated that productivity increased 60 percent in the last century in four high-elevation stands in Washington, three of which contained 48 to 96 percent mountain hemlock. They related this increase most strongly to the increase in growing season temperature during this period (about 2.7 °F). West et al. (2009) projected increases in growth and productivity of mountain hemlock throughout much of its range in Washington and northern Oregon, but increased summer drought stress will reduce productivity in mountain hemlock forests of southern Oregon and near the lower elevation limit of the species. Peters and Lovejoy (1992) estimated that if mean annual temperatures increase 4.5 °F, the mountain hemlock zone may be shifted 1,800 feet higher in elevation and decrease as a proportion of forestland from 9 percent to 2 percent in Oregon. An increase of 9.0 °F may induce an upward shift of 3,700 feet, which is above all but the tallest peaks, effectively eliminating the species from the Northern Rockies (Means 1990). Woodward et al. (1994) speculated the mountain hemlock will produce less frequent and lower cone crops in the warm future.

In summary, the high productivity of mountain hemlock sites in the western Northern Rockies may mitigate the potential decline of mountain hemlock over the next 50 years. Local shifts of the species are likely to occur where it is established in droughty low-elevation areas of Idaho and western Montana, but overall, the species might be somewhat stable under climate warming. It might even increase in productivity and make range expansions into the lower treeline. The species is not highly exposed to climate changes because of its limited range in the region and its somewhat confined niche. The magnitude of climate effects on mountain hemlock responses will be great, but mostly positive, and the likelihood of these effects has a high uncertainty, primarily because of the uncertain role that fire will play in the dynamics of mountain hemlock populations in future landscapes.

Alpine Larch

Autecology

Alpine larch (*Larix lyallii*) is a deciduous conifer that occupies the highest and most remote environments in the Northern Rockies, growing in and near treeline on high mountains across the inland Pacific Northwest (Arno 1990). In the Rocky Mountains, alpine larch extends from the Salmon River Mountains of central Idaho northward to Lake Louise in Banff National Park, Alberta (Arno 1990). Within this distribution, alpine larch is common in the highest areas of the Bitterroot, Anaconda-Pintler, Whitefish, and Cabinet Ranges of western Montana. It is also found in lower abundance in isolated stands atop many other ranges and peaks in western Montana and northern Idaho (Arno and Habeck 1972). Alpine larch grows in cold, snowy, and generally moist climates where for more than half of the year, mean temperatures are below freezing. Mean annual precipitation for most alpine larch sites is between 32 and 75 inches, the larger amount being more prevalent near the crest of the Cascades; most stands in the Montana Bitterroot Range receive 39 to 59 inches. About 75 percent of this precipitation is snow and sleet. Ridgetop alpine larch stands are exposed to violent winds; most alpine larch stands annually experience winds reaching hurricane velocity or more, especially during thunderstorms or during the passage of frontal systems.

Alpine larch is perhaps the most drought-susceptible conifer in the Northern Rockies because of its reliance on subsurface water during the dry summer months (Arno 1990). It achieves its best growth in high cirque basins and near the base of talus slopes where the soils are kept moist throughout the summer by aerated seep water. It can also tolerate boggy wet meadow sites having very acidic organic soils. The species is most abundant on cool, north-facing slopes and high basins, where it forms the uppermost band of forest. It also covers broad ridgetops and grows locally under relatively moist soil conditions on south-facing slopes. The Northern Rockies may have a droughty period for a few weeks in late summer, but the effect is minor in most alpine larch sites; however, dry surface soils may prevent seedling establishment in certain years.

Alpine larch is the most shade-intolerant conifer growing at high-elevation Northern Rockies sites and is classified as very intolerant (Minore 1979) (table 6.3). Its evergreen associates attain their best development in forests below the lower limits of larch. An exception is whitebark pine, another treeline inhabitant, which is most abundant on warm exposures and microsites and thus tends to complement rather than compete with larch. Alpine larch grows mostly in pure stands, but it can be found with whitebark pine, subalpine fir, and Engelmann spruce near their upper limits. Alpine larch roots extend deep into fissures in the rocky substrate. Trees are well anchored by a large taproot and large lateral roots and are very windfirm. Alpine larch is easily replaced by subalpine fir in most upper subalpine sites, but the species can act as a climax species in the lower treeline (Arno and Habeck 1972).

Alpine larch is one of the few deciduous conifers in the Northern Rockies, and as a result, it has a high capacity to survive wind, ice, and desiccation damage during the winter because the needles are off the trees. The species also has an evergreen sapling stage that allows it to quickly take advantage of the short growing season in the early summer after snowmelt (Arno and Habeck 1972). Alpine larch seedlings are frost-tolerant. Mature trees produce good cone crops every fifth year, and these seeds drop in early fall. It is assumed that alpine larch has an average genetic diversity and weak geographic differentiation, but little genetic work has been done on this species.

Disturbance Interactions

Fire is an occasional but localized visitor in alpine larch stands, causing injury or death in most cases. Large fires are infrequent in the cool, moist, and rocky sites where alpine larch occurs, and fire spreads poorly on these sites because of light and discontinuous fuels. Unlike its thick-barked, fire-resistant relative western larch, alpine larch has thin bark, has low resistance to surface fire, and often dies after low-intensity fires (Ryan 1998).

Powerful winds in alpine larch stands often damage crowns, in conjunction with loads of clinging ice or wet snow (Arno and Habeck 1972). Nevertheless, the deciduous habit and supple limbs of this tree make it more resistant to wind damage than its associates. Death usually occurs when advanced heart rot has so weakened the bole that high winds break off the trunk. Brown trunk rot produces the only conks commonly found on living trunks (Arno 1990); this fungus is evidently the source of most heart rot. Snow avalanches and snowslides are an important source of damage in many stands, but again, this species is better adapted to survive these disturbances than its evergreen associates.

Historical and Current Conditions

This species is rarely studied, and as a result, very little is known about its population trends in the Northern Rockies. Our best guess is that alpine larch populations have stayed roughly the same across most of its range in the region over the last 100 years. There have been some losses from fire in some areas, especially the Bitterroot Mountains of Montana, but there have also been gains. Recent anecdotal observations indicate that alpine larch has been increasing in ribbon forests, glades, and high-elevation open areas where snow accumulated historically; over the last two decades, these areas have been clear of snow enough of the year to foster alpine larch regeneration.

Climate Change Responses

We think alpine larch has a high potential to decrease in both productivity and abundance with climate change. On the one hand, alpine larch is a shade- and drought-intolerant species, so it does not do well in areas of increasing dryness and competition (Arno and Habeck 1972). Its high demand for subsurface water during the growing season is apparently the main factor limiting its range in the Northern Rockies (Arno 1990), making it highly susceptible to increasing drought in the future. Alpine larch growth is highly correlated with high snowpack, especially in April, which is usually indicative of high subalpine moisture throughout the year (Colenutt and Luckman 1991; Peterson and Peterson 1994). The lack of summertime groundwater would be more likely in the southern part of the species range in the Northern Rockies, specifically western Montana and central Idaho.

On the other hand, alpine larch can produce copious amounts of seed that may land on upper subalpine and timberline areas that were historically covered with snow for most of the year but in the future may be sufficiently snow-free to allow wind-dispersed seed to germinate and grow into viable seedlings. There is ample anecdotal evidence documenting alpine larch encroachment into ribbon forests, glades, and snowfields. These seedlings could become mature trees, provided there is sufficient moisture. With higher rates of productivity in a warming climate, seedlings and trees may have greater growth and cone production. However, short-term increases in alpine larch regeneration may be offset by the high variability in drought in the upper subalpine, which may eventually cause declines in the larch. Another possibility is that the more shade-tolerant subalpine conifers, such as spruce and subalpine fir, might become established in these new open areas and outcompete alpine larch for dominance. Along those same lines, the more drought-tolerant whitebark pine may also become established in the snow-free areas and survive the anticipated long droughts.

Although alpine larch apparently lacks the morphological, ecophysiological, and genetic capacity to adapt to new environments, it does have the ability to genetically intergrade with western larch to produce hybrids that may be more tolerant of drought and competition (Carlson et al. 1990). Moreover, its superior seed dispersal capability may allow it to become established in treeline areas made environmentally favorable by climate change, mainly from decreasing snowpacks and higher temperatures. These areas, however, may be significantly smaller and more isolated than areas in its current range, where it will decline because of lack of water.

Alpine larch is not well adapted to survive wildland fire (Arno 1990), and as its existing range becomes drier and fires become more probable, it is expected that more alpine larch will burn, providing there are sufficient fuels. Those upper subalpine forests that are co-dominated by whitebark pine and alpine larch are probably the most susceptible of subalpine larch habitats to increases in fire. Rocks, scree, and fuel-free areas protect many alpine larch communities, so it may be years before fire, or the more shade-tolerant competitors, invade pure alpine larch woodlands (Arno and Habeck 1972). Although alpine larch does not seem to be impacted by major insects and pathogens (Arno 1990), future climates may increase the possibility that insects and diseases that were relatively minor in the past could become more significant in the future, especially in timberline environments where damaging agents were depressed by cold (Woods et al. 2010).

In summary, alpine larch is one of the most susceptible tree species in the Northern Rockies region to climatic shifts that result in increasing drought and fire. Its exposure to climate change is likely to be high because upper subalpine areas may experience the greatest climatic change (Luce et al. 2013). Due to its specialized habitat, alpine larch has the highest risk for major range shifts, and impacts to the species may be great if there are insufficient environments created upslope.

Green Ash

Autecology

Green ash (*Fraxinus pennsylvanica*) is the most widely distributed of all the American ashes (Kennedy 1990), but primarily occurs in the eastern and central United States. In the Northern Rockies, green ash is restricted to the northern Great Plains, which is the northwestern edge of its range (Girard et al. 1987; Lesica 2009). Naturally a moist bottomland or streambank tree, it is well adapted to climatic extremes and has been widely planted in the Plains States and Canada. It is probably the most adaptable of all the ashes, growing naturally on a range of sites, from clay soils subject to frequent flooding and overflow, to sandy or silty soils where the amount of available moisture may be limited. In the northern Great Plains, green ash grows best on fertile, moist, well-drained alluvial soils, primarily along river bottomlands and woody draws. It also occurs in broad upland depressions and ridges, which have subsurface water early in the growing season (Girard et al. 1987; Lesica 2003; Lesica and Marlow 2013). Natural stands of green ash are almost completely confined to bottomlands, but the species grows well when planted on moist upland soils. It lines the watercourses in the western parts of its range where rainfall is insufficient to support upland growth.

Green ash varies from intolerant to moderately shade-tolerant in woody draws. It is an early-seral species, colonizing alluvial soils. It regenerates from seed when exotic grass (which inhibits germination through competition for soil moisture) is absent or has low cover (Lesica 2003; Lesica and Marlow 2013). It also regenerates vegetatively through stump sprouting. Uresk and Boldt (1986) reported 90-percent sprouting success following trunk removal in an experimental study in North Dakota. Lesica (2009) found that stump sprouts can achieve full tree height in 20 years. Although green ash is generally drought-tolerant, prolonged drought may affect regeneration success because it is at the most arid edge of its range (Severson and Boldt 1978). Kennedy (1990) found that a population from the arid, northwestern part of the green ash range was more drought-resistant than one from the moister central Great Plains and northeast.

Disturbance Interactions

Green ash is easily killed by fire, but stumps of most size classes of green ash sprout readily after both fire and mechanical trunk or stem removal (Lesica 2009). The species has several insect and disease agents; it is particularly susceptible to white stringy heartroot (*Perenniporia fraxinophila*), which weakens the plant and makes it more susceptible to wind or ice breakage (Lesica and Marlow 2013).

Historical and Current Conditions

Very little is known about the range expansion or contraction of green ash. However, green ash in the Northern Rockies is at the northwestern (most arid) edge of its range (Lesica 2009), and evidence suggests that many of the green ash communities on the western fringe of the northern Great Plains are declining (Boldt et al. 1978; Lesica 1989, 2001).

Climate Change Responses

Green ash has a broad ecological amplitude and can survive droughty conditions, but it grows optimally on moist sites. As soil moisture declines with a warmer, drier climate, marginal sites may become less favorable for regeneration and survival of young green ash trees. With increases in fire frequency, there will probably be increased vegetative regeneration and decreased production of seedlings following fire; fire often kills green ash seed on or near the soil surface, restricting seedling recruitment to surviving seed-producing trees. Green ash may benefit from increased temperatures because seedling and mature tree growth may increase with increasing soil temperatures. However, those green ash populations associated with moist upland microsites (e.g., northeast-facing residual snow-loaded depressions) may suffer severe drought stress as snowpack declines and melts sooner, and regeneration may decrease, eventually resulting in loss of those communities.

Most mature green ash communities are somewhat resistant to wildland fire, given that the species can sprout afterward, so the projected increases in fire in the future may not impact most green ash stands, especially the moist communities. Low-severity fires might promote regeneration by thinning stands and stimulating sprouting; green ash has both root crown and epicormic sprouts, and both are typical following fire events, especially in the woody draws and riparian areas of the Great Plains. High-severity fires, however, may result in mortality. Browsing pressure on green ash communities is also likely to increase with increased drought, as upland grasses and forbs desiccate and senesce earlier, or are replaced by invasive, less palatable species.

Cottonwood

Autecology

Black cottonwood (*Populus trichocarpa*) is the largest of the American poplars and the largest hardwood tree in western North America (Debell 1990). Narrowleaf cottonwood (*P. angustifolia*) and black cottonwood grow primarily on seasonally wet to moist open canopy sites (typically along streams and rivers) in the western portions of the Northern Rockies. Plains cottonwood (*Populus deltoides*) occurs in eastern Montana and the Dakotas portion of the region. All three species typically occupy fluvial surfaces along floodplains of streams and rivers.

Cottonwoods typically dominate riparian communities on alluvial sites at low to mid-elevations. Various riparian shrubs (e.g., willows [*Salix* spp.], alder [*Alnus* spp.)], birch [*Betula* spp.], dogwood [*Cornus* spp.)]) and a variety of graminoids and forbs occur in the understory of cottonwood stands (Merigliano 2005). Cottonwood is very shade intolerant, and conifers (e.g., Douglas-fir, Rocky Mountain juniper [*Juniperus scopulorum*], ponderosa pine, Engelmann spruce, subalpine fir) may encroach and become dominant in upland cottonwood forests (typically on river and stream terraces). The species is also drought intolerant, and requires an accessible water table (i.e., free, unbound water) during most of the growing season (Rood et al. 2003). Older cottonwood individuals can reach very deep water tables. Plains cottonwood is probably more able to extract water in the unsaturated zone once the water table has dropped below the extent of the roots (Merritt et al. 2010). The finer textured soils in the northern Great Plains hold more water, but it is harder to extract due to the finer soil texture. However, plains cottonwood has apparently adapted to extract water and is likely to be more resilient to drought than the other species.

For all three species of cottonwood, high streamflows are required for successful seedling establishment; the associated scouring action and deposition of fresh alluvium creates optimal surfaces for germination. All species of cottonwood are prolific seed producers, and the windborne seeds disperse widely once the catkins have matured and seeds are released. Seeds are viable for only about 2 weeks, and thus timing of seed release and recession of flood flows is essential to successful germination (Malanson and Butler 1991). Black and narrowleaf cottonwood seedlings are usually established on a yearly basis, depending on flood frequency, timing, and duration. Plains cottonwood establishment is less frequent and more episodic because flows are more variable in both magnitude and frequency. Scott et al. (1997) found that about 72 percent of the plains cottonwoods along the Missouri River of eastern Montana established after a very large flood event (flow >1,800 cubic yards per second with a recurrence interval of 9.3 years). High numbers of seedlings become established in the first year after a flood, but they naturally thin out up to several years later, if they have not been scoured away by high flow events. The young seedlings and saplings that survive are frequently injured and sometimes killed by unseasonably early or late frosts (Debell 1990).

Disturbance Interactions

Cottonwood is mildly fire tolerant owing to its thick bark, high branches, and foliage that is too moist to burn in most years. It is considered a weak stump sprouter, but unlike aspen, cottonwood rarely regenerates from suckers (Brown 1996). Gom and Rood (1999) found that black and narrowleaf cottonwood were more successful at coppice (stump) sprouting and suckering than plains cottonwood. Cottonwood is able to survive low-intensity fires in the short term, but fire injuries can lead to the introduction of diseases that weaken and perhaps kill the tree (Borman and Larson 2002).

Although several insects attack cottonwood, none has yet been reported as a pest of economic significance. Tent caterpillars (*Malacosoma* spp.) are the most important foliar feeders that affect the Northern Rockies. At least 70 fungal species cause decay in cottonwood, but only 6 fungi cause significant losses; 2 of these—brown stringy heart rot (*Spongipellis delectans*) and yellow laminated butt rot (*Pholiota populnea*)—cause 92 percent of the loss. Russian olive (*Elaeagnus angustifolia*) and saltcedar (*Tamarix ramosissima*) are aggressive invasive trees that often outcompete plains cottonwood, particularly during or following drought (Shafroth et al. 2002). These species, along with invasive herbaceous species, are a threat to cottonwoods in general.

Historical and Current Conditions

Black cottonwood was common throughout the Columbia River watershed in Lewis and Clark's day, and can still be found today, but it is greatly reduced in extent.

Climate Change Responses

As snowpack declines and melts earlier with warming temperatures, there will be reduced, attenuated river flows (loss of extreme high and low flows), along with a possible shift in timing of peakflows to earlier in the season, before cottonwood seed is viable for germination. These shifts in timing, magnitude, and variability may result in both decreased germination and establishment of young cottonwoods (Whited et al. 2007). Human demand for water is likely to increase in the future, which will probably result in creation of additional diversions and reservoir expansions. Any alteration of hydrologic flow regime (i.e., timing, magnitude, and duration) will affect both floodplain interaction and water available to cottonwoods, which in turn may reduce recruitment and establishment of seedlings (Auble and Scott 1998; Beschta and Ripple 2005). Decreased streamflows and floodplain interactions may result in a conversion of streamside vegetation from cottonwood to upland species, along with reduced growth and regeneration (recruitment) and increased mortality of cottonwood (Beschta and Ripple 2005). Upland conifers (e.g., Engelmann spruce, lodgepole pine, and Douglas-fir) typically establish once the stream and local water table have dropped, and they can shade out the remaining cottonwoods. In addition to competition from upland conifers, there may be increased browsing pressure on cottonwoods, which will further contribute to declines in cottonwood regeneration and recruitment.

Plains cottonwood may be more persistent under a changing climate because of greater plant-available soil water in the unsaturated zone (as a result of finer textured soils) in its habitat. Black and narrowleaf cottonwood typically occur in coarser substrate, which will become much drier as flows are lower and recede earlier than in the past, or are attenuated due to diversions. Seedling and sapling mortality may increase in these species. Plains cottonwood regeneration occurs with episodic flooding, whereas black and narrowleaf cottonwood regenerate with 1- to 3-year bankfull flow return intervals (typically an annual recruitment cycle); therefore, plains cottonwood will probably be better adapted to irregular flows that may occur with climate change. Black and narrowleaf cottonwood are likely to be at greater risk to changing climate because of soil water characteristics in their habitats and their narrow amplitude in terms of germination and flood events on specific fluvial surfaces.

Vegetation Types

Vegetation types are broad species assemblages that are used to identify the geographic distribution of vegetation in the Northern Rockies. Vegetation types are different from species in that species can be a major to minor component in a vegetation type, but vegetation types can be composed of a number of species. Here we describe generally the likely response of forest vegetation types in the Northern Rockies to climate change. This section is less detailed than the section on species because readers can refer to the individual species for each vegetation type as presented in the previous section.

Dry Ponderosa Pine-Douglas-fir Forests

Ecology

Dry ponderosa pine and Douglas-fir forests are the driest forests in the Northern Rockies. These forests are relatively rare in northern Idaho, more common in western Montana, and prominent in central and eastern Montana where moisture is most limited. They are often found at the foothills of mountain ranges in the region, but also in extensive flatlands bordering perennial grasslands and shrublands. Historically, frequent fires often maintained pure to mixed ponderosa pine woodlands and savannas in areas currently occupied by this type. However, fire exclusion has led to increased tree density and abundance of Douglas-fir, making these forests susceptible to uncharacteristically severe fire.

Disturbance Interactions

These forests recover from disturbance by slowly shifting from fire-tolerant pioneer species to less fire-tolerant and shade-tolerant “climax” species over time. This successional process can occur over 200 to 1,000 years. Ponderosa pine is often able to colonize the hot dry surface conditions of a disturbed site. Over time, as it matures, it provides a shaded environment where less heat-tolerant Douglas-fir can establish. In a frequent low-severity fire regime, the thick-barked ponderosa pine survives fire, whereas the thinner barked Douglas-fir and ponderosa pine seedlings do not. If frequent fires are sustained, the ponderosa pine forest can develop into large patches of open-grown old growth, intermixed with relatively small openings that can persist for centuries. During a cool wet climatic period, or through fire suppression, Douglas-fir or denser ponderosa pine can become established. The increased biomass and structural heterogeneity of these denser forests allow fires to develop into active crown fires that return the site to the initial stand establishment phase. If fires burn these areas again, forest establishment may be limited because of loss of seed source, limited soil moisture, and high surface temperature.

Historical and Current Conditions

A century of fire exclusion, coupled with extensive logging and grazing, has changed these open dry woodlands to closed, dense forests that are often dominated by Douglas-fir.

Climate Change Responses

This vegetation type may be reduced in some areas of its current range under a changing climate because of dry, hot conditions. However, this type may expand into the mixed mesic forest type (next subsection), especially on south slopes, as drought increases. This forest type will probably be the most dynamic in the future, with many of the current areas of this type seeing losses in Douglas-fir, balanced by gains in ponderosa pine. Dry Douglas-fir communities that are currently too cool to support ponderosa pine may support more ponderosa pine with warming climate. Fire exclusion in this type has increased forest density and accumulation of surface fuels; both conditions are likely to support high-severity fires in the next century (Keane et al. 2002).

Western Larch Mixed Mesic Forests

Ecology

Western larch mixed-conifer forests, found in northern Idaho and northwestern Montana, evolved under a combination of moist air masses from the west and cold air masses from Canada, resulting in a patchy forest condition with a mixture of western larch, ponderosa pine, lodgepole pine, Douglas-fir, and spruce subalpine forests. Western larch is most prominent on cooler, moist topographic positions (Touzel 2013); thus, the influence of a warming climate may change the potential distribution of western larch to the more northerly aspects with soils most capable of retaining needed moisture during the growing season (Rehfeldt and Jaquish 2010).

Disturbance Interactions

These forests evolved under a mixed-severity fire regime, which produced a diverse pattern of shade-intolerant western larch, ponderosa pine, lodgepole pine, and Douglas-fir. High-severity fire was more common on moist and cool sites, and produced very large burn patches, often with legacy western larch (Marcoux et al. 2015). Under fire exclusion, many of these forests have become denser, putting them at risk to high-severity fire (Arno 2010; Harrington 2012; Hopkins et al. 2013).

Historical and Current Conditions

In the past, wildland fire maintained extensive stands of western larch across Montana and Idaho. Due to its great value as a timber species, many older stands of western larch were harvested across much of the Northern Rockies, and these forests were often planted back to western larch after extensive site treatments. With fire exclusion, succession advanced and western larch was replaced with mixed stands of lodgepole pine, Douglas-fir, and grand fir.

Climate Change Responses

The western larch mixed-conifer forests of northern Idaho and northwestern Montana are a forest type that has been changing and is likely to continue to change. Fire exclusion, coupled with climate change, will probably continue to reduce western larch and increase the more shade-tolerant Douglas-fir, grand fir, and subalpine fir in some areas. Continued fire exclusion will result in further accumulation of surface and canopy fuels, and coupled with hotter and drier conditions with climate change, will put these forests at risk of high-severity fire. High mortality of the seed-bearing western larch and ponderosa pine may result.

Some attributes of this cover type may serve to make it more resilient in the future. Western larch is not susceptible to the wide range of insects and diseases common to its associated tree species, and it has the best ability to survive fire of all its tree associates. As such, it makes an excellent candidate to feature in management to increase resilience. However, western larch mixed mesic forests are one of our most vulnerable forests to climate change impacts, mainly because past land management has made natural western larch forests susceptible to the damaging effects of uncharacteristically high-severity wildfires. Management of these forest types to create stand and landscape conditions within the historical range of variability are likely to increase resilience to climate change.

Mixed Mesic Western White Pine-Western Redcedar-Western Hemlock-Grand Fir Forests

Ecology

Moist forests within the Northern Rockies range from 500 to 1,750 feet and occasionally occur at 1,900 feet (Jain and Graham 2005). These forests are influenced by a maritime climate with wet winters and dry summers. Precipitation ranges from 20 to 91 inches and predominantly occurs from November to May. A defining characteristic of these forests is a layer of fine-textured ash (up to 24 inches thick) that caps the residual soils. In addition, these forests are characterized by complex topography, including dissected slopes and varying degrees of slope angle, all of which influence soil development and ash cap depth. Disturbance is another important component of these forests that contributes to creating vegetative mosaics. Thus, the combination of disturbance, topography, moisture and temperature regimes, parent material, soil weathering, and ash cap depth results in productive vegetation that is complex in composition and structure. For example, up to 10 different tree species can occupy a given square yard of this forest type (Jain and Graham 2005).

The nine primary tree species that grow together in this wet forest type, outside of riparian areas, are ponderosa pine, western larch, Douglas-fir, grand fir, western white pine, western redcedar, western hemlock, lodgepole pine, and Engelmann spruce. The niche that these species occupies varies by habitats. For example, western redcedar is a mid-seral species in western hemlock types but is late seral on western redcedar riparian and upland habitat types. The genetic adaptive capacity and autecological characteristics of individual trees species and their tolerances to light, moisture, temperature, and disturbance is also highly variable among the different tree species in this forest type (Minore 1979). Some species are better adapted to regenerating in shade (western hemlock and western redcedar), whereas other species can regenerate in a wide range of conditions (western white pine). Some have a stronger competitive capacity than other species when growing together; for example, western larch cannot compete with western hemlock. Some are drought-tolerant (ponderosa pine, western larch, and western white pine), and others are drought-intolerant (western hemlock).

Disturbance Interactions

Natural disturbances (snow, ice, insects, disease, and fire), when combined, create heterogeneity in patch sizes, forest structures, and composition in this forest type. Ice and snow create small gaps and openings, reducing forest densities and altering species composition. Native insects (e.g., bark beetles) and diseases (e.g., Armillaria root rot and dwarf mistletoes) infect and kill the very old or stressed individuals, and tend to diversify vegetation communities (Hessburg et al. 1994). A mixed-severity fire regime also plays a role in creating a mosaic of forest compositions and structures. Historically, nonlethal surface fires occurred at relatively frequent intervals (every 15 to 25 years) in a quarter of the area of this forest type. Lethal crown fires burned about a quarter of the area at intervals of 20 to 150 years, occasionally extending to 300 years. A mixed-severity fire regime characterized the rest of the moist forests, with return intervals of 20 to 150 years. Fires typically started burning in July and were usually out by early September (Hann et al. 1997).

Historical and Current Conditions

In addition to white pine blister rust and salvage that removed most of the western white pine, harvesting removed the early-seral, shade-intolerant species (e.g., ponderosa pine and western larch) that were resistant to fire and other disturbances. Partial canopy removal and minimal soil surface disturbance in these harvests were ideal for Douglas-fir and grand fir, which regenerated aggressively, in contrast with the shade-intolerant pines and larch species. Fire exclusion also prevented the creation of canopy openings and receptive seedbeds for the regeneration of pine and larch. Similar to the dry forests, high canopies (>165 feet) of western white pine, western larch, and ponderosa pine and other early and mid-seral species are currently absent. In their place, the present forest structure and composition (grand fir and Douglas-fir) favor the compression of nutrients, microbial processes, and root activity toward the soil surface (Harvey et al. 2008). When wildfires occur, surface organic layers can be consumed, decreasing the nutrition and microbial processes important for sustaining these forests. In general, the lack of the early seral species and historical structures most likely have altered the disturbance regimes that sustained these forests.

Climate Change Responses

Habitat types are not static but reflect the operational environment that supports a particular set of plant species. As the moist forests experience climate change, the competition among species and how these forests evolve will be particularly dynamic. Thus, any discussion concerning climate change and an individual tree species that grows in moist mixed-conifer (and dry mixed-conifer) forest must be placed within the context of species cohorts, the adaptive capacity of an individual tree species, the interaction of disturbance, and how environmental niches change over time and space. For example, if future moisture regimes no longer support the current distribution of western hemlock, the remaining species that thrive on the upland western redcedar habitat types are likely to become dominant (Graham 1990). How disturbance changes (intensity, extent, and return interval) with a warming climate can also influence the subsequent effects on particular tree species. In a drier climate, western redcedar may become the late-seral species in what we consider to be western hemlock habitat types.

Lodgepole Pine Mixed Subalpine Forests

Ecology

Lodgepole pine forests straddling and occurring east of the Continental Divide are associated with the cold continental air mass that influenced their development. The higher elevations combined with the relatively dry cold climate associated with this type exclude many of the warm and moisture-dependent tree species found on the west side of the Continental Divide. Aspen, which is often associated with moisture seeps, swales, and other moist sites within this type, is released from conifer suppression by fire. Disturbance is needed to maintain aspen in this type and to keep seral lodgepole pine communities from becoming dense with subalpine fir.

Disturbance interactions. Lodgepole pine mixed subalpine forests in the Northern Rockies evolved with both high-severity and mixed-severity fire regimes. Mixed-severity fire regimes were common in central Montana on flatter slope positions and produced a diverse pattern of various-sized patches of different ages and tree sizes. Stand-replacing fire return intervals were 100 to 500 years (Fischer and Clayton 1983). However, stands reaching 60 to 80 years of age often suffered severe mortality from mountain pine beetle, creating snags and down fuel (Jenkins et al. 2008).

Historical and Current Conditions

This forest type was probably the most extensive in the Northern Rockies, with vast subalpine areas dominated by even-aged and multiaged stands of lodgepole pine, mixed with aspen, created by mixed-severity fire. Subalpine fir has probably increased as a result of fire exclusion, but more importantly, most of this type is currently dominated by large, mature lodgepole pine. Landscapes of these mature forests have fostered the large mountain pine beetle outbreak observed in many parts of the region (Central Rockies and GYA).

Climate Change Responses

This type will probably expand and contract, but provided that fire is not excluded from these areas, it is not likely to change substantially in a warmer climate.

Whitebark Pine Mixed Upper Subalpine Forests

Ecology

Perhaps the most threatened forest type, whitebark pine mixed upper subalpine forests are associated with high elevations, and the distribution of this type is primarily influenced by the cold continental air masses in Montana and higher elevations in northern Idaho. In this type, whitebark pine is found with subalpine fir, Engelmann spruce, and mountain hemlock, and subalpine larch in the area west of the Continental Divide. This type occurs on about 5 million acres in the Northern Rockies, primarily on the higher ridges and mountaintops. At the lower elevations within the range of this type, whitebark pine typically serves as a minor early-seral species in mixed-conifer stands. At the uppermost elevations, whitebark pine can serve as a major climax species.

Disturbance Interactions

Whitebark pine and its associates developed under both a stand-replacing fire regime on steep north slopes, and under a mixed-severity fire regime on other aspects and flatter slope positions. Various sized patches are common within the range, with density depending on moisture availability. The future could bring more-intense fire that could further threaten whitebark pine distribution.

There have been three outbreaks of mountain pine beetle in the Northern Rockies over the last 100 years. The first one in the 1920s–1930s killed significant areas of whitebark pine (Tomback et al. 2001). Snags from this outbreak can still be seen today. Another major outbreak occurred in the early 1980s, but the largest outbreak started in the mid-2000s and has continued, especially in the GYA.

Historical and Current Conditions

More than 14 percent of the Northern Rockies could have consisted of whitebark pine forests, with late-seral mixed fir-spruce patches mixed throughout, prior to 1910. However, with extensive white pine blister rust epidemics and mountain pine beetle outbreaks over the last several decades, the upper subalpine landscape has slowly shifted from whitebark pine to more spruce and fir and nonforest vegetation in some places (Tomback et al. 2001).

Climate Change Responses

There may be substantial change in the upper subalpine forests over the next century. However, that change will probably be driven by whitebark pine mortality from white pine blister rust rather than climate change, and the changes will primarily be in forest composition and structure rather than distribution. Over the last 40 years, whitebark pine has become a minor component of this forest type in many parts of the western Northern Rockies because of white pine blister rust, allowing subalpine fir to become dominant in both the overstory and understory. Although the GYA has yet to have massive die-offs from white pine blister rust, it has not escaped recent mountain pine beetle outbreaks, and the whitebark pine mortality rates in cone-bearing trees from these outbreaks exceed 50 percent in most areas. Recent fires in the upper subalpine have served to reset the successional clock to the earliest seral stages of shrub and herbaceous communities, but whitebark pine regeneration levels are low in these burns because of low population levels (Leirfallom et al. 2015). Clark's nutcracker apparently is eating most of the seeds from the few remaining whitebark pine trees and not enough of their seed caches go unclaimed to germinate and grow into trees (Keane and Parsons 2010). This has served to keep recently burned areas in the shrub/herb stage for long periods, which may allow time for other wind-dispersed tree species to populate the burn. Thus, whitebark pine may continue to decline in this type, and species dominance is likely to shift to subalpine fir, Engelmann spruce, and lodgepole pine.

Most of the range shifts of this forest type will probably be in wilderness areas, as about 50 percent of this type is found in wilderness (Keane 2000). Many Northern Rockies wilderness areas have lands that are above the elevations at which this type occurs, so there are potential areas for this type to expand. Wildland fire will be the catalyst for any range shifts in this forest type. Continued fire exclusion may seem appropriate for whitebark pine types, but it is contraindicated in many situations. Most whitebark pine will eventually succumb to white pine blister rust; thus, suppressing fire does not necessarily protect it. Fire is needed to create conditions in which whitebark pine can become established and grow to maturity. If fires are suppressed and no rust-resistant trees are planted, then whitebark pine is likely to remain a minor component of this forest type. However, if wildland fires occur and burned areas are planted with rust-resistant trees, then whitebark pine may become more abundant in the high elevation settings of the Northern Rockies. Therefore, land management is likely to be more critical than climate in dictating the future composition and extent of this forest type.

Resources of Concern

Landscape Heterogeneity

Background

Historically, most Northern Rockies landscapes were shaped by disturbance regimes interacting with vegetation and climate creating shifting mosaics of diverse vegetation assemblages. Wildfire was the primary sculptor of historical landscape composition and structure, especially at lower elevations (including ignitions by Native Americans and lightning), with other disturbances (mountain pine beetle outbreaks, root rot pockets, windthrow) woven into the patchwork of forestlands. Forest patterns were constantly shifting over time and space at rates governed by interactions among vegetation, disturbance, and climate, resulting in different patch sizes, shapes, and distributions. Therefore, understanding the variability and scale of disturbance and succession is critical to quantifying historical landscape heterogeneity, which in turn affects biological diversity and ecosystem resilience.

High landscape heterogeneity creates diverse biological structure and composition, which are considered more resilient and resistant to disturbances (Bannerman 1997; Cohn et al. 2015; Haire and McGarigal 2010; Turner 1987). For example, the effects of mountain pine beetle outbreaks are less severe in landscapes with diverse age structures of host tree species (Schoettle and Sniezko 2007). Heterogeneous landscapes also promote population stability (Oliver et al. 2010) because fluctuations in plant and animal population are less when landscape structure is diverse (Turner et al. 1993). Heterogeneous landscapes may also have more corridors, buffers, and refugia for wildlife and plant migration.

During the past 100 years, land management practices have altered the temporal and spatial characteristics of Northern Rockies landscapes. Timber management has modified patch shape and structure at lower elevation, and fire exclusion has changed patch size and diversity. Fire exclusion has in many cases created landscapes with large contiguous patches of old, dense stands with high surface and canopy fuel accumulations (Keane et al. 2002), although some areas with frequent disturbance (e.g., frequently burned ponderosa pine forest) are also homogeneous compared to presettlement montane forests (Romme 2005). Because we have directly or indirectly managed for late seral conditions, some Northern Rockies landscapes are highly susceptible to insects and disease, owing to low tree vigor from intense competition, and have an abundance of live and dead fuels that will contribute to the severity of future wildfires.

Climate Change Responses

Many current Northern Rockies landscapes have less ability to buffer potential climate change effects because of widespread increases in the density of shade-tolerant species in forests, although some landscapes, especially in subalpine forests, still have structures and compositions similar to those observed in the historical record. Recent wildfires, restoration activities, and timber harvest have helped return some heterogeneity, especially in wilderness areas and national parks. However, most Northern Rockies landscapes are outside their historical range and variability (HRV) in landscape structure, making it challenging to implement effective climate change adaptation.

Landscape heterogeneity may increase if climate-mediated changes in disturbance regimes increase (Funk and Saunders 2014). During the past 20 years, wildfire area burned and mountain pine beetle outbreaks have both increased, replacing late seral forests with younger age and size classes and thereby increasing heterogeneity. Continued increases in wildfires and other disturbances are projected in a warmer climate (Bentz et al. 2010; Marlon et al. 2009), so projected declines in biodiversity (e.g., Botkin et al. 2007) could be balanced by gains in landscape heterogeneity (Kappelle et al. 1999).

Continued fire exclusion in a warmer climate may promote late seral forests that would be stressed from competition and drought (van Mantgem and Stephenson 2007). Wildfires that will eventually burn these landscapes may become large and burn more severely, thereby creating large patches of homogeneous postburn conditions (Flannigan et al. 2005, 2009). These fires may also create semipermanent shrublands and grasslands in areas that have become too dry for conifer establishment or where seed sources are eliminated (Fulé et al. 2004). However, some have found a high degree of heterogeneity in severity and vegetation conditions following large fires (Collins and Stephens 2010; Keane et al. 2008). Although the size, shape, and distribution of forest management treatments are a concern for landscape heterogeneity, the effects of management on landscape properties may be overwhelmed by other disturbances.

Is there an appropriate level of heterogeneity for Northern Rockies landscapes? How can management facilitate landscape heterogeneity and minimize adverse climate change effects? Mechanistic ecosystem models can be used to simulate landscape structure and composition in the future and to understand effects of management actions (Keane 2013), but cannot generate heterogeneity metrics as design criteria for ecosystems. Using the HRV of landscape characteristics is a more straightforward and useful approach (Keane 2013; Morgan et al. 1994a; Nonaka and Spies 2005) (box 6.1). The HRV of landscape metrics may not represent future conditions (Millar 1997; Millar and Woolfenden 1999), but does provide an estimate of landscape conditions under which ecosystems have developed over the last 1,000 years, conditions that produced functional, heterogeneous ecosystems (Landres et al. 1999). It is preferable to first use HRV as a reference for landscape heterogeneity (Keane et al. 2015b), then ecological models can be parameterized for historical conditions and used to generate a set of useful landscape metrics (Keane 2012).

Timber Production

Background

The area managed for timber production as one of the objectives for management is about 8,700 out of 34,000 square miles of forested lands throughout the USFS Northern Region. This area includes approximately 2,600 square miles in northern Idaho, 4,250 square miles in western Montana, 1,400 square miles in central and eastern Montana, and 450 square miles in the Greater Yellowstone portion of the Northern Region.

During the 1970s and 1980s, an average of 98 square miles were harvested each year, which amounted to about 1,900 square miles that had some type of harvest treatment implemented to meet various management objectives including timber production. From 1990 through 2014, an average of 39 square miles were harvested each year, which amounted to about 970 square miles. Recent harvest during 2014 of 32 square miles may be more typical of current and near-term future harvest levels.

The species composition of timber harvests has fluctuated during the past 45 years, as harvest has often followed some disturbance agent such as mountain pine beetle in western white pine and lodgepole pine, spruce beetle in Engelmann spruce, white pine blister rust in western white pine, root disease in Douglas-fir and grand fir, Douglas-fir beetle and spruce budworm in Douglas-fir, and wildfire in a variety of species types. The current percentage of acreage in each of the major species composition groups within the lands suitable for timber production across the Northern Region is 6 percent ponderosa pine, 13 percent dry Douglas-fir, 27 percent lodgepole pine, 6 percent western larch, 12 percent mixed subalpine fir and Engelmann spruce, and 35 percent mixed western white pine, grand fir, western hemlock, moist site Douglas-fir, and western redcedar forests.

Many of the current timber harvests in mixed mesic types of northern Idaho and western Montana are removing grand fir, Douglas-fir, and western hemlock, and replanting western white pine, western larch, and ponderosa pine. Other harvests involve removal of lodgepole pine and replanting of western larch. Thinning in ponderosa pine and dry Douglas-fir forests is also common. Within eastern Montana and the GYA, harvesting is concentrated on mountain pine

Box 6.1—Using Historical Range and Variability to Assess and Adapt to Climate Change

To effectively implement ecosystem-based management, land managers often find it necessary to obtain a reference or benchmark to represent the conditions that describe fully functional ecosystems (Cissel et al. 1994; Laughlin et al. 2004). Contemporary conditions can be evaluated against this reference to determine status, trend, and magnitude of change, and to design treatments that provide society with valuable ecosystem services while returning declining ecosystems to a more sustainable condition (Hessburg et al. 1999; Swetnam et al. 1999). Reference conditions are assumed to represent the dynamic character of ecosystems and landscapes, varying across time and space (Swanson et al. 1994; Watt 1947).

The concept of *historical range and variability* (HRV) was introduced in the 1990s to describe past spatial and temporal variability of ecosystems (Landres et al. 1999), providing a spatial and temporal foundation for planning and management. HRV has sometimes been equated with “target” conditions (Harrod et al. 1999), although targets can be subjective and somewhat arbitrary; they may represent only one possible situation from a range of potential conditions (Keane et al. 2009). HRV encompasses a full range of conditions that have occurred across multiple spatiotemporal scales.

HRV represents a broad historical envelope of possible ecosystem conditions—burned area, vegetation cover type area, patch size distribution—that can provide a time series of reference conditions. This assumes that (1) ecosystems are dynamic, not static, and their responses to changing processes are represented by past variability; (2) ecosystems are complex and have a range of conditions within which they are self-sustaining, and beyond this range they make a transition to disequilibrium (Egan and Howell 2001); (3) historical conditions can serve as a proxy for ecosystem health; (4) the time and space domains that define HRV are sufficient to quantify observed variation; and (5) the ecological characteristics being assessed for the ecosystem or landscapes match the management objective (Keane et al. 2009).

The use of HRV has been challenged because a warmer climate may permanently alter the environment of ecosystems beyond what was observed under historical conditions (Millar et al. 2007a). In particular, disturbance processes, plant species distribution, and hydrologic dynamics may be permanently changed (Notaro et al. 2007). However, a critical evaluation of possible alternatives suggests that HRV might still be the most viable approach in the near term because it has relatively low uncertainty.

An alternative to HRV is forecasting future variations of landscapes under changing climates by using complex empirical and mechanistic models. However, the range of projections for future climate from the commonly used global climate models may be greater than the variability of climate over the past three centuries (Stainforth et al. 2005). This uncertainty increases when we factor in projected responses to climate change through technological advances, behavioral adaptations, and population growth (Schneider et al. 2007). Moreover, the variability of climate extremes, not the gradual change of average climate, will drive most ecosystem response to climate-mediated disturbance and plant dynamics (Smith 2011) that are difficult to project. Uncertainty will also increase as climate projections are extrapolated to the finer scales and longer time periods needed to quantify *future range and variability* (FRV) for landscapes (Araujo et al. 2005; Keane et al. 2009).

Given these cumulative uncertainties, time series of HRV may have lower uncertainty than simulated projections of future conditions, especially because large variations in past climates are already captured in the time series. It may be prudent to wait until simulation technology has improved enough to create credible FRV landscape pattern and composition, a process that may require decades. In the meantime, attaining HRV would be a significant improvement in the functionality of most ecosystems in the Northern Rockies, and would be unlikely to result in negative outcomes from a management perspective. As with any approach to reference conditions, HRV is useful as a guide, not a target, for restoration and other management activities.

beetle-susceptible or dead lodgepole pine and ponderosa pine, and thinning in ponderosa pine and dry forest Douglas-fir forests is also common.

Climate Change Responses

With increasing temperatures and the potential for increases in forest productivity (Aston 2010; Joyce 1995) and biomass accumulation (Lin et al. 2010) will probably come potential increases in timber production for most Northern Rockies forests (Garcia-Gonzalo et al. 2007). Productivity increases are projected to be substantial because most forested lands in the region are in the mesic montane, subalpine, and upper subalpine. The increase in biomass might result in higher basal areas, greater timber value, and increased regeneration (Sohngen et al. 2001). However, these mesic temperate forests might also become denser, which may result in decreased vigor that may offset gains in productivity from climate alone. Depressed vigor might also increase susceptibility to insects and disease; because insect and disease outbreaks are projected to increase in severity and frequency, there may be some major timber losses from forest pathogen and insect mortality (Joyce et al. 2008).

There will also be an increase in potential mortality from wildland fire with increased fuel, drier conditions, and longer fire seasons, and this might facilitate even more timber losses. Future timber harvests from mature timberland might be a race against losses from increased insects, disease, and fire. The greatest climate change impacts on commercial forestry may come from changes in the disturbance regimes rather than changes in productivity (Kirilenko and Sedjo 2007).

There are other considerations in addressing how timber resources in the Northern Rockies region will change with warming climates. First, most of the roads on Northern Rockies lands are in drier, lower elevation forests where productivity may decline and more trees are projected to die from drought. Fewer roads are in the subalpine and upper subalpine where productivities and associated timber values are likely to increase, resulting in limited ability to transport timber to markets. Creation of new roads is expensive, risky, and environmentally damaging. These higher lands are distant from timber markets and sawmills, and are also more topographically complex and steep, thereby limiting the potential for mechanized timber removal while increasing harvesting costs. These higher elevation lands are also where most of the threatened and endangered plant and animal species are found, especially grizzly bear (*Ursus arctos*), Canada lynx (*Lynx canadensis*), and wolverine (*Gulo gulo*), making it more difficult to implement timber harvest projects in these sensitive areas. If increases in insects, diseases, and fire are realized, the quality of timber will probably be reduced, and the value of the timber for building material will drop dramatically (Gillette et al. 2014; Kirilenko and Sedjo 2007; Spittlehouse and Stewart 2004). Longer fire seasons will probably mean there will be less time to perform forestry tasks, such as inventory, sale layout, and cruising. This may also mean that less agency money will be spent on forestry projects, such as ecosystem restoration, fuels treatments, and timber harvest sales, and more money will be spent on fire suppression activities. As the risk of uncharacteristic fire severity due to uncharacteristically high forest density increases, there will most likely be reductions in timber production opportunities, especially in dry forest areas that may be lost and converted to grass and shrub lands (Allen et al. 2010).

Timber species will also shift in the future. Increases in temperature and soil moisture deficits may result in shifts of desirable timber species, such as western larch, to species compositions that are susceptible to root disease, such as Douglas-fir and grand fir. Any increases in production at mid- and higher elevations from warming temperatures could be offset by losses from root disease because of continued fire exclusion. Land management efforts that create late-seral, shade-tolerant communities, namely fire exclusion and some fuels treatments, will increase the risk that standing timber will be affected by damaging agents before it can be harvested.

Many new forest practices, harvesting techniques, and markets are being proposed to offset carbon emissions from fossil fuels with carbon emissions from harvested biomass (Kirilenko and Sedjo 2007). Most of these new technologies will result in better utilization of timber resources and a more diverse and vibrant timber market. Biomass burning for energy, for example, could provide a market for noncommercial material removed from proposed fuels treatments. Slash piles could have value as biomass for energy. A more diverse market for wood products would surely enhance potential timber harvests in the region, but it is essential that any proposed cutting activity be done in an ecological context, especially in this time of rapidly changing climates. Proposed harvesting activities must address a wide diversity of issues, such as landscape character, species mix, successional dynamics, and fuels, to ensure these activities are effective and to minimize the long-term environmental impact.

It is essential that ecological principles be used to design harvest treatments of the future to ensure the creation of resistant, resilient forests that can withstand major impacts of climate change. Designing fuels treatments without considering ecosystem restoration concerns, for example, might create forests that are highly susceptible to insects and disease or fire. Favoring shade-tolerant, fire-susceptible species over fire-tolerant, sun-loving, early-seral tree species is ecologically inconsistent and likely to create landscapes that are intolerant of future climate change.

One proposed management alternative, carbon sequestration (see next subsection), might be cause for concern. The main assumption of most carbon sequestration options is to maximize biomass to sequester carbon from the atmosphere and put it into timber products to offset fossil fuel burning. The problem is that this approach must recognize the role of disturbance to be effective in the long term. Many studies have shown that the most resilient forests are ones with suboptimal carbon sequestration.

Again, the major issue related to climate change and timber production in the near term is loss from disturbance. The anticipated increases in drought, severe large wildfires, root disease, other diseases such as white pine blister rust, and insect damage such as that from large-scale bark beetle outbreaks, need to be addressed throughout the Northern Rockies. Tactics to increase landscape heterogeneity overall and reduce forest density in the dry forest types will be key climate change responses in the near term. Adjusting species composition and distribution may help sustain long-term timber production.

Carbon Sequestration

Background

North American forests are considered important carbon sinks and currently offset about 13 percent of annual continental fossil fuel emissions (Pacala et al. 2007). Size and persistence of forest carbon sinks depend on land use, land management, and environmental factors such as vegetation composition, structure, and distribution, climate, and disturbance processes including wildfire.

Vegetated landscapes play an important role in storing carbon in the form of plant and animal materials (both live and dead), aboveground and in soils. Forests store carbon in soils (about 45 percent of total storage), aboveground and belowground live biomass (about 42 percent), dead wood (about 8 percent), and litter (about 5 percent) (Bonan 2008; Pan et al. 2011). Because forests contain large reservoirs of carbon (i.e., carbon sinks) and facilitate flows of carbon from the atmosphere to the biosphere (i.e., carbon sequestration), they are an important component of the global carbon cycle and are thought to have the potential to mitigate climate change (Ingerson 2007; Pan et al. 2011). The carbon sequestration potential of Earth's forests is about 33 percent of global anthropogenic emissions from fossil fuels and land use (Denman 2007). Carbon typically accumulates in woody biomass and soils for decades to centuries until a disturbance event releases this stored carbon into the atmosphere (Goward et al. 2008). Disturbance and decomposition are recognized as primary mechanisms that shift ecosystems from carbon sinks to carbon sources (Baldocchi 2008), and wildfire in forested ecosystems is one of the primary disturbances that regulates patterns of carbon storage and release (Kasischke et al. 2000a,b). Forest insect outbreaks can also release carbon through decomposition of needles and other fine fuels from attacked trees (Kurz et al. 2008). The amount and rate of carbon release from a disturbance event depends on the extent and severity of the disturbance, as well as predisturbance site conditions and productivity (Bigler et al. 2005; Falk et al. 2007). In the case of both wildfires and insect outbreaks, although long intervals between events can allow carbon to accumulate for years to centuries, probability of disturbance increases with time (Clark 1989). Changing climate, in combination with other ecosystem stressors such as disturbance, may be sufficient to cause structural or functional changes in ecosystems, and thus fundamentally alter carbon dynamics of landscapes.

Although long intervals between disturbance events, such as wildfires or insect outbreaks, can allow carbon to accumulate for years to centuries, probability of disturbance increases with time (Goward et al. 2008; Loehman et al. 2014). Thus, disturbance-prone forests will eventually move stored carbon to the atmosphere, regardless of management strategies designed to limit or prevent disturbance events. However, unless structural or functional ecosystem shifts occur, net carbon balance in disturbance-adapted systems at steady state is zero when assessed over long time periods and at landscape scales. This is significant to management for two reasons: (1) disturbance-prone ecosystems cannot be managed to increase stored carbon over historical amounts without limiting the occurrence and magnitude of disturbance events; and (2) major shifts in vegetation composition, distribution, and structure resulting from climate change will result in different patterns of carbon storage on the landscape as compared with the historical period. Thus, it is important to develop expectations for landscape carbon storage potential in the context of projected climate change effects on both disturbance dynamics and vegetation patterns, and the relationships between them.

Climate Change Responses

As described in chapter 8, future warmer, drier conditions are likely to result in more frequent, larger wildfires, and greater annual area burned, which will serve to move carbon from biomass storage to the atmosphere. Warmer temperatures and increased drought stress are also projected to increase the area susceptible to or affected by beetle outbreaks. Regrowth of forests following disturbance may be delayed if the climatic conditions stress remaining or reestablishing species. Disturbance events in combination with additional climate-caused stressors may also result in functional transitions, such as a shift from forests to montane woodland or grassland-dominated vegetation types, which would probably result in less stored carbon. Frequent fires may also maintain open woodlands and savannas that might sequester less carbon than forests, but these carbon pools may be more stable and resilient to climate shifts.

Strategies that aim to manage carbon resources should consider the following:

1. Is the system disturbance-prone? If so, is it reasonable to expect the system to accumulate carbon over historical (steady-state) levels, especially given future climatic conditions that may increase the frequency, severity, and magnitude of disturbance events?
2. What are appropriate temporal and spatial scales over which to measure carbon storage? For example, in forests with multicentury disturbance cycles, it may take hundreds of years for forests to attain pre-disturbance levels of carbon, but this does not mean that they have become carbon sources at the appropriate, ecological scale of measurement.
3. Can potential future disturbance events be managed? For example, will it be possible to suppress or exclude wildfires from the system in the future, and at what economic or ecological costs?
4. Can additional stressors (e.g., drought stress, invasive weeds, and other management activities) be mitigated, to help maintain existing vegetation communities?
5. How might the system change with changing climate and disturbance? For example, are future climatic conditions conducive to persistence of forests, or will conditions become too warm or dry for the current dominant species?
6. Do planned carbon accounting methods assess ecological benefits of natural disturbance processes in carbon-equivalent units so that they can be weighed against carbon losses from disturbance? For example, wildfires confer many important ecological benefits not measurable in carbon units (e.g., nutrient release and redistribution and stimulation of plant growth, increased productivity in soil systems from decomposition of burned material, initiation of

vegetation succession and forest regeneration, and increased availability of resources for surviving trees) that may actually increase carbon sequestration rates.

Two complementary activities, monitoring and modeling, can improve our understanding of cross-scale ecological drivers and responses to disturbance (Loehman et al. 2014). Monitoring programs can be used to quantify long-term carbon dynamics before and after disturbance, evaluate responses of ecosystems to changes in climate, and identify shifts in ecosystem patterns and processes emergent under changing climates. Monitoring data can also be used to provide inputs to, calibrate, and validate models. Models, in turn, can be used to simulate emergent environmental patterns, compare effects of potential treatments, identify vulnerable landscapes or ecosystem components, and bridge gaps between landscape-scale ecological processes and variables measured in small areas and over short periods of time. There is room for improvement on both fronts, as described previously in this chapter. Although it may be tempting to meet policy-driven goals of increased carbon storage via management strategies designed to exclude or limit the extent and magnitude of disturbance events (e.g., wildfires), it is important to remember that native disturbance processes confer many important ecological benefits not measurable in carbon units (see item 6). Thus, it will be important to develop accounting methods that can assess ecological benefits in carbon-equivalent units so that they can be weighed against carbon losses from disturbance.

Vulnerability Assessment

All items in each of the three levels of assessment were rated as to their vulnerability to climate change using the same rubric employed across all chapters in this report (table 6.4), and the results are summarized in detail in Appendix 6B. We populated a table with information for the eight categories in table 6.4 using a thorough consideration of five factors: climate, disturbance, life history, dependencies, and other stressors. For climate, we considered whether the species, vegetation type, or resource concern was sensitive to changes in direct effects of climate (temperature and precipitation) or indirect effects of climate (e.g., soil moisture, snowpack, seasonality of flows, climatic water deficit, altered flow regimes, and stream temperature). For example, we asked, “Does the system inhabit a relatively

Table 6.4—Categories used to assess the vulnerability of species, vegetation types, and resource concerns in this chapter.

Evaluation category	Description	Example
Habitat, ecosystem function, or species	Specific biophysical or social entity of interest	Whitebark pine
Broad-scale climate change effect	Overarching change in climate that is expected to affect a resource	Warming temperatures
Current condition, existing stressors	Current status of resource relative to desired conditions, including factors that are reducing the quality or quantity of the resource	Reduced abundance, wildland fire, mountain pine beetle, white-pine blister rust
Sensitivity to climatic variability and change	Specific sensitivity of a habitat, species, or ecosystem function that responds to climate	Low ability to compete with encroaching conifers
Expected effects of climate change	How specific habitat, species, or ecosystem function is expected to respond to climate change (develop inferences from model projections and known responses to climatic variability)	Regeneration may be reduced by combination of warming and low seed availability
Adaptive capacity	Ability to adjust to climate change, to moderate potential damages, or to cope with the consequences; usually more appropriate for species than for systems and processes	Variable: unable to compete with other tree species, but bird-mediated seed dispersal allows quick colonization of burned over areas
Exposure	The extent to which each species’ physical environment will change expressed as low, moderate, or high	High
Risk assessment, magnitude of effects	Estimate of the magnitude of climate change effects expressed as low, moderate, or high by time period	Moderate
Risk assessment, likelihood of effects	Estimate of the likelihood that climate change effects will occur expressed as low, moderate, or high by time period	High

narrow climatic zone, or does it experience large changes in composition or structure with small climatic changes?" We also considered both direct sensitivity to climate change (e.g., ecophysiology and life history) and indirect sensitivity to climate change (e.g., ecological relationships such as competition, dispersal, and migration). Vulnerability to disturbance was assessed in reference to whether the species, type, or concern was sensitive to major disturbances, primarily wildland fire, insect outbreaks, drought, and pathogens. Disturbances are major catalysts for vegetation change and can combine with climate stressors and nonclimate stressors to create a broader stress complex with multiple interactions. Life history aspects of the species and vegetation type were considered to address the impact of the growth rate, susceptibility to mortality, longevity, and reproductive strategy of a species, all of which may influence sensitivity to climate change. Species with long lifespans may have lower vulnerability than short-lived species. We also addressed the dependence of species on other ecosystem processes or landscape elements. Riparian species, for example, are dependent on wet conditions. Last, nonclimate stressors, such as land use, grazing, timber harvest, and fire exclusion were integrated into our assessment.

The vulnerability assessment in Appendix 6B is further summarized in table 6.5 for only the tree species included in this report. We ranked each species by subregion (fig. 6.1) and removed those species that did not occur in a given subregion. We also included the rankings of two other efforts for comparison purposes. The Devine et al. (2012) report assessed vulnerability for tree species in the Pacific Northwest, and the Hansen and Phillips (2015) effort assessed vulnerability for some Northern Rockies tree species using SDMs. This information is presented as a means of helping land managers to integrate climate change impacts into their planning documents and analyses.

Adaptation Strategies and Tactics

This chapter documents what could happen to Northern Rockies forest resources under potential future climates. Land managers need options for adapting to climate change and mitigating any adverse impacts incurred as a result of changing climate. Adaptation can be defined as initiatives

Table 6.5—Final tree species vulnerability ratings (1 = lowest vulnerability) for the entire Northern Rockies (NR), and the five subregions of the NR. Also included are ratings from the Pacific Northwest in Devine et al. (2012) report and from the northern Rocky Mountains in Hansen and Phillips (2015).

Tree species	NR	West	Central	East	GYA	Grass	Devine et al. (2012)	Hansen and Phillips (2015)
Alpine larch	1	2	1	NA[a]	NA	NA	4	NRT[b]
Whitebark pine	2	1	2	1	1	NA	1	1
Western white pine	3	5	3	NA	NA	NA	13	NRT
Western larch	4	6	4	NA	NA	NA	12	8
Douglas-fir	5	8	8	2	2	1	11	9
Western redcedar	6	4	5	NA	NA	NA	15	7
Western hemlock	7	3	6	NA	NA	NA	10	6
Grand fir	8	7	7	NA	NA	NA	5	11
Engelmann spruce	9	9	11	3	4	5	3	5
Subalpine fir	10	10	12	4	5	6	2	4
Lodgepole pine	11	11	10	5	6	7	8	3
Mountain hemlock	12	3	9	NA	NA	NA	7	2
Cottonwood	13	12	13	6	3	2	17	NRT
Quaking Aspen	14	13	14	8	7	3	6	NRT
Limber pine	15	NA	15	7	8	4	18	NRT
Ponderosa pine-west	16	14	16	NA	NA	NA	14	10
Ponderosa pine-east	17	NA	NA	8	9	8	NRT	10
Green ash	18	NA	NA	9	10	9	19	NRT

[a] NA = Not applicable.
[b] NRT = Not rated.

and measures to reduce the vulnerability of natural and human systems against actual or expected climate change effects (IPCC 2007). Adaptation actions range from the simple, such as doing nothing or increasing the harvest rotation age, to the complex, such as implementing fuels treatments to reduce the risk of high-severity fire in ecosystems with rare plants (Spittlehouse and Stewart 2003). Most land managers have the tools, knowledge, and resources to begin to address climate change, but as Swanston and Janowiak (2012) note, managers need to expand their thinking to consider new issues, spatial scales, timing, and prioritization of efforts. For example, managers need to account for the high variability and trend of climate in the design of alternative land management actions.

There are some fundamental principles that can serve as starting points in the development of adaptation approaches (Joyce et al. 2008; Millar et al. 2007a; West et al. 2009). First, it will be increasingly important to prioritize management actions based on both the vulnerability of resources and on the likelihood that actions to reduce vulnerability will be effective (i.e., prioritization). Next, adaptive management principles provide a decisionmaking framework that maintains flexibility and incorporates new knowledge and experience over time (i.e., adaptive management). Management actions that result in a wide variety of benefits under multiple scenarios but have little or no risk may be the first places to look for near-term implementation (i.e., low-hanging fruit). Where vulnerability to a particular resource is high, precautionary actions to reduce risk in the near term, even with existing uncertainty, may be essential (i.e, triage). It is important to remember that climate change is much more than increasing temperatures; increasing climate variability across all components of climate, such as precipitation, humidity, and radiation, will lead to equal or greater impacts that will need to be addressed (i.e., increased uncertainty). Last, many adaptation actions are often complementary with other land management actions, and any actions to adapt forests to future conditions may also help restore these forests to healthy conditions (i.e., multiple objectives). When designing adaptation actions, it is important to address and integrate these principles to maximize efficiency.

The concepts of resistance, resilience, and response serve as the fundamental options for managers to consider when responding to climate change using adaptation (Millar et al. 2007a; Swanston and Janowiak 2012). *Resistance options* improve the defenses of an ecosystem against anticipated climate change responses or directly defend the ecosystem against disturbance to maintain current conditions. Resistance actions are often effective in the short term, but resistance options are likely to require greater effort over the long term as the climate shifts further from historical norms. Moreover, there is a real risk that the ecosystem will undergo irreversible change because of large climatic shifts, thereby rendering all resistance activities ineffective. *Resilience options* allow some change, but emphasize a quick return to prior conditions after a disturbance. Resilience actions are also short-term and should be used for high-value resources or areas that are buffered from climate change impacts. *Response options* intentionally accommodate change and allow ecosystems to adaptively respond to changing and new conditions. A wide range of actions exists under this option, all working to influence ways in which ecosystems adapt to future conditions.

Resistance, resilience, and response options serve as the broadest and most widely applicable level of a continuum of management responses to climate change. Along this continuum, adaptation actions become increasingly specific from options to strategies to tactics. Adaptation *strategies* describe how adaptation options could be employed, but they are still broad and general in their application across ecosystems.*Tactics* are more-specific adaptation responses, and they can provide prescriptive directions on how actions can be applied on the ground.

There are many broad strategies and associated tactics that can be used to adapt to climate change impacts in the Northern Rockies region, and the major ones that were identified by managers and scientists in a series of workshops in fall 2015 are detailed in table 6.6 and described next. Adaptation tactics for all Northern Rockies species, vegetation types, and resource concerns discussed in this chapter are summarized in table 6.7. Adaptation tactics were designed at different scales and levels of organization. Some involve Northern Rockies managers at the highest levels of agency organization, and others apply to ecologists, silviculturists, and resource specialists at the lower levels of organization. Some tactics concern multiple species or resources, while others are specific to just one entity. These tactics were designed so that Northern Rockies managers can use these recommendations to directly address climate change impacts in their planning and implementation of any action, specifically National Environmental Policy Act analysis.

As in other adaptation efforts, many tactics developed by Northern Rockies managers were focused on protecting forests from severe disturbance, mainly fire (table 6.6). For example, managers identified promoting disturbance-resilient forest structure and species as key strategies. Both thinning and prescribed fire can be used to reduce forest density and promote disturbance-resilient species. Disturbance-resilient species can also be planted. Managers recognized the importance of promoting and planting site-adapted species, specifically western larch and western white pine on moist sites, ponderosa pine on dry sites, Douglas-fir on extremely dry sites, and lodgepole pine on harsh sites that are difficult to regenerate.

Preparing for disturbance will also be important under a changing climate. Tree regeneration after severe fire may be more limited in the future with increased drought. Promoting legacy trees of disturbance-resilient species may help to increase postfire regeneration. Managers may also want to increase seed collection and ensure that adequate nursery stock is available for post-disturbance planting.

Another theme in the adaptation strategies and tactics developed by Northern Rockies managers was promoting diversity, including species diversity, genetic diversity, and

Table 6.6—Adaptation options that address climate change effects on forested vegetation in the Northern Rockies.

Sensitivity to climatic variability and change: The frequency and scale of disturbance will likely increase with climate change.			
Adaptation strategy/approach: Promote disturbance-resilient species, including western larch and western white pine on moist sites, ponderosa pine on dry sites, Douglas-fir on extremely dry sites, and lodgepole pine on harsh sites that are difficult to regenerate.			
	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Thin to favor disturbance-resilient species.	Plant disturbance-resilient species.	Promote disturbance-resilient species with prescribed fire and/or natural fire use.
Where can tactics be applied?	On sites most likely to be affected by wildfire	On sites most likely to be affected by wildfire	On sites most likely to be affected by wildfire
Sensitivity to climatic variability and change: Increasing temperatures with climate change will lead to increased moisture stress and fire.			
Adaptation strategy/approach: Reduce forest density and maintain low densities.			
	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Conduct thinning treatments (pre-commercial and commercial).	Use prescribed fire to maintain structure and promote fire-tolerant conifer species.	Use regeneration and planting to influence forest structure.
Where can tactics be applied?	Dense stands with low risk of root disease problems	Dense stands	Where there are no opportunities for thinning because of species or stand condition
Sensitivity to climatic variability and change: Areas with limited species and genetic diversity will likely be more susceptible to climate change stressors.			
Adaptation strategy/approach: Promote species and genetic diversity.			
	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Plant potential microsites with mix of species (hedge your bets).	Maintain species diversity during thinning.	Interplant to supplement natural regeneration and genetic diversity.
Where can tactics be applied?	In sufficiently large openings on sites most at risk for drought stress	In dense or overstocked stands on sites more at risk for drought stress and where sufficient diversity exists for thinning	In sufficiently large openings on sites most at risk for drought stress

Table 6.6 (cont.)—Adaptation options that address climate change effects on forested vegetation in the Northern Rockies.

Sensitivity to climatic variability and change: Increasing temperatures with climate change will lead to longer growing seasons, increased drought stress, and increased fire.			
Adaptation strategy/approach: Promote ecosystem resilience.			
	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Promote legacy trees of western larch, Douglas-fir, western white pine, Engelmann spruce, whitebark pine, and subalpine larch for regeneration and wildlife habitat.	Collect seed for post-wildfire reforestation and other planting needs, especially for subalpine larch, whitebark pine, and high-elevation western larch.	Promote landscape heterogeneity and create connectivity at multiple scales.
Where can tactics be applied?	On dense spruce, fir, and lodgepole pine sites	Where species of interest occur	At broad spatial scales
Sensitivity to climatic variability and change: Climate change will likely lead to increased disturbance, increased moisture deficits, and species distribution shifts.			
Adaptation strategy/approach: Increase knowledge for agency land managers and stakeholders.			
	Specific tactic – A	**Specific tactic – B**	**Specific Ttactic – C**
Tactic	Determine what connectivity means for different species and guilds.	Track tree species regeneration and distribution.	Improve integration between wildlife managers and forest ecologists, and between research and management.
Where can tactics be applied?	All lands	All lands	All lands
	Specific tactic – D	**Specific tactic – E**	**Specific tactic – F**
Tactic	Monitor blister rust resistance within planted white pine stands, and try to understand the relationship between infection rates and climatic drivers (e.g., fog).	Identify other resource management goals (not directly related to stand structure and composition) that may modify management strategies for forest vegetation, such as water yield, snow retention, and wildlife habitat.	Support the adaptive management research framework; develop consistent monitoring framework that can capture long-term change.
Where can tactics be applied?	On sites planted with white pine	Federal lands	Federal lands

Table 6.6 (cont.)—Adaptation options that address climate change effects on forested vegetation in the Northern Rockies.

Sensitivity to climatic variability and change: Climate change will likely lead to increased whitebark pine mortality through increased mountain pine beetle activity, fire, and white-pine blister rust. There will also likely be a loss of site conditions that support whitebark pine.			
Adaptation strategy/approach: Promote resilient whitebark pine communities.			
	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Identify sites that are less likely to be affected by climate change (refugia), and focus on those sites for restoration.	Implement a variety of management strategies and options to promote whitebark pine, including fire management, planting at lower elevations, and removing other dominant species (e.g., lodgepole pine, spruce and fir).	Plant genetically-selected seedlings to promote blister rust resistance.
Where can tactics be applied?	Federal lands	Federal lands	Federal lands
Sensitivity to climatic variability and change: Climate change will lead to increased risk of stand-replacing wildfire, mortality from drought, and loss of large ponderosa pine individuals in ponderosa pine forests.			
Adaptation strategy/approach: Decrease density within stands, and increase structural diversity across the landscape.			
	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Reduce density by thinning, prescribed fire, and wildfire use, with density and structural goals based on past and predicted future conditions.	Promote age class and structural diversity across the landscape, through regeneration harvest, thinning, prescribed fire and wildfire use.	Monitor establishment, survival and development of ponderosa by age class and in different topoedaphic conditions using Forest Inventory and Analysis data and project-level stocking exams.
Where can tactics be applied?	Ponderosa pine, Douglas-fir forests	In locations where activities are allowed	Ponderosa pine, Douglas-fir forests
Sensitivity to climatic variability and change: Climate change stressors cross boundaries, forcing agencies to coordinate and work across boundaries.			
Adaptation strategy/approach: Work across jurisdictions at a larger scales.			
	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Align budgets and priorities for program of work with neighboring lands.	Communicate about projects adjacent to other lands, and coordinate on the ground.	Work across boundaries to preserve roads, trails, and access with increasing fire and flood events.
Where can tactics be applied?	All lands	All lands	All lands

Table 6.6 (cont.)—Adaptation options that address climate change effects on forested vegetation in the Northern Rockies.

Sensitivity to climatic variability and change: The vigor and extent of quaking aspen may be reduced with increased frequency of disturbance and drought under changing climate.			
Adaptation strategy/approach: Restore and promote the health and vigor of aspen clones.			
	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Manage ungulates during the regeneration phase (e.g., with slash barriers, increased hunting pressure, grazing management, and salt block placement).	Remove conifers around aspen, at multiple scales and using multiple tools (i.e., hand and commercial treatments).	Monitor the extent and condition of aspen clones to determine the effect of disturbance frequency on aspen survival, effects of sudden aspen decline, and incidence of new clones; this may lead to strategic protection measures for specific clones.
Where can tactics be applied?	Wherever aspen regeneration prescriptions are applied	Wherever aspen occurs outside of protected areas	Wherever aspen occurs
Sensitivity to climatic variability and change: The distribution of subalpine forests is likely to shift as a result of increasing temperatures with climate change.			
Adaptation strategy/approach: Monitor and detect change in seedling survival, species composition, and mortality of mature trees in subalpine forests.			
	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Install and analyze additional plots to gather trend information over time, targeting areas where changes are expected.	Use Forest Inventory and Analysis plot information to determine trends in subalpine forests.	Expand reforestation monitoring and post-treatment monitoring.
Where can tactics be applied?	Federal lands	All lands	In reforestation or treatment locations

Table 6.7—Management recommendations for each of the evaluated species, vegetation types, and resource concerns.

Habitat, ecosystem function, or species	Primary adaptive tactics	Restoration potential	Further management recommendations
Species			
Limber pine	Promote rust resistance while preserving genetic diversity; monitor mortality rates and distributions; evaluate which stands need treatment to ensure conservation of species; determine impact of fire exclusion era.	Low to moderate – the main drawback is the exotic disease white-pine blister rust. Most management actions should be to increase rust resistance in native populations.	Initiate rust-resistant programs; identify plus trees; collect cones and determine rust resistance; map limber pine populations and identify those stands that have become established before and after the fire exclusion era.
Ponderosa pine	Restore fire to historically fire-dominated stands; reduce fuel loadings to mitigate uncharacteristic fire severities; use HRV as guide to design and conduct restoration treatments.	Moderate to high – reintroducing fire or fire surrogates into fire-excluded stands is the first step; addressing where to plant in the future is the next.	Reduce or eliminate Douglas-fir in fire-excluded stands; remove competition through thinnings and prescribed burns; monitor lower treeline in SW Montana and central Idaho to evaluate potential ponderosa pine increases.
Douglas-fir	Reduce competition and increase vigor; keep stand density low; replace Douglas-fir with ponderosa pine, western white pine, and western larch in areas where root disease is a concern; emphasize ponderosa pine on lower elevation dry forests.	Moderate to high – mitigating effects of the fire exclusion era is the first step; might be more difficult in cool, dry environs.	Keep stocking low; change species composition on sites where root disease and soil moisture deficits will be increasing; concentrate planting on the higher elevation, mesic sites and avoid planting in the lower treeline.
Western larch	Restore declining larch stands from adverse fire exclusion effects; prioritize northerly aspects and ash cap soils for treatment; reduce competition; manage larch more intensively on more xeric sites where larch may not be the species to plant on south slopes even though current climate may be suitable. Reduce forest density in all successional stages to provide larch with better chance to utilize soil moisture.	Moderate to high in west region; moderate in central region.	Cut to remove more shade-tolerant species using group selections and thinnings; consider planting options on north slopes where long term persistence will have highest potential; consider genetic stock with best adaptive traits for drought and moisture stress compatibility; maintain stocking and structure diversity to favor moisture regime for larch in all successional stages and to be more able to cope with fire; consider most intensive management for larch on sites with least moisture deficits.
Western white pine	Promote rust resistance.	Moderate in west region; low to moderate in central region, mainly due to the devastating effects of rust mortality.	Increased planting of blister rust resistant material; thinning dense stands to increase young pine vigor.
Aspen	Restore quasi-historical fire regimes; prioritize areas were aspen already exist albeit at lower than historical levels.	Moderate for all NR.	Plant aspen where it is now absent but once flourished; manage aspen at the landscape level by ensuring mix of age classes and successional stages.
Grand fir	Foster landscape heterogeneity; ensure age class structure is near HRV.	High in the west and central regions.	Low priority for restoration; no need to spend valuable restoration dollars restoring this forest unless it is locally declining.

Table 6.7(cont.)—Management recommendations for each of the evaluated species, vegetation types, and resource concerns.

Habitat, ecosystem function, or species	Primary adaptive tactics	Restoration potential	Further management recommendations
Species (cont.)			
Western redcedar	Ensure high landscape heterogeneity; multiple age classes	High in west and central NR regions	Low priority for restoration; no need to spend valuable restoration dollars restoring this forest unless it is locally declining.
Western hemlock	Maintain diversity of age classes and high landscape diversity	High in west and central NR regions	Low priority for restoration; no need to spend valuable restoration dollars restoring this forest unless it is locally declining.
Lodgepole pine	Perhaps the most important subalpine species to manage for mixed age classes and successional stages that approximate HRV	Moderate to high in most if its NR range	Allow managed fires to burn in moderate years.
Whitebark pine	Most climate change adaptation strategies are detailed in Keane et al. (2015a): promote rust resistance; conserve genetic diversity; implement treatments on the higher stands in project areas; plant rust-resistant seedlings in old or recent burns.	Low to moderate in most of its range primarily due to the devastating effects of blister rust	Follow Keane et al. (2010) rangewide strategy: protect rust-resistant plus trees; promote rust resistance; implement prescribed burn and mechanical cuttings to reduce competition; plant and direct-seed blister rust resistant seedlings on burn and treated areas; cold hardy and drought tolerant material.
Subalpine fir	Populations can best be modified by wildfire suppression efforts at local scales.	High in most of the NR	Low priority for climate impact mitigation; no need to spend valuable restoration dollars restoring this forest unless it is locally declining.
Engelmann spruce	Populations can best be modified by wildfire suppression efforts at local scales; planting may be warranted in areas with declining populations.	High in most of the NR; moderate in those low elevation wetland stands	Low priority for climate change adaptation actions; no need to spend valuable restoration dollars conserving this forest unless it is declining locally
Mountain hemlock	Increased fire suppression efforts will increase populations if needed.	Moderate to high in west and central regions of NR	Although this species may have low priority for mitigating adverse climate change effects, it is important to monitor so it does not go locally extinct.
Alpine larch	Preserve genetic diversity by collecting and storing seed.	Low to moderate across most of its range	Monitor changes in alpine larch populations; document
Green ash	Reduce grazing; increasing fire suppression and grazing can be used to modify ash populations along with planting in areas where populations are critically low.	High in the eastern NR	Plant in recently burned areas where it last existed.
Cottonwood	Foster high variability in river flows to increase seedling establishment potential; eliminate competition.	Moderate to high	Prioritize the most mesic sites first; allow fire to burn in areas that are not too dense; conduct cuttings to remove conifer competition.

Table 6.7 (cont.)—Management recommendations for each of the evaluated species, vegetation types, and resource concerns.

Vegetation types			
Dry ponderosa pine-Douglas-fir forests	Create less dense forests with composition related to moisture deficit tolerance.	High everywhere in dry settings	Manage density and appropriate composition related to projected moisture limited sites
Western larch mixed mesic forests	Create a greater landscape heterogeneity forest pattern that may limit the extent of large uncharacteristically severe disturbances using mechanical and prescribed fire as tools; greater moisture deficits, less available moisture for trees will require consideration of appropriate species distribution and forest density in the future; create within stand structure diversity and more open conditions to allow larch to maintain dominance.	High in western portion of the region	Manage pattern of the forest to increase the diversity of successional stages and arrangement; tools in areas suitable for timber production will include timber harvest and prescribed burning, and areas in roadless areas will be restored with wildfire for resource benefit with some planting where seed source is absent due to large fires; maintain stocking levels to favor larch moisture relationship in all successional stages and to be able to better cope with fire; manage species composition less susceptible to root disease especially in northern Idaho; manage larch in lynx habitat knowing that larch needs more open conditions to maintain dominance over the long term in a stand.
Mixed mesic western white pine-western redcedar-western hemlock-grand fir forests	Attempt to emphasize the restoration of western white pine while also maintaining landscape heterogeneities similar to historical conditions; promote rust resistance and plan treated and burned areas with pine seedlings.	Moderate because of blister rust	Monitor local conditions to evaluate losses or gains in the four species so management options can be adjusted.
Lodgepole pine mixed subalpine forests		High in most ares	Again, manage pattern to increase the diversity of successional stages and arrangement; roadless areas need wildfire for resource benefit; planting to lodgepole may be needed where seed source is absent due to large fires; manage species composition in lynx habitat knowing that returning natural fire regimes will reduce subalpine fir populations.
Whitebark pine mixed upper subalpine forests	Most climate change adaptation strategies are detailed in Keane et al. 2015a: most important – create landscape heterogeneity in composition and size classes; also promote rust resistance; conserve genetic diversity; implement treatments on the higher stands in project areas; plant rust-resistant seedlings in old or recent burns.	Low to moderate in most of its range primarily due to blister rust	Follow Keane et al. (2010) rangewide strategy: protect rust-resistant plus trees; promote rust resistance; implement prescribed burn and mechanical cuttings to reduce competition; plant and direct-seed blister rust resistant seedlings on burn and treated areas; cold hardy and drought tolerant material.

Table 6.7 (cont.)—Management recommendations for each of the evaluated species, vegetation types, and resource concerns.

Resource concerns			
Landscape heterogeneity	Design restoration and climate change treatments at multiple scales, especially the landscape scale; allow wildfires to burn.	Moderate	Landscape heterogeneity should be evaluated periodically to determine trends and magnitudes.
Timber production	Thinning and planting western larch, ponderosa pine and western white pine on appropriate habitats will be important to long-term timber production to enable the forest to better cope with disturbance such as fire and root disease, manage density within dry forest settings.	High in western portion of region for larch and western white pine. High for ponderosa pine everywhere.	Manage landscapes for species composition, structure and pattern best able to cope with moisture deficits and disturbance, especially root disease, bark beetles and fire. Manage dry forest settings to be more open grown ponderosa pine compared to current dense mixed Douglas-fir forests. Increase percent of ponderosa pine, western larch and western white pine on mixed mesic settings to cope with increase in fire and root disease. Manage for a diversity of lodgepole pine, spruce, subalpine fir, larch and whitebark pine in higher elevations considering increasing soil moisture deficits. Manage for a diversity of successional stages (landscape heterogeneity) in a pattern that will better cope with disturbance agents.
Carbon sequestration	Carbon can be best sequestered by embracing ecosystem management and restoring landscapes and ecosystems; carbon sequestration is a side effect of an ecosystem treatment, not the objective.	N/A	The best management approach for managing carbon sequestration is to foster a timber management program that emphasizes an ecosystem approach.

landscape diversity. Increasing diversity is a "hedge your bets" strategy that reduces risk of major forest loss. Areas with low species and genetic diversity are likely to be more susceptible to the stressors associated with climate change; thus, promoting species and genetic diversity, through plantings and in thinning treatments, is likely to increase forest resilience to changing climate. Promoting heterogeneity of species and structure across the landscape is also expected to increase resilience to wildfire, insects, and disease.

There is a lot of uncertainty associated with climate change, and managers identified several ways to increase knowledge and manage in the face of uncertainty. Implementation of an adaptive management framework can help managers deal with uncertainty and adjust management over time. In the context of climate change adaptation, adaptive management involves: definition of management goals, objectives and timeframes; analyzing vulnerabilities; determining priorities; developing adaptation strategies and tactics; implementing plans and projects; and monitoring, reviewing, and adjusting (Millar et al. 2014). Development of a consistent monitoring framework that can capture ecosystem changes with shifting climate is a key component of the adaptive management framework. For example, tracking tree species regeneration and distribution will help managers determine how species are responding to climatic changes and ways to adjust management accordingly (e.g., guidelines for planting). Integration between research and management and across resource areas (e.g., forest management and wildlife) will also be key in implementation of the adaptive management framework to ensure that the best available science is being considered in on-the-ground management and that management approaches do not conflict (e.g., effects of a particular thinning treatment on wildlife).

Managers also identified adaptation strategies and tactics to maintain particular species or community types of concern. For example, climate change is likely to lead to increased whitebark pine mortality through increased mountain pine beetle activity, fire, and white pine blister rust. There will probably also be a loss of site conditions that support whitebark pine. To promote resilient whitebark pine communities, managers may want to focus restoration efforts on sites less likely to be affected by climate change (i.e., refugia). A variety of management strategies can be implemented to promote whitebark pine, including fire management, planting at lower elevations, and removing other dominant species (e.g., lodgepole pine, spruce, and fir). Genetically selected seedlings can also be planted to promote blister rust resistance.

Finally, managers recognized that stressors associated with climate change cross boundaries, making it increasingly important that agencies coordinate and work across boundaries. Agencies can coordinate by aligning budgets and priorities for programs of work, communicating about projects adjacent to other lands, and working across boundaries to maintain roads, trails, and access that are likely to be more frequently impacted by fire and flood events under a changing climate.

Discussion

Given the high uncertainties in predicting climate, vegetation, and disturbance responses to increasing CO_2, we think that assessing vegetation change and vulnerabilities is currently more of an educated guess based on inconsistent and contradictory studies rather than a highly confident evaluation of comprehensive scientific investigation. Many of the techniques used to predict tree species response to climate change in the literature present only one possible future out of seemingly unlimited possibilities. These predictions would change if a new climate change scenario were used, if new data were augmented with existing data, if new variables were included in the analysis, if simulation parameters were modified, or if new algorithms were included in existing models. Moreover, there are still many unknowns in ecosystem science, and if we link those unknowns to the unknowns in climate systems, these uncertainties would certainly swamp any educated guesses that we might have. Consequently, these projections and assessments must be interpreted in the context of high uncertainty.

One important lesson that we learned while writing this chapter is that climate change is only one of the many challenges facing land managers, and some of these other challenges might be more important than mitigating climate change. We found that successfully mitigating ecosystem impacts from past management actions, such as fire exclusion and introductions of exotic species, will also mitigate climate change impacts. Restoring fire-prone ecosystems declining due to fire exclusion, for example, might successfully solve two issues: It would increase ecosystem health and create resilient forests that could thrive under future climate changes. These fire-dominated forests have already experienced great variation in past climate and clearly have broad amplitudes of resilience with respect to climate. In another example, fostering greater rust-resistance in our native five-needle pines may allow us to create forests that are less vulnerable to changes in climate. There will be places in the Northern Rockies where the primary challenge will be climate change impacts, such as drought at the lower treeline, but overall, we think that the best approach is to integrate climate change considerations into current management actions rather than conducting management actions for the sole purpose of climate change mitigation. Ecosystem restoration, as a prime example, could be the best approach for preparing for climate change.

The main question then is: How do we restore ecosystems in the Northern Rockies? Managers need reference conditions at the stand and landscape scales to prioritize, plan, design, and implement effective restoration activities. This becomes somewhat problematic when we need that reference to include the trend and variability of future climate. Considering the high uncertainty of future climate and vegetation projections, and knowing the resilience of fire-adapted species, we suggest that any conclusions about the infeasibility of ecosystem restoration under changing climates are imprudent. It may be more prudent to wait until simulation technology has improved to include credible pattern and process interactions with realistic regional climate dynamics for the future so that we base decisions about the restoration of ecosystems on better information. But improving ecosystem models may take decades before simulations can be used to predict species and landscape response to climate change with reasonable accuracy. While we wait, we lose valuable populations and rust-resistant trees, and our options for restoration diminish greatly. Even with climate change, restoration activities will probably be appropriate considering the high genetic variation across the range of forest species, which provides the foundation for adaptation (Bower and Aitken 2006, 2008; Mahalovich and Hipkins 2011; Rehfeldt et al. 1999). Therefore, we think that the current emphasis on ecosystem restoration in the Northern Rockies will lead to more-resilient ecosystems for the future. Until we have realistic models and less uncertain climate change projections, we desperately need a construct to use as a reference for restoration. Using historical data to guide future management actions may entail less uncertainty than building new references based on uncertain climate change projections. Therefore, we think that historical ranges and variability may provide sufficient reference conditions in the future.

We think that the concept of HRV still has a valid place in land management, at least for the near future. Landscape models can be used to simulate fire regimes and their interaction with climate and vegetation to create HRV time series that can be used as reference conditions to assess, plan, evaluate, design, and implement ecosystem restoration treatments. HRV should be used only to guide land management—not as a target on which to evaluate success or failure. There are few measures of ecosystem health that match the scale, scope, flexibility, and robustness of HRV analysis. HRV might provide a useful, though not ideal, reference for land management over the next several decades until simulation modeling advances to a level where models can forecast both accurate climate and climate responses by the ecosystems.

Conclusions

Climate change is one of many challenges facing land managers, and some of these other challenges might be more important than climate change. In addition, mitigating past ecosystem damage (e.g., fire exclusion and nonnative introductions) is a climate-smart practice. For example, restoring fire-prone ecosystems can both improve ecosystem function and create forests that will be resilient in a warmer climate. Fire-prone forests have already withstood variation in past climate and have broad amplitudes of resilience with respect to climate. There will be places in the Northern Rockies region where climate change will be the primary challenge (e.g., drought at lower treeline), but integrating climate change considerations into current management operations is preferable to a climate-centric management strategy.

Multiresource monitoring will be critical for managing ecosystems in the future, building on existing monitoring systems but with additional elements to accommodate the effects of climate change (Janowiak et al. 2014). Although costly in terms of money and personnel, an extensive monitoring system will save money in the long run by evaluating the effectiveness of adaptation tactics and providing a means to adjust them. Without monitoring, it will be impossible to know the magnitude and trend of climate effects on vegetation, or if actions proposed in this document (see section on adaptation strategies and tactics) are useful for planning and management. Monitoring data can also be used to provide inputs to calibrate and validate models. Models, in turn, can be used to simulate emergent environmental patterns, compare effects of potential treatments, identify vulnerable landscapes or ecosystem components, and bridge gaps between large-scale ecological processes and variables measured in small areas and over short periods of time. Therefore, any future land management planning will be complete only if a plan for monitoring proposed actions is included.

Acknowledgments

We acknowledge Art Zack, Megan Strom, Chris Schnepf, Paul Zambino, and Kevin Greenleaf from the U.S. Forest Service for technical reviews. We thank Gregg DiNitto, Paul Zambino, Blakey Lockman, Marcus Jackson, Joel Egan, Sandra Kegley, and Brytten Steed of U.S. Forest Service Forest Health and Protection, and Barbara Bentz, U.S. Forest Service Rocky Mountain Research Station, for additions to the disturbance sections. This work was partially funded by the Rocky Mountain Research Station Fire, Fuels and Smoke Program and U.S. Forest Service Office of Sustainability and Climate.

References

Agee, J.K. 1991. Fire history of Douglas-fir forests in the Pacific Northwest. In: Ruggiero, L.F.; Aubry, K.B.; Carey, A.B.; [et al.], eds. Wildlife and vegetation of unmanaged Douglas-fir forests. Gen. Tech. Rept. PNW-GTR-285. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 25–33.

Ahlqvist, O.; Shortridge A. 2010. Spatial and semantic dimensions of landscape heterogeneity. Landscape Ecology. 25: 573–590.

Aitken, S.N.; Yeaman, S.; Holliday, J.A.; [et al.]. 2008. Adaptation, migration or extirpation: Climate change outcomes for tree populations. Evolutionary Applications. 1: 95–111.

Alberto, F.J.; Aitken, S.N.; Alía, R.; [et al.]. 2013. Potential for evolutionary responses to climate change—Evidence from tree populations. Global Change Biology. 19: 1645–1661.

Alexander, R.R.; Shearer, R.C.; Shepperd, W.D. 1990. *Abies lasiocarpa (Hook.) Nutt*. subalpine fir. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1 conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 149–166.

Alexander, R.R.; Shepperd, W.D. 1990. *Picea engelmannii Parry ex Engelm*. Engelmann spruce. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1 conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 403–444.

Allen, C.D.; Macalady, A.K.; Chenchouni, H.; [et al.]. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management. 259: 660–684.

Allison, S.D.; Treseder, K.K. 2008. Warming and drying suppress microbial activity and carbon cycling in boreal forest soils. Global Change Biology. 14: 2898–2909.

Amaranthus, M.P.; Cazares, E.; Perry, D.A. 1999. The role of soil organisms in restoration. Proceedings: Pacific Northwest Forest & Rangeland Soil Organism Symposium: Organism functions and processes, management effects on organisms and processes, and role of soil organisms in restoration. LaSells Stewart Center, Oregon State University, Corvallis, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 179–189.

Anderegg, W.R.L.; Berry, J.A.; Smith, D.D.; [et al.]. 2012. The roles of hydraulic and carbon stress in a widespread climate-induced forest die-off. Proceedings of the National Academy of Sciences. 109: 233–237.

Anderson, L.J.; Winterton, A.J. 1996. Germination as a determinant of seedling distributions among natural substrates in *Picea Engelmanni* (Pinaceae) and *Abies lasiocarpa* (Pinaceae). American Journal of Botany. 83: 112–117.

Antos, J.A. 1972. Grand fir (*Abies grandis* (Dougl.) Forbes) forests of the Swan Valley, Montana. Thesis. Missoula, MT: University of Montana. 130 p.

Antos, J.A.; Shearer, R.C. 1980. Vegetation development on disturbed grand fir sites, swan valley, northwestern Montana. Res. Pap. INT-RP-251. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 53 p.

Araujo, M.B.; Whittaker, R.J.; Ladle, R.J.; [et al.]. 2005. Reducing uncertainty in projections of extinction risk from climate change. Global Ecology and Biogeography Letters. 14: 529–538.

Armson, K.A. 1977. Forest soils: Properties and processes. University of Toronto Press.

Arno, S.F. 1980. Forest fire history of the northern Rockies. Journal of Forestry. 78: 460–465.

Arno, S.F. 1988. Fire ecology and its management implications in ponderosa pine forests. In: Ponderosa pine: The species and its management; symposium proceedings; 1987 September 29–October 1. Spokane, WA: Washington State University Extension Service: 133–139.

Arno, S.F. 1990. *Larix lyallii* Parl. alpine larch. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1 conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 330–347.

Arno, S.F. 1998. Fire disturbance and associated impacts on forest values: Some implications of fire exclusion. In: Proceedings from the Western Forest Insect Work Conference; 1998 March 2–March 5; Jackson, WY. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 1–3.

Arno, S.F. 2001. Community types and natural disturbance processes. In: Tomback, D.F.; Arno, S.F.; Keane, R.E., eds. Whitebark pine communities: Ecology and restoration. Washington, DC: Island Press: 74–89.

Arno, S.F. 2010. The Seeley Lake larch: Living link to Indian and frontier history. Forest History Today Spring/Fall 2010: 13–19.

Arno, S.F.; Gruell, G.E. 1983. Fire history at the forest-grassland ecotone in southwestern Montana. Journal of Range Management. 36: 332–336.

Arno, S.F.; Gruell, G.E. 1986. Douglas-fir encroachment into mountain grasslands in southwestern Montana. Journal of Range Management. 39: 272–275.

Arno, S.F.; Habeck, J.R. 1972. Ecology of alpine larch (*Larix lyallii* Parl.) in the Pacific Northwest. Ecological Monographs. 42: 417–450.

Arno, S.F.; Hoff, R. 1990. *Pinus albicaulis* Engelm. whitebark pine. In: Silvics of North America. Volume I conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 268–279.

Arno, S.F.; Parsons, D.J.; Keane, R.E. 2000. Mixed-severity fire regimes in the northern Rocky Mountains: Consequences of fire exclusion and options for the future. In: Wilderness science in a time of change conference, volume 5: Wilderness ecosystems, threat and management; 1999 May 23–27; Missoula, MT. Proceedings RMRS-P-15-Vol 5. Fort Collins CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 225–232.

Arno, S.F.; Reinhardt, E.D.; Scott, J.H. 1993. Forest structure and landscape patterns in a subalpine lodgepole pine type: A procedure for quantifying past and present conditions. Gen. Tech. Rep. INT-294. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 17 p.

Arno, S.F.; Scott, J.H.; Hartwell, M.G. 1995. Age-class structure of old growth ponderosa pine/Douglas-fir stands and its relationship to fire history. Res. Pap. INT-RP-481. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 25 p.

Arno, S.F.; Simmerman, D.G.; Keane, R.E. 1986. Characterizing succession within a forest habitat type: An approach designed for resource managers. Res. Note INT-RN-357. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 8 p.

Arno, S.F.; Smith, H.Y.; Krebs, M.A. 1997. Old growth ponderosa pine and western larch stand structures: Influences of pre-1900 fires and fire exclusion. Res. Pap. INT-RP-495. Ogden, UT: U.S. Department of Agiculture, Forest Service, Intermountain Research Station. 20 p.

Aston, I.W. 2010. Observed and projected ecological response to climate change in the Rocky Mountains and Upper Columbia Basin: A synthesis of current scientific literature. Natural Resource Report NPS/ROMN/NPR—2010/220. Fort Collins, CO: National Park Service. 98 p.

Attiwill, P.M.; Leeper, G.W. 1987. Forest soils and nutrient cycles. New York: Springer Press.

Auble, G.T.; Scott, M.L. 1998. Fluvial disturbance patches and cottonwood recruitment along the upper Missouri River, Montana. Wetlands. 18: 546–556.

Axelson, J.N.; Alfaro, R.I.; Hawkes, B.C. 2009. Influence of fire and mountain pine beetle on the dynamics of lodgepole pine stands in British Columbia, Canada. Forest Ecology and Management. 257: 1874–1882.

Ayres, M.P.; Lombardero, M.J. 2000. Assessing the consequences of global change for forest disturbance from herbivores and pathogens. The Science of the Total Environment. 262: 263–286.

Bachelet, D.; Neilson, R.P.; Hickler, T.; [et al.]. 2003. Simulating past and future dynamics of natural ecosystems in the United States. Global Biogeochemical Cycles. 17: 1045.

Bachelet, D.; Neilson, R.P.; Lenihan, J.M.; [et al.]. 2001. Climate change effects on vegetation distribution and carbon budget in the United States. Ecosystems. 4: 164–185.

Baker, W.L. 1989. A review of models of landscape change. Landscape Ecology. 2: 111–133.

Baker, W.L. 2009. Fire ecology in Rocky Mountain landscapes. Washington, DC: Island Press. 628 p.

Baldocchi, D. 2008. Breathing of the terrestrial biosphere: Lessons learned from a global network of carbon dioxide flux measurement systems. Australian Journal of Botany. 56: 1–26.

Bannerman, S. 1997. Landscape ecology and natural disturbances: Relationships to biodiversity. Victoria, BC: British Columbia Ministry of Forests, Research Program. 9 p.

Barrett, S.W. 1988. Fire regime classification for coniferous forests of the northwestern United States. Final Report RJVA-INT-82123. Missoula, MT: U.S. Department of Agriculture, Forest Service.

Barrett, S.W.; Arno, S.F. 1991. Classifying fire regimes and defining their topographic controls in the Selway-Bitterroot Wilderness. In: Andrews, P.L.; Potts, D.F., eds. Proceedings of the 11th Conference on Fire and Forest Meteorology. Missoula, MT: Society of American Foresters: 299–307.

Barrett, S.W.; Arno, S.F.; Key, C.H. 1991. Fire regimes of western larch-lodgepole pine forests in Glacier National Park, Montana. Canadian Journal of Forest Research. 21: 1711–1720.

Barrows, J.S.; Sandberg, D.V.; Hart, J.D. 1977. Lightning fires in Northern Rocky Mountain forests. Unpublished report on file with: U.S. Department of Agriculture, Forest Service, Intermountain Fire Sciences Laboratory. P.O. Box 8089, Missoula, MT. 221 p.

Bartlein, P.J.; Whitlock, C.; Shafer, S.L. 1997. Future climate in the Yellowstone National Park region and its potential impact on vegetation. Conservation Biology. 11: 782–792.

Bartos, D. 1998. Aspen, fire and wildlife. Spokane, WA: The Wildlife Society, Northwest Section, Oregon and Washington Chapters: 44–48.

Bartos, D.L. 1978. Modeling plant succession in aspen ecosystems. In: Hyder, D.N., ed. Proceedings of the First International Rangeland Congress. Denver, CO: Society for Range Management: 208–211.

Baumeister, D.; Callaway, R.M. 2006. Facilitation by *Pinus flexilis* during succession: A hierarchy of mechanisms benefits other plant species. Ecology. 87: 1816–1830.

Baumgartner, D.M.; Lotan, J.E.; Tonn, J.R. 1994. Interior cedar-hemlock-white pine forests: Ecology and management. Pullman, WA: Washington State University Extension Service. 367 p.

Beaufait, W.R.; Hardy, C.E.; Fischer, W.C. 1977. Broadcast burning in larch-fir clearcuts: the Miller Creek-Newman Ridge study. Res. Pap. INT-RP-175. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 53 p.

Bell, D.M.; Bradford, J.B.; Lauenroth, W.K. 2014. Early indicators of change: Divergent climate envelopes between tree life stages imply range shifts in the western United States. Global Ecology and Biogeography. 23: 168–180.

Bentz, B.J.; Jönsson, A.M. 2015. Modeling bark beetle responses to climate change. In: Vega, F.; Hofstetter, R., eds. Bark beetles, biology and ecology of native and invasive species. Elsevier. 640 p.

Bentz, B.J.; Régnière, J.; Fettig, C.J.; [et al.]. 2010. Climate change and bark beetles of the western United States and Canada: Direct and indirect effects. BioScience. 60: 602–613.

Beschta, R.L.; Ripple, W.J. 2005. Rapid assessment of riparian cottonwood recruitment: Middle Fork John Day River, northeastern Oregon. Ecological Restoration. 23: 150–156.

Bigler, C.; Kulakowski, D.; Veblen, T.T. 2005. Multiple disturbance interactions and drought influence fire severity in Rocky Mountain subalpine forests. Ecology. 86: 3018–3029.

Bollenbacher, B. 2012. Characteristics of primary tree species in the Northern Region. Missoula, MT: U.S. Department of Agriculture, Forest Service. Northern Regional Office. 10 p.

Boldt, C.E.; Uresk, D.W.; Severson, K.E. 1978. Riparian woodlands in jeopardy on the northern High Plains. In: Johnson, R.R.; McCormick, J.F., eds. Proceedings of the national symposium on strategies for protection and management of floodplain wetlands and other riparian ecosystems. Gen. Tech. Rep. WO-GTR-12. Washington, DC: U.S. Department of Agriculture, Forest Service: 184–189.

Bonan, G.B. 2008. Forests and climate change: Forcings, feedbacks, and the climate benefits of forests. Science. 320: 1444–1449.

Borman, M.M.; Larson, L.L. 2002. Cottonwood: Establishment, survival, and stand characteristics. Corvallis, OR: Extension Service, Oregon State University. 12 p.

Botkin, D.B.; Saxe, H.; Araujo, M.B.; [et al.]. 2007. Forecasting the effects of global warming on biodiversity. BioScience. 57: 227–236.

Bower, A.D.; Aitken, S.N. 2006. Geographic and seasonal variation in cold hardiness of whitebark pine. Canadian Journal of Forest Research. 36: 1842–1850.

Bower, A.D.; Aitken, S.N. 2007. Mating system and inbreeding depression in whitebark pine (*Pinus albicaulis* Engelm.). Tree Genetics and Genomes. 3: 379–388.

Bower, A.D.; Aitken, S.N. 2008. Ecological genetics and seed transfer guidelines for *Pinus albicaulis* (Pinaceae). American Journal of Botany. 95: 66–76.

Brandt, J.P.; Cerezke, H.F.; Mallet, K.I.; [et al.]. 2003. Factors affecting trembling aspen (*Populus tremuloides* Michx.) health in Alberta, Saskatchewan, and Manitoba, Canada. Forest Ecology and Management. 178: 287–300.

Broadhurst, L.M.; Lowe, A.; Coates, D.J.; [et al.]. 2008. Seed supply for broadscale restoration: Maximizing evolutionary potential. Evolutionary Applications. 1: 587–597.

Brown, J.K. 1996. Fire effects on aspen and cottonwood. In: Aspen and cottonwood in the Blue Mountains Workshop; 1996 April 2–4. La Grande, OR: Blue Mountains Natural Resources Institute. 11 p.

Brown, P.M.; Schoettle, A.W. 2008. Fire and stand history in two limber pine (*Pinus flexilis*) and Rocky Mountain bristlecone pine (*Pinus aristata*) stands in Colorado. International Journal of Wildland Fire. 17: 339–347.

Brunelle, A.; Whitlock, C.; Bartlein, P.; [et al.]. 2005. Holocene fire and vegetation along environmental gradients in the Northern Rocky Mountains. Quaternary Science Reviews. 24: 2281–2300.

Bugmann, H. 2001. A review of forest gap models. Climatic Change. 51: 259–305.

Bugmann, H.; Cramer, W. 1998. Improving the behavior of forest gap models along drought gradients. Forest Ecology and Management. 103: 247–263.

Bunn, A.G.; Lawrence, R.L.; Bellante, G.J.; [et al.]. 2003. Spatial variation in distribution and growth patterns of old growth strip-bark pines. Arctic, Antarctic, and Alpine Research. 35: 323–330.

Burns, R.M.; Honkala, B.H. 1990. Silvics of North America: Volume I conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service. 675 p.

Burton, P.J.; Cumming, S.G. 1995. Potential effects of climatic change on some western Canadian forests, based on phenological enhancement to a patch model of forest succession. Water, Air and Soil Pollution. 82: 401–414.

Butler, D.R. 1986. Conifer invasion of subalpine meadows, central Lemhi Mountains, Idaho. Northwest Science. 60: 166–173.

Callaway, R.; Sala, A.; Carey, E.; [et al.]. 1998. Replacement of whitebark pine by subalpine fir: The consequences for stand carbon, water, and nitrogen cycles. Contract Completion Report Research Joint Venture Agreement INT-95086-RJVA. Missoula, MT: U.S. Department of Agriculture, Forest Service, Fire Sciences Laboratory. 31 p.

Campbell, R.B., Jr.; Bartos, D. 2001. Aspen ecosystems—Objectives for sustaining biodiversity. In: Sustaining aspen ecosystems in western landscapes. Proceedings. RMRS-P-18. Grand Junction, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 299–306.

Candau, J.N.; Abt, V.; Keatley, L. 2002. Bioclimatic analysis of declining aspen stands in northeastern Ontario. Ontario Forest Research Report. Toronto: Ontario Ministry of Natural Resources. 154 p.

Carlson, C.E.; Arno, S.F.; Menakis, J. 1990. Hybrid larch of the Carlton Ridge Research Natural Area in Western Montana. Natural Areas Journal. 10: 134–139.

Carroll, A.L.; Taylor, S.W.; Régnière, J.; [et al.]. 2003. Effects of climate change on range expansion by the mountain pine beetle in British Columbia. In: Mountain pine beetle symposium: Challenges and solutions. Victoria, BC: Natural Resources Canada, Canadian Forest Service: 223–231.

Cayan, D.R.; Dettinger, M.D.; Kammerdiener, S.A.; [et al.]. 2001. Changes in the onset of spring in the western United States. Bulletin of the American Meteorological Society. 82: 399–415.

Chakraborty, S.; Luck, J.; Hollaway, G.; [et al.] 2008. Impacts of global change on diseases of agricultural crops and forest trees. CAB reviews: Perspectives in agriculture, veterinary science, nutrition and natural resources. 3: 1–15.

Chapman, T.B.; Veblen, T.T.; Schoennagel, T. 2012. Spatiotemporal patterns of mountain pine beetle activity in the southern Rocky Mountains. Ecology. 93: 2175–2185.

Chhin, S.; Hogg, E.T.; Lieffers, V.J.; [et al.]. 2008. Potential effects of climate change on the growth of lodgepole pine across diameter size classes and ecological regions. Forest Ecology and Management. 256: 1692–1703.

Chmura, D.J.; Anderson, P.D.; Howe, G.T.; [et al.]. 2011. Forest responses to climate change in the northwestern United States: Ecophysiological foundations for adaptive management. Forest Ecology and Management. 261: 1121–1142.

Cissel, J.H.; Swanson, F.J.; McKee, W.A.; [et al.]. 1994. Using the past to plan the future in the Pacific Northwest. Journal of Forestry. 92: 30–31, 46.

Clark, J.; Loehman, R.; Keane, R. 2017. Climate changes and wildfire alter vegetation of Yellowstone National Park, but forest cover persists. Ecosphere. 8: e01636.

Clark, J.S. 1989. Ecological disturbance as a renewal process: Theory and application to fire history. Oikos. 56: 17–30.

Clark, J.S.; Bell, D.M.; Hersh, M.H.; [et al.]. 2011. Climate change vulnerability of forest biodiversity: Climate and competition tracking of demographic rates. Global Change Biology. 17: 1834–1849.

Clark, J.S.; Carpenter, S.R.; Barber, M.; [et al.]. 2001. Ecological forecasts: An emerging imperative. Science. 293: 657–660.

Cohn, J.S.; Di Stefano, J.; Christie, F.; [et al.]. 2015. How do heterogeneity in vegetation types and post-fire age-classes contribute to plant diversity at the landscape scale? Forest Ecology and Management. 346: 22–30.

Colenutt, M.E.; Luckman, B.H. 1991. Dendrochronological investigation of *Larix lyallii* at Larch Valley, Alberta. Canadian Journal of Forest Research. 21: 1222–1233.

Collins, B.M.; Stephens, S.L. 2010. Stand-replacing patches within a ‘mixed severity’ fire regime: Quantitative characterization using recent fires in a long-established natural fire area. Landscape Ecology. 25: 927–939.

Coop, J.D.; Schoettle, A.W. 2009. Regeneration of Rocky Mountain bristlecone pine (*Pinus aristata*) and limber pine (*Pinus flexilis*) three decades after stand-replacing fires. Forest Ecology and Management. 257: 893–903.

Cooper, S.V.; Neiman, K.E.; Roberts, D.W. 1991. Forest habitat types of northern Idaho: A second approximation. Gen. Tech. Rep. INT-236. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 143 p.

Coops, N.C.; Waring, R.H. 2011. A process-based approach to estimate lodgepole pine (*Pinus contorta* Dougl.) distribution in the Pacific Northwest under climate change. Climatic Change. 105: 313–328.

Cramer, W.; Bondeau, A.; Woodward, F.I.; [et al.]. 2001. Global response of terrestrial ecosystem structure and function to CO_2 and climate change: Results from six dynamic global vegetation models. Global Change Biology. 7: 357–373.

Creeden, E.P.; Hicke, J.A.; Buotte, P.C. 2014. Climate, weather, and recent mountain pine beetle outbreaks in the western United States. Forest Ecology and Management. 312: 239–251.

Crimmins, S.M.; Dobrowski, S.Z.; Greenberg, J.A.; [et al.]. 2011. Changes in climatic water balance drive downhill shifts in plant species' optimum elevations. Science. 331: 324–327.

Cripps, C.; Miller, O.K. 1993. Ectomycorrhizal fungi associated with aspen on three sites in the north-central Rocky Mountains. Canadian Journal of Botany. 71: 1414–1420.

Cumming, S.G.; Burton, P.J. 1996. Phenology-mediated effects of climatic change on some simulated British Columbia forests. Climatic Change. 34: 213–222.

Cushman, S.A.; McKenzie, D.; Peterson, D.L.; [et al.]. 2007. Research agenda for integrated landscape modeling. Gen. Tech. Rep. RMRS-GTR-194. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 50 p.

Dale, V.H.; Joyce, L.A.; McNulty, S.; [et al.]. 2001. Climate change and forest disturbances. BioScience. 51: 723–734.

Daly, C.; Bachelet, D.; Lenihan, J.M.; [et al.]. 2000. Dynamic simulation of tree-grass interactions for global change studies. Ecological Applications. 10: 449–469.

Daly, C.; Halbleib, M.; Smith, J.I.; [et al.]. 2008. Physiographically sensitive mapping of climatological temperature and precipitation across the conterminous United States. International Journal of Climatology. 28: 2031–2064.

Davidson, E.A.; Janssens, I.A. 2006. Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. Nature. 440:165–173.

Davis, K.M. 1980. Fire history of a western larch/Douglas-fir forest type in northwestern Montana. In: Proceedings of the fire history workshop; 1980 October 20–24; Tucson, Arizona. RM-GTR-81. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 69–74.

Davis, M.B. 1989. Lags in vegetation response to greenhouse warming. Climatic Change. 15: 75–82.

Davis, M.B.; Shaw, R.G. 2001. Range shifts and adaptive responses to Quaternary climate change. Science. 292: 673–679.

Davis, M.B.; Shaw, R.G.; Etterson, J.R. 2005. Evolutionary responses to changing climates. Ecology. 86: 1704–1714.

Debell, D.S. 1990. *Populus trichocarpa* Torr. & Gray black cottonwood. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 2 hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 1322–1355.

DeByle, N.V. 1985. Wildlife. In: Debyle, N.F., Winokur, R.P., eds. Aspen: Ecology and management in the western United States. Gen. Tech. Rep. RM-GTR-119. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 125–155.

DeNitto, G. 2013. FHP annual report for the Northern Region. Forest Health and Protection, R1-FHP-2013. Missoula, MT: U.S. Department of Agriculture, Forest Service. 23 p.

Denman, K.L.; Brasseur, G.; Chidthaisong, A.; [et al.]. 2007. Couplings between changes in the climate system and biogeochemistry. In: Solomon, S.; Qin, D.; Manning, M. M.; [et al.], eds. Climate change 2007: The physical science basis. Contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, UK; New York: Cambridge University Press: 541–584.

Despain, D.G. 1983. Nonpyrogenous climax lodgepole pine communities in Yellowstone National Park. Ecology. 64: 231–234.

Devine, W.; Aubry, C.; Bower, A.; [et al.]. 2012. Climate Change and forest trees in the Pacific Northwest: A vulnerability assessment and recommended actions for National Forests. Olympia, WA: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 102 p.

Dickman, A.; Cook, S. 1989. Fire and fungus in a mountain hemlock forest. Canadian Journal of Botany. 67: 2005–2016.

Dooley, S.R.; Treseder, K.K. 2012. The effect of fire on microbial biomass: a meta-analysis of field studies. Biogeochemistry. 109: 49–61.

Dullinger, S.; Gattringer, A.; Thuiller, W.; [et al.]. 2012. Extinction debt of high-mountain plants under twenty-first-century climate change. Nature Climate Change. 2: 619–622.

Eamus, D.; Jarvis, P.G. 1989. The direct effects of increase in the global atmospheric CO_2 concentration on natural and commercial temperate trees and forests. Advances in Ecological Research. 19: 1–55.

Easterling, D.R.; Meehl, G.A.; Parmesan, C.; [et al.]. 2000. Climate extremes: Observations, modeling, and impacts. Science. 289: 2068–2074.

Egan, D.; Howell, E.A., eds. 2001. The historical ecology handbook. Washington, DC: Island Press. 457 p.

Ehleringer, J.R.; Cerling, T.E. 1995. Atmospheric CO_2 and the ratio of intercellular to ambient CO_2 concentrations in plants. Tree Physiology. 15: 105–111.

Eisenberg, C.; Seager, S.T.; Hibbs, D.E. 2013. Wolf, elk, and aspen food web relationships: Context and complexity. Forest Ecology and Management. 299: 70–80.

Etterson, J.R.; Shaw, R.G. 2001. Constraint to adaptive evolution in response to global warming. Science. 294: 151–154.

Falk, D.A.; Miller, C.; McKenzie, D.; [et al.]. 2007. Cross-scale analysis of fire regimes. Ecosystems. 10: 809–823.

Feldman, R.; Tomback, D.F.; Koehler, J. 1999. Cost of mutualism: Competition, tree morphology, and pollen production in limber pine clusters. Ecology. 80: 324–329.

Ferguson, D.E.; Johnson, F.D. 1996. Classification of grand fir mosaic habitats. Gen. Tech. Rep. INT-GTR-337. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 16 p.

Ferry, G.W.; Clark, R.G.; Montgomery, R.E; [et al.]. 1995. Altered fire regimes within fire-adapted ecosystems. In: LaRoe, E.T.; Farris, G.S.; Puckett, C.E. Our living resources: A report to the nation on the distribution, abundance, and health of U.S. plants, animals, and ecosystems. Washington, DC: U.S Department of the Interior, National Biological Service: 222–224.

Fiedler, C.E.; Lloyd, D.A. 1995. Autecology and synecology of western larch. In: Schmidt, W.C.; McDonald K.J., comps. Proceedings of a symposium on ecology and management of *Larix* forests: A look ahead. Gen. Tech. Rep. GTR-INT-319. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 118–122.

Fins, L.; Steeb, L.W. 1986. Genetic variation in allozymes of western larch. Canadian Journal of Forest Research. 16: 1013–1018.

Fins, L.; Byler, J.; Ferguson, D.; [et al.]. 2002. Return of the giants: Restoring white pine ecosystems by breeding and aggressive planting of blister rust-resistant white pines. Journal of Forestry. 100: 20–26.

Fischer, W.C.; Clayton, B.D. 1983. Fire ecology of Montana forest habitat types east of the Continental Divide. Gen. Tech. Rep. INT-141. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 83 p.

Fitter, A.H.; Heinemeyer, A.; Staddon, P.L. 2000. The impact of elevated CO_2 and global climate change on arbuscular mycorrhizas: A mycocentric approach. New Phytologist. 147: 179–187.

Flannigan, M.D.; Amiro, B.D.; Logan, K.A.; [et al.]. 2005. Forest fires and climate change in the 21st century. Mitigation and Adaptation Strategies for Global Change. 11: 847–859.

Flannigan, M.D.; Krawchuk, M.A.; de Groot, W.J.; [et al.]. 2009. Implications of changing climate for global wildland fire. International Journal of Wildland Fire. 18: 483–507.

Foiles, M.W.; Graham, R.T.; Olson, D.F. 1990. *Abies grandis* (Dougl. ex D. Don) Lindl. grand fir. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1 conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 132–155.

Fowells, H.A.; Kirk, B.M. 1945. Availability of soil moisture to ponderosa pine. Journal of Forestry. 43: 601–604.

Fowler, H.J.; Blenkinsop, S.; Tebaldi, C. 2007. Linking climate change modelling to impacts studies: Recent advances in downscaling techniques for hydrological modelling. International Journal of Climatology. 27: 1547–1578.

Franklin, J.F.; Swanson, F.J.; Harmon, M.E.; [et al.]. 1991. Effects of global climatic change on forests in Northwestern North America. The Northwest Environmental Journal. 7: 233–254.

Frey, B.R.; Lieffers, V.J.; Hogg, E.H.; [et al.]. 2004. Predicting landscape patterns of aspen dieback: Mechanisms and knowledge gaps. Canadian Journal of Forest Research. 34: 1379–1390.

Fulé, P.Z.; Cocke, A.E.; Heinlein, T.A.; [et al.]. 2004. Effects of an intense prescribed forest fire: Is it ecological restoration. Restoration Ecology. 12: 220–230.

Funk, J.; Saunders, S. 2014. Rocky Mountain forests at risk: Confronting climate-driven impacts from insects, wildfires, heat, and drought. Cambridge, MA: Union of Concerned Scientists. 64 p.

Garcia-Gonzalo, J.; Peltola, H.; Briceño-Elizondo, E.; [et al.]. 2007. Effects of climate change and management on timber yield in boreal forests, with economic implications: A case study. Ecological Modelling. 209: 220–234.

Gavin, D.G.; Hallett, D.J.; Hu, F.S.; [et al.]. 2007. Forest fire and climate change in western North America: Insights from sediment charcoal records. Frontiers in Ecology and the Environment. 5: 499–506.

Geiszler, D.R.; Gara, R.I.; Driver, C.H.; [et al.]. 1980. Fire, fungi, and beetle influences on a lodgepole pine ecosystem of south-central Oregon. Oecologia. 46: 239–243.

Gershunov, A.; Barnett, T.P. 1998. Interdecadal Modulation of ENSO Teleconnections. Bulletin of the American Meteorological Society. 79: 2715–2725.

Gillette, N.E.; Wood, D.L.; Hines, S.J.; [et al.]. 2014. The once and future forest: Consequences of mountain pine beetle treatment decisions. Forest Science. 60: 527–538.

Girard, M.M.; Goetz, H.; Bjustad, A.J. 1987. Factors influencing woodlands of southwestern North Dakota. Prairie Naturalist. 19: 189–198.

Girardin, M.P.; Raulier, F.; Bernier, P.Y.; [et al.]. 2008. Response of tree growth to a changing climate in boreal central Canada: A comparison of empirical, process-based, and hybrid modelling approaches. Ecological Modelling. 213: 209–228.

Glick, P.; Stein, B.; Edelson, N.A. 2010. Scanning the conservation horizon: A guide to climate change vulnerability assessment. Washington, DC: National Wildlife Federation.

Gom, L.A.; Rood, S.B. 1999. Fire induces clonal sprouting of riparian cottonwoods. Canadian Journal of Botany. 77: 1604–1616.

Goward, S.N.; Masek, J.G.; Cohen, W.; [et al.]. 2008. Forest disturbance and North American carbon flux. Eos. 89: 105–106.

Gower, S.T.; Kloeppel, B.D.; Reich, P.B. 1995. Carbon, nitrogen, and water use by larches and co-occurring evergreen conifers. In: Ecology and management of *Larix* forests: A look ahead. October 5–9, 1992, Whitefish, MT. Gen. Tech. Rep. GTR-INT-319. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region: 110–117.

Graham, R.T. 1990. *Pinus monticola* Dougl. ex D. Don western white pine. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1 conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 348–353.

Graumlich, L.J.; Brubaker, L.B.; Grier, C.C. 1989. Long-term trends in forest net primary productivity: Cascade Mountains, Washington. Ecology. 70: 405–410.

Gray, L.K.; Hamann, A. 2013. Tracking suitable habitat for tree populations under climate change in western North America. Climatic Change. 117: 289–303.

Grissino-Mayer, H.D.; Romme, W.H.; Floyd, M.L.; Hanna, D.D. 2004. Climatic and human influences on fire regimes of the southern San Juan Mountains, Colorado, USA. Ecology. 85: 1708–1724.

Gruell, G.E.; Schmidt, W.C.; Arno, S.F.; [et al.]. 1982. Seventy years of vegetative change in a managed ponderosa pine forest in western Montana: Implications for resource management. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 41 p.

Guisan, A.; Zimmermann, N.E. 2000. Predictive habitat distribution models in ecology. Ecological Modelling. 135: 147–186.

Gustafson, E. 2013. When relationships estimated in the past cannot be used to predict the future: Using mechanistic models to predict landscape ecological dynamics in a changing world. Landscape Ecology. 28: 1429–1437.

Gworek, J.R.; Vander Wall, S.B.; Brussard, P.F. 2007. Changes in biotic interactions and climate determine recruitment of Jeffrey pine along an elevation gradient. Forest Ecology and Management. 239: 57–68.

Habeck, J.R. 1978. A study of climax western redcedar (*Thuja plicata* Donn.) forest communities in the Selway-Bitterroot Wilderness, Idaho. Northwest Science. 52: 67–76.

Habeck, J.R. 1990. Old-growth ponderosa pine-western larch forests in western Montana: ecology and management. The Northwest Environmental Journal. 6: 271–292.

Hagle, S.K.; Gibson, K.; Tunnock, S. 2003. Field guide to diseases and insect pests of northern and central Rocky Mountain conifers. Rep. Num. R1-03-08. Missoula, MT: U.S. Department of Agriculture, Forest Service, State and Private Forestry, Northern Region and Intermountain Region. 197 p.

Haire, S.L.; McGarigal, K. 2010. Effects of landscape patterns of fire severity on regenerating ponderosa pine forests (*Pinus ponderosa*) in New Mexico and Arizona, USA. Landscape Ecology. 25: 1055–1069.

Hamann, A.; Wang, T. 2006. Potential effects of climate change on ecosystem and tree species distribution in British Columbia. Ecology. 87: 2733–2786.

Hamrick, J.L. 2004. Response of forest trees to global environmental changes. Forest Ecology and Management. 197: 323–335.

Hamrick, J.L.; Godt, M.J.; Sherman-Broyles, S.L. 1992. Factors influencing levels of genetic diversity in woody plant species. New Forests. 6: 95–124.

Hann, W.J.; Jones, J.L.; Karl, M.G.; [et al.]. 1997. Landscape dynamics of the basin. In: Quigley, T.M.; Arbelbide, S.J., eds. An assessment of ecosystem components in the Interior Columbia Basin and portions of the Klamath and Great Basins: Volume II. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 338–1055.

Hann, W.J.; Keane, R.E.; McNicoll, C.; [et al.]. 1994. Ecology of a western Montana landscape with cedar potential. In: Interior cedar-hemlock-white pine forests: Ecology and management symposium proceedings; 1993 March 2–4; Spokane, WA. Pullman WA: Washington State University: 69–70.

Hanninen, H. 1995. Effects of climatic change on trees from cool and temperate regions: An ecophysiological approach to modelling of bud burst phenology. Canadian Journal of Botany. 73: 183–199.

Hansen, A.J.; Neilson, R.P.; Dale, V.H.; [et al.]. 2001. Global change in forests: Responses of species, communities, and biomes: Interactions between climate change and land use are projected to cause large shifts in biodiversity. BioScience. 51: 765–779.

Hansen, A.J.; Phillips, L.B. 2015. Which tree species and biome types are most vulnerable to climate change in the US Northern Rocky Mountains? Forest Ecology and Management. 338: 68–83.

Hansen, H.P. 1942. The influence of volcanic eruptions upon post-Pliocene forest succession in central Oregon. American Journal of Botany. 29: 214–219.

Hansen, H.P. 1947. Post-glacial vegetation of the northern Great Basin. American Journal Botany. 34: 164–171.

Hansen, H.P. 1949. Post-glacial forests in west central Alberta, Canada. Torrey Botanical Club Bulletin. 76: 278–289.

Hardy, C.C.; Keane, R.E.; Stewart, C.A. 2000. Ecosystem-based management in the lodgepole pine zone. In: The Bitterroot ecosystem management research project: What we have learned; 1999 May 18–20; Missoula, MT. Proceedings RMRS-P-17. Fort Collins CO: U.S. Deptartment of Agriculture, Forest Service, Rocky Mountain Research Station: 31–35.

Harrington, M.G. 2012. Duff mound consumption and cambium injury for centuries-old western larch from prescribed burning in western Montana. International Journal of Wildland Fire. 22: 359–367.

Hart, S.J.; Veblen, T.T.; Eisenhart, K.S.; [et al.]. 2013. Drought induces spruce beetle (*Dendroctonus rufipennis*) outbreaks across northwestern Colorado. Ecology. 95: 930–939.

Harvey, A.E.; Byler, J.W.; McDonald, G.I.; [et al.]. 2008. Death of an ecosystem: Perspectives on western white pine ecosystems of North America at the end of the twentieth century. Gen. Tech. Rep. RMRS-GTR-208. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 10 p.

Harvey, A.E.; Jurgensen, M.F.; Larsen, M.J. 1978. Seasonal distribution of ectomycorrhizae in a mature Douglas-fir/larch forest soil in western Montana. Forest Science. 24: 203–208.

Hasumi, H.; Emori, S., eds. 2004. K-1 coupled model (MIROC) description. Tech. Rep. 1. Tokyo: University of Tokyo, Center for Climate System Research.

He, H.; Keane, R.E.; Iverson, L. 2008. Forest landscape models, a tool for understanding the effect of the large-scale and long-term landscape processes. Forest Ecology and Management. 254: 371–374.

Heidmann, L.J. 1983. Seed production in southwestern ponderosa pine on a sedimentary soil. Res. Note. RM-RN-434. Flagstaff, AZ: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 2 p.

Hepting, G.H. 1963. Climate and forest diseases. Annual Review of Phytopathology. 1: 31–50.

Hermann, R.K.; Lavender, D.P. 1990. *Pseudotsuga menziesii* (Mirb.) Franco Douglas-fir. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1 conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 1119–1133.

Hessburg, P.F.; Mitchell, R.G.; Filip, G.M. 1994. Historical and current roles of insects and pathogens in eastern Oregon and Washington forested landscapes. Gen. Tech. Rep. PNW-GTR-327. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 72 p.

Hessburg, P.F.; Smith, B.G.; Salter, R.B. 1999. Detecting change in forest spatial patterns from reference conditions. Ecological Applications. 9: 1232–1252.

Heusser, L. 1998. Direct correlation of millennial-scale changes in western North American vegetation and climate with changes in the California Current System over the past ~60 kyr. Paleoceanography. 13: 252–262.

Heyerdahl, E.K.; Brubaker, L.B.; Agee, J.K. 2002. Annual and decadal climate forcing of historical fire regimes in the interior Pacific Northwest, USA. The Holocene. 12: 597–604.

Heyerdahl, E.K.; McKenzie, D.; Daniels, L.D.; [et al.]. 2008. Climate drivers of regionally synchronous fires in the inland Northwest (1651–1900). International Journal of Wildland Fire. 17: 40–49.

Hoff, R.J.; Ferguson, D.E.; McDonald, G.I.; [et al.]. 2001. Strategies for managing whitebark pine in the presence of white pine blister rust. In: Whitebark pine communities: Ecology and restoration. Washington, DC: Island Press: 346–366.

Hogg, E.H.; Brandt, J.P.; Kochtubajda, B. 2002. Growth and dieback of aspen forest in northwestern Alberta, Canada, in relation to climate and insects. Canadian Journal of Forest Research. 32: 823–832.

Hogg, E.H.; Hurdle, P.A. 1995. The aspen parkland in western Canada: A dry-climate analogue for the future boreal forest? Water Air and Soil Pollution. 82: 391–400.

Holling, C.S. 1973. Resilience and stability of ecological systems. Annual Review of Ecology & Systematics. 4: 1–23.

Holsinger, L.; Keane, R.E.; Isaak, D.J.; [et al.]. 2014. Relative effects of climate change and wildfires on stream temperatures: A simulation modeling approach in a Rocky Mountain watershed. Climatic Change. 124: 191–206.

Hood, S.; Bentz, B.J.; Gibson, K.; [et al.]. 2007. Assessing post-fire Douglas-fir mortality and Douglas-fir beetle attacks in the northern Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-199. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 31 p.

Hopkins, T.A.; Larson, J.; Belote, R.T. 2013. Contrasting effects of wildfire and ecological restoration in old-growth western larch forests. Forest Science. 60: 1005–1013.

Huntley, B. 1991. How plants respond to climate change: Migration rates, individualism and the consequences for plant communities. Annals of Botany. 67: 15–22.

Ibáñez, I.; Clark, J.S.; LaDeau, S.; [et al.]. 2007. Exploiting temporal variability to understand tree recruitment response to climate change. Ecological Monographs. 77: 163–177.

Intergovernmental Panel on Climate Change [IPCC]. 2007. Climate change 2007: The physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Solomon, S.; Qin, D.; Manning, M.; [et al.], eds. Cambridge, United Kingdom; New York: Cambridge University Press. 427 p.

Ingerson, A.L. 2007. U.S. forest carbon and climate change. Washington, DC: The Wilderness Society. 19 p.

Ireland, K.B.; Moore, M.M.; Fulé, P.Z.; [et al.]. 2014. Slow lifelong growth predisposes *Populus tremuloides* trees to mortality. Oecologia. 175: 847–859.

Iverson, L.R.; McKenzie, D. 2013. Tree-species range shifts in a changing climate: Detecting, modeling, assisting. Landscape Ecology. 28: 879–889.

Iverson, L.R.; Prasad, A.M. 2002. Potential redistribution of tree species habitat under five climate change scenarios in the eastern US. Forest Ecology and Management. 155: 205–222.

Jackson, M.; Gannon, A.; Kearns, H.; [et al.]. 2010. Current status of limber pine in Montana. Numbered report 10-06. Missoula, MT: U.S. Department of Agriculture, Forest Service, Northern Region, Forest Health and Protection.

Jain, T.B.; Graham, R.T. 2005. Restoring dry and moist forests of the inland northwestern U.S. In: Stranhauf, J.A.; Madsen, P., eds. Restoration of boreal and temperate forests. New York: CRC Press: 463–480.

Janowiak, M.K.; Swanston, C.W.; Nagel, L.M.; [et al.]. 2014. A practical approach for translating climate change adaptation principles into forest management actions. Journal of Forestry. 112: 424–433.

Jenkins, M.J.; Hebertson, E.; Page, W.; [et al.]. 2008. Bark beetles, fuels, fires and implications for forest management in the Intermountain West. Forest Ecology and Management. 254: 16–34.

Johnstone, J.F.; Chapin, F.S. 2003. Non-equilibrium succession dynamics indicate continued northern migration of lodgepole pine. Global Change Biology. 9: 1401–1409.

Jones, H.G. 2013. Plants and microclimate: A quantitative approach to environmental plant physiology. New York: Cambridge University Press.

Joyce, L.A. 1995. Productivity of America's forests and climate change. Gen. Tech. Rep. RMRS-GTR-271. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 70 p.

Joyce, L.A.; Birdsey, R.A. 2000. The impact of climate change on America's forests: A technical document supporting the 2000 USDA Forest Service RPA Assessment. Gen. Tech. Rep. RMRS-GTR-59. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 133 p.

Joyce, L.A.; Blate, G.M.; Littell, J.S.; [et al.]. 2008. National forests. In: Julius, S.H.; West, W., eds. Preliminary review of adaptation options for climate-sensitive ecosystmes and resources. Chapter 3. Washington, DC: U.S. Environmental Protection Agency: 3-1–3-127.

Joyce, L.A.; Running, S.W.; Breshears, D.D.; [et al.]. 2014. Forests. In: Melillo, J.M.; Richmond, T.C.; Yohe, G.W., eds. Climate change impacts in the United States: Third national climate assessment. Chapter 7. Washington, DC: U.S. Global Change Research Program: 175–194.

Jump, A.S.; Peñuelas, J. 2005. Running to stand still: Adaptation and the response of plants to rapid climate change. Ecology Letters. 8: 1010–1020.

Kappelle, M.; Van Vuuren, M.M.I.; Baas, P. 1999. Effects of climate change on biodiversity: A review and identification of key research issues. Biodiversity & Conservation. 8: 1383–1397.

Kasischke, E.S.; French, N.H.F.; O'Neill, K.P.; [et al.]. 2000a. Influence of fire on long-term patterns of forest succession in Alaskan boreal forests. In: Kasischke, E.S., Stocks, B.J., eds. Fire, climate change, and carbon cycling in the North American boreal forest. New York: Springer-Verlag: 214–325.

Kasischke, E.S.; O'Neill, K.P.; French, N.; [et al.]. 2000b. Controls on patterns of biomass burning in Alaskan boreal forests. In: Kasischke, E.S., Stocks, B.J., eds. Fire, climate change, and carbon cycling in the North American boreal forest. New York: Springer-Verlag: 173–196.

Keane, R.E. 1985. Predicting successional plant composition on a Douglas-fir/ninebark habitat type in western Montana. Missoula, MT: University of Montana. 165 p.

Keane, R.E. 2000. The importance of wilderness to whitebark pine research and management. In: Proceedings of the symposium: Wilderness science in a time for change. Volume 3. Wilderness as a place for scientific inquiry. Gen. Tech. Rep. RMRS-P-15-VOL-3. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 84–93.

Keane, R.E. 2001. Successional dynamics: modeling an anthropogenic threat. In: Tomback, D.; Arno, S.; Keane, R., eds. Whitebark pine communities: Ecology and restoration. Washington, DC: Island Press: 159–192.

Keane, R.E. 2012. Creating historical range of variation (HRV) time series using landscape modeling: Overview and issues. In: Wiens, J.A.; Hayward, G.D.; Stafford, H.S.; [et al.], eds. Historical environmental variation in conservation and natural resource management. Hoboken, NJ: John Wiley and Sons: 113–128.

Keane, R.E. 2013. Disturbance regimes and the historical range of variation in terrestrial ecosystems. In: Levin, Simon, A., ed. Encyclopedia of biodiversity. 2nd ed. Waltham, MA: Academic Press: 568–581.

Keane, R.E.; Agee, J.; Fulé, P.; [et al.]. 2008. Ecological effects of large fires in the United States: Benefit or catastrophe. International Journal of Wildland Fire. 17: 696–712.

Keane, R.E.; Arno, S.F.; Brown, J.K. 1990. Simulating cumulative fire effects in ponderosa pine/Douglas-fir forests. Ecology. 71: 189–203.

Keane, R.E.; Austin, M.; Dalman, R.; [et al.]. 2001. Tree mortality in gap models: Application to climate change. Climatic Change. 51: 509–540.

Keane, R.E.; Finney, M.A. 2003. The simulation of landscape fire, climate, and ecosystem dynamics. In: Veblen, T.T.; Baker, W.L.; Montenegro, G.; [et al.], eds. Fire and global change in temperate ecosystems of the western Americas. Ecological Studies Vol. 160. New York: Springer-Verlag: 32–68.

Keane, R.E.; Hessburg, P.F.; Landres, P.B.; [et al.]. 2009. A review of the use of historical range and variation (HRV) in landscape management. Forest Ecology and Management. 258: 1025–1037.

Keane, R.E.; Loehman, R.; Clark, J.; [et al.]. 2015a. Exploring interactions among multiple disturbance agents in forest landscapes: Simulating effects of fire, beetles, and disease under climate change. In: Perera, A.H.; Remmel, T.K.; Buse, L.J., eds. Modeling and mapping forest landscape patterns. New York: Springer.

Keane, R.E.; McKenzie, D.; Falk, D.A.; [et al.]. 2015b. Representing climate, disturbance, and vegetation interactions in landscape models. Ecological Modelling. 309–310: 33–47.

Keane, R.E.; Morgan, P.; Menakis, J.P. 1994. Landscape assessment of the decline of whitebark pine (*Pinus albicaulis*) in the Bob Marshall Wilderness Complex, Montana, USA. Northwest Science. 68: 213–229.

Keane, R.E.; Parsons, R. 2010. Restoring whitebark pine forests of the northern Rocky Mountains, USA. Ecological Restoration. 28: 56–70.

Keane, R.E.; Ryan, K.C.; Running, S.W. 1996. Simulating effects of fire on northern Rocky Mountain landscapes with the ecological process model Fire-BGC. Tree Physiology. 16: 319–331.

Keane, R.E.; Tomback, D.F.; Aubry, C.A.; [et al.]. 2012. A range-wide restoration strategy for whitebark pine forests. Gen. Tech. Rep. RMRS-GTR-279. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 108 p.

Keane, R.E.; Veblen, T.; Ryan, K.C.; [et al.]. 2002. The cascading effects of fire exclusion in the Rocky Mountains. In: Baron, J., ed. Rocky Mountain futures: An ecological perspective. Washington, DC: Island Press: 133–153.

Kemp, K. 2015. Resilience and regeneration after wildfire in dry mixed-conifer forests of the U.S. northern Rockies. http://figshare.com/articles/Kemp_NRFSNPresentation_PDF_pdf/1333592 [Accessed July 05, 2016].

Kennedy, H.E. 1990. *Fraxinus pennsylvanica* Marsh. green ash. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 2 hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 1234–1255.

Kern, J.S. 1995. Geographic patterns of soils water-holding capacity in the contiguous United States. Soil Science Society of America Journal. 59: 1126–1133.

Kern, J.S. 2000. Erratum for Geographic patterns of soils water-holding capacity in the contiguous United States. Soil Science Society of America Journal. 64: 382.

Kirilenko, A.P.; Sedjo, R.A. 2007. Climate change impacts on forestry. Proceedings of the National Academy of Sciences. 104: 19697–19702.

Klopfenstein, N.B.; Kim, M.; Hanna, J.W.; [et al.]. 2009. Approaches to predicting potential impacts of climate change on forest disease: An example with Armillaria root disease. Res. Pap. RMRS-RP-76. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 10 p.

Knapp, A.K.; Smith, W.K. 1982. Factors influencing understory seedling establishment of Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) in southeast Wyoming. Canadian Journal of Botany. 60: 2753–2761.

Knutti, R.; Sedlacek, J. 2013. Robustness and uncertainties in the new CMIP5 climate model projections. Nature Climate Change. 3: 369–373.

Koteen, L. 1999. Climate change, whitebark pine, and grizzly bears in the greater Yellowstone ecosystem. In: Schneider, S.H.; Root, T.L., eds. Wildlife responses to climate change. Washington, DC: Island Press: 343–364.

Krawchuk, M.A.; Moritz, M.A.; Parisien, M.-A.; [et al.]. 2009. Global pyrogeography: The current and future distribution of wildfire. PLoS ONE. 4: e5102.

Krist, F.J., Jr.; Ellenwood, J.R.; Woods, M.E.; [et al.]. 2014. 2013–2027 National insect and disease forest risk assessment. FHTET-14-01. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Forest Health Protection, Forest Health Technology Enterprise Team. 199 p.

Kulakowski, D.; Kaye, M.W.; Kashian, D.M. 2013. Long-term aspen cover change in the western US. Forest Ecology and Management. 299: 52–59.

Kurz, W.A.; Dymond, C.C.; Stinson, G.; [et al.]. 2008. Mountain pine beetle and forest carbon feedback to climate change. Nature. 452: 987–990.

LaDeau, S.L.; Clark, J.S. 2001. Rising CO_2 levels and the fecundity of forest trees. Science. 292: 95–98.

Lambrecht, S.C.; Loik, M.E.; Inouye, D.W.; [et al.]. 2007. Reproductive and physiological responses to simulated climate warming for four subalpine species. New Phytologist. 173: 121–134.

Landhäusser, S.M.; Deshaies, D.; Lieffers, V.J. 2010. Disturbance facilitates rapid range expansion of aspen into higher elevations of the Rocky Mountains under a warming climate. Journal of Biogeography. 37: 68–76.

Landres, P.B.; Morgan, P.; Swanson, F.J. 1999. Overview and use of natural variability concepts in managing ecological systems. Ecological Applications. 9: 1179–1188.

Langor, D.W. 2007. Status of the limber pine (*Pinus flexilis*) in Alberta. Alberta Wildlife Status Report No. 62. Peters, S.; Gutsell, R.; Sharp, N.; Matthais, L., series eds. Edmonton, Alberta, Canada: Alberta Sustainable Resource Development and Alberta Conservation Association. 17 p.

Lankau, R.A.; Zhu, K.; Ordonez, A. 2015. Mycorrhizal strategies of tree species correlate with trailing range edge responses to current and past climate change. Ecology. 96: 1451–1458.

Lanner, R.M. 1980. Avian seed dispersal as a factor in the ecology and evolution of limber and whitebark pines. In: Sixth North American Forest Biology Workshop. Alberta, Canada: University of Alberta: 15–47.

Lanner, R.M.; Vander Wall, S.B. 1980. Dispersal of limber pine seed by Clark's nutcracker. Journal of Forestry. 78: 637–639.

Laughlin, D.C.; Bakker, J.D.; Stoddard, M.T.; [et al.]. 2004. Toward reference conditions: Wildfire effects on flora in an old growth ponderosa pine forest. Forest Ecology and Management. 199: 137–152.

Lawler, J.J.; White, D.; Neilson, R.P.; [et al.]. 2006. Predicting climate-induced range shifts: Model differences and model reliability. Global Change Biology. 12: 1568–1584.

Ledig, F.T; Kitzmiller, J.H. 1992. Genetic strategies for reforestation in the face of global climate change. Forest Ecology and Management. 50: 153–169.

Leirfallom, S.; Keane, R.E.; Tomback, D.; [et al.]. 2015. The effects of seed source mortality on whitebark pine (*Pinus albicaulis*) regeneration dynamics after wildfire. Canadian Journal Forest Research. 45: 1597–1606.

Lenihan, J.M.; Daly, C.; Bachelet, D.; [et al.]. 1998. Simulating broad scale fire severity in a dynamic global vegetation model. Northwest Science. 72: 91–103.

Lenihan, J.M.; Drapek, R.; Bachelet, D.; [et al.]. 2003. Climate change effects on vegetation distribution, carbon, and fire in California. Ecological Applications. 13: 1667–1681.

Lenoir, J.; Gégout, J.C.; Marquet, P.A.; [et al.]. 2008. A significant upward shift in Plant species optimum elevation during the 20th century. Science. 320: 1768–1771.

Lesica, P. 1989. The vegetation and condition of upland hardwood forests in eastern Montana. Proceedings of the Montana Academy of Sciences. 49: 45–62.

Lesica, P. 2001. Recruitment of *Fraxinus pennsylvanica* (Oleaceae) in eastern Montana woodlands. Madrono. 48: 286–292

Lesica, P. 2003. Effects of wildfire on recruitment of *Fraxinus pennsylvanica* in eastern Montana woodlands. American Midland Naturalist. 149: 258–267.

Lesica, P. 2009. Can regeneration of green ash (*Fraxinus pennsylvanica*) be restored in declining woodlands in eastern Montana? Rangeland Ecology and Management. 62: 564–571.

Lesica, P.; Marlow, C. 2013. Green ash woodlands: A review. Montana State University Extension Research Bulletin No. 4601. Bozeman, MT: Montana State University Extension. 20 p.

Liang, Y.; He, H.S.; Wang, W.J.; [et al.]. 2015. The site-scale processes affect species distribution predictions of forest landscape models. Ecological Modelling. 300: 89–101.

Lin, D.; Xia, J.; Wan, S. 2010. Climate warming and biomass accumulation of terrestrial plants: A meta-analysis. New Phytologist. 188: 187–198.

Littell, J.S.; McKenzie, D.; Kerns, B.K.; [et al.]. 2011. Managing uncertainty in climate-driven ecological models to inform adaptation to climate change. Ecosphere. 2: 102–111.

Littell, J.S.; Oneil, E.E.; McKenzie, D.; [et al.]. 2009. Forest ecosystems, disturbance, and climatic change in Washington State, USA. In: Elsner, M.M.; Littell, J.S.; Binder L.W., eds. The Washington climate change impacts assessment: evaluating Washington's future in a changing climate. Seattle, WA: University of Washington, College of the Environment, Climate Impacts Group: 255–284.

Littell, J.S.; Oneil, E.E.; McKenzie, D.; [et al.]. 2010. Forest ecosystems, disturbance, and climatic change in Washington State, USA. Climatic Change. 102: 129–158.

Little, R.L. 1992. Subalpine tree regeneration following fire: Effects of climate and other factors. Thesis. Seattle, WA: University of Washington.

Little, R.L.; Peterson, D.L. 1991. Effects of climate on regeneration of subalpine forests after wildfire. Northwest Environmental Journal. 7: 355–357.

Little, R.L.; Peterson, D.L.; Conquest, L.L. 1994. Regeneration of subalpine fir (*Abies lasiocarpa*) following fire: Effects of climate and other factors. Canadian Journal of Forest Research. 24: 934–944.

Liu, S.G.; Bond-Lamberty, B.; Hicke, J.A.; [et al.]. 2011. Simulating the impacts of disturbances on forest carbon cycling in North America: Processes, data, models, and challenges. Journal of Geophysical Research-Biogeosciences. 116.

Loehman, R.A.; Clark, J.A.; Keane, R.E. 2011a. Modeling effects of climate change and fire management on western white pine (*Pinus monticola)* in the Northern Rocky Mountains, USA. Forests. 2: 832–860.

Loehman, R.A.; Corrow, A.; Keane, R.E. 2011b. Modeling climate changes and wildfire interactions: Effects on whitebark pine (*Pinus albicaulis*) and implications for restoration, Glacier National Park, Montana, USA. In: Keane, Robert E.; Tomback, Diana F.; Murray, Michael P.; [et al.], eds. 2011. The future of high-elevation, five-needle white pines in Western North America: Proceedings of the high five symposium; 2010 June 28–3; Missoula, MT. Proceedings RMRS-P-63. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 176–188.

Loehman, R.A.; Reinhardt, E.; Riley, K.L. 2014. Wildland fire emissions, carbon, and climate: Seeing the forest and the trees—A cross-scale assessment of wildfire and carbon dynamics in fire-prone, forested ecosystems. Forest Ecology and Management. 317: 9–19.

Logan, J.A.; Powell, J.A. 2001. Ghost forests, global warming, and the mountain pine beetle (*Coleoptera: Scolytidae*). American Entomologist. 47: 160–173.

Logan, J.A.; Régnière, J.; Powell, J.A. 2003. Assessing the impacts of global warming on forest pest dynamics. Frontiers in Ecology and the Environment. 1: 130–137.

Lorenz, T.J.; Aubry, C.; Shoal, R. 2008. A review of the literature on seed fate in whitebark pine and the life history traits of Clark's Nutcracker and pine squirrels. Gen. Tech. Rep. PNW-GTR-742. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 62 p.

Lorenz, T.J.; Sullivan, K.A. 2009. Seasonal differences in space use by Clark's nutcrackers in the Cascade Range. The Condor. 11: 326–340.

Lotan, J.E.; Brown, J.K.; Neuenschwander, L.F. 1984. Role of fire in lodgepole pine forests. In: Lodgepole pine: The species and its management; 1984 May 8–10. Spokane, WA: Washington State University: 133–152.

Lotan, J.E.; Critchfield, W.B. 1990. *Pinus contorta* Dougl. ex. Loud. lodgepole pine. In: Burns, R.M.; Honkala, B.H., tech. eds. Silvics of North America. Volume 1 conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 648–666.

Lotan, J.E.; Perry, D.A. 1983. Ecology and regeneration of lodgepole pine. Agric. Handb. 606. Washington, DC: U.S. Department of Agriculture, Forest Service. 51 p.

Luce, C.H.; Abatzoglou, J.T.; Holden, Z.A. 2013. The missing mountain water: Slower westerlies decrease orographic enhancement in the Pacific Northwest USA. Science. 342: 1360–1364.

Luckman, B.H.; Jozsa, L.A.; Murphy, P.J. 1984. Living seven-hundred-year-old *Picea engelmannii* and *Pinus albicaulis* in the Canadian Rockies. Arctic & Alpine Research. 16: 419–422.

Mahalovich, M.F; Hipkins, V.D. 2011. Molecular genetic variation in whitebark pine (*Pinus albicaulis* Engelm.) in the Inland West. In: Keane, R.E.; Tomback, D.F.; Murray, M.P.; [et al.], eds. The future of high-elevation, five-needle white pines in Western North America: Proceedings of the high five symposium; 2010 June 28–30; Missoula, MT. Proceedings RMRS-P-63. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 124–139.

Mahalovich, M.F.; Burr, K.E.; Foushee, D.L. 2006. Whitebark pine germination, rust resistance and cold hardiness among seed sources in the Inland Northwest: Planting strategies for restoration. In: National proceedings: Forest and Conservation Nursery Association. 2005 July 18–20. Proc. RMRS-P-43. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 91–101.

Malanson, G.P.; Butler, D.R. 1991. Floristic variation among gravel bars in a subalpine river, Montana, U.S.A. Arctic and Alpine Research. 23: 273–278.

Malcolm, J.R.; Markham, A.; Neilson, R.P.; [et al.]. 2002. Estimated migration rates under scenarios of global climate change. Journal of Biogeography. 28: 835–849.

Marchetti, S.B.; Worrall, J.J.; Eager, T. 2011. Secondary insects and diseases contribute to sudden aspen decline in southwestern Colorado, USA. Canadian Journal of Forestry Research. 41: 2315–2325.

Marcoux, H.M.; Daniels, L.D.; Gergel, S.E.; [et al.]. 2015. Differentiating mixed- and high-severity fire regimes in mixed-conifer forests of the Canadian Cordillera. Forest Ecology and Management. 341: 45–58.

Marlon, J.R.; Bartlein, P.J.; Walsh, M.K.; [et al.]. 2009. Wildfire responses to abrupt climate change in North America. Proceedings of the National Academy of Sciences of the United States of America. 106: 2519–2524.

McCaughey, W.; Tomback, D.F. 2001. The natural regeneration process of whitebark pine. In: Tomback, D.F.; Arno, S.A.; Keane, R.E., eds. Whitebark pine communities: Ecology and restoration. Washington, DC: Island Press: 105–122.

McDermid, G.J.; Smith, I.U. 2008. Mapping the distribution of whitebark pine (*Pinus albicaulis*) in Waterton Lakes National Park using logistic regression and classification tree analysis. Canadian Journal of Remote Sensing. 34: 356–366.

McKenney, D.W.; Pedlar, J.H.; Lawrence, K.; [et al.]. 2007. Potential impacts of climate change on the distribution of North American trees. BioScience. 57: 939–948.

McKenzie, D.; Miller, C.; Falk, D.A., eds. 2011. The landscape ecology of fire. Dordrecht, The Netherlands: Springer Ltd. 312 p.

McKenzie, D.; Peterson, D.L.; Littell, J.S. 2008. Global warming and stress complexes in forests of western North America. In: Bytnerowicz, A.; Arbaugh, M.J.; Riebau, A.R.; [et al.], eds. Developments in environmental science. Volume 8. Wild land fires and air pollution. The Netherlands: Elsevier: 317–337.

McKenzie, D.; Shankar, U.; Keane, R.E.; [et al.]. 2014. Smoke consequences of new wildfire regimes driven by climate change. Earth's Future. 2(2): 35–59.

McKinney, S.T.; Fiedler, C.E.; Tomback, D.F. 2009. Invasive pathogen threatens bird-pine mutualism: Implications for sustaining a high-elevation ecosystem. Ecological Applications. 19: 597–607.

McNulty, S.G.; Vose, J.M.; Swank, W.T.; [et al.]. 1994. Regional-scale forest ecosystem modeling: Database development, model predictions and validation using Geographic Information System. Climate Research. 4: 223–231.

Means, J.E. 1990. *Tsuga mertensiana* (Bong.) Carr. mountain hemlock. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America. Volume 1 conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 1318–1332.

Merigliano, M.F. 2005. Cottonwood understory zonation and its relation to floodplain stratigraphy. Wetlands. 25: 356–374.

Merritt, D.M.; Scott, M.L.; Poff, N.L.; [et al.]. 2010. Theory, methods and tools for determining environmental flows for riparian vegetation: Riparian vegetation-flow response guilds. Freshwater Biology. 55: 206–225.

Merriam, G.; Henein, K.; Stuart-Smith, K. 1992. Landscape dynamics models. In: Turner, M.G.; Gardner, R.H., eds. Quantitative methods in landscape ecology: The analysis and interpretation of landscape heterogeneity. New York: Springer-Verlag: 399–416.

Millar, C.I. 1997. Comments on historical variation and desired future conditions as tools for terrestrial landscape analysis. In: Sixth biennial watershed management conference. Water Resources Center Report No. 9. Davis, CA: University of California: 105–131.

Millar, C.I.; Woolfenden, W.B. 1999. The role of climate change in interpreting historical variability. Ecological Applications. 9: 1207–1216.

Millar, C.I.; Swanston, C.W.; Peterson, D.L. 2014. Adapting to climate change. In: Peterson, D.L.; Vose, J.M.; Patel-Weynand, T., eds. Climate change and United States forests. Dordrecht, The Netherlands: Springer: 183–222.

Millar, C.I.; Stephenson, N.L.; Stephens, S.L. 2007a. Climate change and forests of the future: Managing in the face of uncertainty. Ecological Applications. 17: 2145–2151.

Millar, C.I.; Westfall, R.D.; Delany, D.L. 2007b. Response of high-elevation limber pine (*Pinus flexilis*) to multiyear droughts and 20th century warming, Sierra Nevada, California, USA. Canadian Journal Forest Research. 37: 2508–2520.

Miller, J.D.; Safford, H.D.; Crimmins, M.; [et al.]. 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA. Ecosystems. 12: 16–32.

Milner, K.S. 1992. Site index and height growth curves for ponderosa pine, western larch, lodgepole pine, and Douglas fir in western Montana. Western Journal of Applied Forestry. 7: 9–14.

Minore, D. 1979. Comparative autecological characteristics of northwestern tree species: A literature review. Gen. Tech. Rep. PNW-GTR-087. Portland, OR.: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 28 p.

Minore, D. 1990. *Thuja plicata* Donn ex D. Don western red cedar. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1 conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 1249–1267.

Mitton, J.B.; Grant, M.C. 1996. Genetic variation and the natural history of quaking aspen. Bioscience. 46: 25–31.

Mohatt, K.; Cripps, C.L.; Lavin, M. 2008. Ectomycorrhizal fungi of whitebark pine (a tree in peril) revealed by sporocarps and molecular analysis of mycorrhizae from treeline forests in the Greater Yellowstone Ecosystem. Botany. 86: 14–15.

Montana State Library and Natural Resources Conservation Service [NRCS]. [n.d.]. Montana relative effective annual precipitation (REAP) data access. Helena, MT: Montana State Library. http://nris.mt.gov/nrcs/reap/index.asp. [Accessed March 31, 2017].

Morales, J.M.; Mermoz, M.; Gowda, J.H.; [et al.]. 2015. A stochastic fire spread model for north Patagonia based on fire occurrence maps. Ecological Modelling. 300: 73–80.

Moran, C.J.; Corcoran, M.A. 2012. Do mountain pine beetle outbreaks change the probability of active crown fire in lodgepole pine forests? Comment. Ecology. 93: 939–950.

Moran-Palma, P.; McTague, J.P. 1997. Stand dynamics of the spruce-fir forest in east-central Arizona. Western Journal of Applied Forestry. 12: 55–61.

Morgan, P.; Aplet, G.H.; Haufler, J.B.; [et al.]. 1994a. Historical range of variability: A useful tool for evaluating ecosystem change. Journal of Sustainable Forestry. 2: 87–111.

Morgan, P.; Bunting, S.C.; Keane, R.E.; [et al.]. 1994b. Fire ecology of whitebark pine (*Pinus albicaulis*) forests in the Rocky Mountains, USA. In:. Proceedings of the international symposium subalpine stone pines and their environment: The status of our knowledge; 1992 September 5–11; St. Moritz, Switzerland. Gen. Tech. Rep. INT-GTR-309. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 136–142

Moritz, C.; Agudo, R. 2013. The future of species under climate change: Resilience or decline? Science. 341: 504–508.

Morris, W.F.; Pfister, C.A.; Tuljapurkar, S.; [et al.]. 2008. Longevity can buffer plant and animal populations against changing climatic variability. Ecology. 89: 19–25.

Mote, P.W.; Salathé, P. 2010. Future climate in the Pacific Northwest. Climatic Change. 102: 29–50.

Mote, P.W.; Hamlet, A.F.; Clark, M.P.; [et al.]. 2005. Declining mountain snowpack in western North America. Bulletin of the American Meteorological Society. 86: 39–49.

Mueggler, W.F. 1985. Vegetation associations. In: DeByle, N.V.; Winokur, R.P., eds., Aspen: Ecology and management in the western United States. Gen. Tech. Rep. GTR-RM-119. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 45–55.

Mueggler, W.F. 1988. Aspen community types of the intermountain region. Gen. Tech. Rep. GTR-INT-250. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 135 p.

Murray, M.P.; Bunting, S.C.; Morgan, P. 1997. Subalpine ecosystems: The roles of whitebark pine and fire. In: Fire effects on rare and endangered species and habitats conference; 1995 November 13–16; Coeur d'Alene, ID. International Association of Wildland Fire: 295–299.

Murray, M.P.; Bunting, S.C.; Morgan, P. 1998. Fire history of an isolated subalpine mountain range of the intermountain region, United States. Journal of Biogeography. 25: 1071–1080.

Nakićenović, N.; Davidson, O.; Davis, G.; [et al.]. 2000. Special report on emissions scenarios. A special report of Working Group III of the Intergovernmental Panel on Climate Change. Cambridge and New York: Cambridge University Press. 599 p.

Neilson, R.P. 1995. A model for predicting continental-scale vegetation distribution and water balance. Ecological Applications. 5: 362–385.

Neilson, R.P.; Pitelka, L.F.; Solomon, A.M.; [et al.]. 2005. Forecasting regional to global plant migration in response to climate change. BioScience. 55: 749–760.

Newman, M.; Compo, G.P.; Alexander, M.A. 2003. ENSO-forced variability of the Pacific Decadal Oscillation. Journal of Climate. 16: 3853–3857.

Nigh, G. 2014. Mitigating the effects of climate change on lodgepole pine site height in British Columbia, Canada, with a transfer function. Forestry. 87: 377–387.

Nitschke, C.R.; Innes, J.L. 2008. A tree and climate assessment tool for modelling ecosystem response to climate change. Ecological Modelling. 210: 263–277.

Nonaka, E.; Spies, T.A. 2005. Historical range of variability in landscape structure: A simulation study in Oregon, USA. Ecological Applications. 15: 1727–1746.

Norum, R.A. 1974. Fire intensity fuel reduction relationships associated with understory burning in larch/Douglas-fir stands. In: Proceedings of a symposium; 1974 October 8-10; Missoula, MT. Tall Timbers Fire Ecology Conference Number 14. Tallahassee, FL: Tall Timbers Research Station: 559–572.

Notaro, M. 2008. Response of the mean global vegetation distribution to interannual climate variability. Climate Dynamics. 30: 845–854.

Notaro, M.; Mauss, A.; Williams, J.W. 2012. Projected vegetation changes for the American Southwest: Combined dynamic modeling and bioclimatic-envelope approach. Ecological Applications. 22: 1365–1388.

Notaro, M.; Vavrus, S.; Liu, Z. 2007. Global vegetation and climate change due to future increases in CO_2 as projected by a fully coupled model with dynamic vegetation. Journal of Climate. 20: 70–88.

Nyland, R.D. 1998. Patterns of lodgepole pine regeneration following the 1988 Yellowstone fires. Forest Ecology and Management. 111: 23–33.

Oliver, T.; Roy, D.B.; Hill, J.K.; [et al.]. 2010. Heterogeneous landscapes promote population stability. Ecology Letters. 13: 473–484.

Oliver, W.W.; Ryker, R.A. 1990. *Pinus ponderosa* Dougl. ex Laws ponderosa pine. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1 conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 348–353.

Owens, J.N. 2008. The reproductive biology of western larch. FGC Extension Note 08. Moscow, ID: University of Idaho, Inland Empire Tree Improvement Cooperative; British Columbia: Forest Genetics Council. 80 p.

Owens, J.N.; Molder, M. 1979. Sexual reproduction of *Larix occidentalis*. Canadian Journal of Botany. 57: 2673–2690.

Pacala, S.; Birdsey, R.A.; Bridgham, S.D.; [et al.]. 2007. The North American carbon budget past and present. In: King, A.W.; Dilling, L.; Zimmerman, G.P.; [et al.], eds. The first state of the carbon cycle report (SOCCR): The North American carbon budget and implications for the global carbon cycle. Asheville, NC: National Oceanic and Atmospheric Administration, National Climatic Data Center: 29–36, 167–170.

Packee, E.C. 1990. *Tsuga heterophylla* (Raf.) Sarg. western hemlock. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1 conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 1295–1344.

Pan, Y.; Birdsey, R.A.; Fang, J.; [et al.]. 2011. A large and persistent carbon sink in the world's forests. Science. 333: 988–993.

Parmesan, C. 2006. Ecological and evolutionary responses to recent climate change. Annual Review of Ecology, Evolution, and Systematics. 37: 637–669.

Parton, W.J.; Schimel, D.S.; Cole, C.V.; [et al.]. 1987. Analysis of factors controlling soil organic levels of grasslands in the Great Plains. Soil Science Society of America Journal. 51: 1173–1179.

Perala, D.A. 1983. Modeling aspen and red pine shoot growth to daily weather variations. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 11 p.

Perala, D.A. 1990. *Populus tremuloides* Michx. quaking aspen. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 2 hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 348–353.

Perry, G.L.W.; Millington, J.D.A. 2008. Spatial modelling of succession-disturbance dynamics in forest ecosystems: Concepts and examples. Perspectives in Plant Ecology, Evolution and Systematics. 9: 191–210.

Peters, R.L.; Lovejoy, T.E. 1992. Global warming and biological diversity. New Haven, CT: Yale University Press.

Peterman, W.; Bachelet, D.; Ferschweiler, K.; [et al.]. 2014. Soil depth affects simulated carbon and water in the MC2 dynamic global vegetation model. Ecological Modelling. 294: 84–93.

Peterson, D.W.; Peterson, D.L. 1994. Effects of climate on radial growth of subalpine conifers in the North Cascade Mountains. Canadian Journal of Forest Research. 24: 1921–1932.

Petrie, M.D.; Wildeman, A.M.; Bradford, J.B.; [et al.]. 2016. A review of precipitation and temperature control on seedling emergence and establishment for ponderosa and lodgepole pine forest regeneration. Forest Ecology and Management. 361: 328–338.

Pfister, R.D.; Kovalchik, B.L.; Arno, S.F.; [et al.]. 1977. Forest habitat types of Montana. Gen. Tech. Rep. INT-GTR-34. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 174 p.

Potter, K.M.; Hipkins, V.D.; Mahalovich, M.F.; Means, R.E. 2015. Nuclear genetic variation across the range of ponderosa pine (*Pinus ponderosa*): Phylogeographic, taxonomic and conservation implications. Tree Genetics & Genomes. 11: 1–23.

Prentice, I.C.; Bartlein, P.J.; Webb, T., III. 1991. Vegetation and climate change in eastern North America since the last glacial maximum. Ecology. 72: 2038–2056.

Purves, D.; Pacala, S. 2008. Predictive models of forest dynamics. Science. 320: 1452–1453.

Rehfeldt, G.E. 1978. Genetic differentiation of Douglas-fir populations from the northern Rocky Mountains. Ecology. 1264–1270.

Rehfeldt, G.E. 1982. Differentiation of *Larix occidentalis* populations from the Northern Rocky Mountains. Silvae Genetica. 31: 13–19.

Rehfeldt, G.E. 1989. Ecological adaptations in Douglas-fir (*Pseudotsuga menziesii* var. *glauca*): A synthesis. Forest Ecology and Management. 28: 203–215.

Rehfeldt, G.E. 1994. Evolutionary genetics, the biological species, and the ecology of the cedar-hemlock forests. In: Baumgartner, D.M.; Lotan, J.E.; Tonn, J.R., eds. Interior cedar-hemlock-white pine forests: Ecology and management; 1993 March 2–4. Spokane, WA: Washington State University Cooperative Extension: 91–100.

Rehfeldt, G.E. 1995a. Domestication and conservation of genetic variability in western larch. In: Proceedings of an international symposium: Ecology and management of *Larix* forests: A look ahead; 1992 October 5–9; Whitefish, MT. Gen. Tech. Rep. GTR-INT-319. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 91–96.

Rehfeldt, G.E. 1995b. Genetic variation, climate models and the ecological genetics of *Larix occidentalis*. Forest Ecology and Management. 78: 21–37.

Rehfeldt, G.E.; Jaquish, B.C. 2010. Ecological impacts and management strategies for western larch in the face of climate-change. Mitigation and Adaptation Strategies for Global Change. 15: 283–306.

Rehfeldt, G.E.; Leites, P.L.; St Clair, J.B.; [et al.]. 2014. Comparative genetic responses to climate in the varieties of *Pinus ponderosa* and *Pseudotsuga menziesii*: Clines in growth potential. Forest Ecology and Management. 324: 138–146.

Rehfeldt, G.E.; Ying, C.C.; Spittlehouse, D.L.; [et al.]. 1999. Genetic responses to climate in *Pinus contorta*: Niche breadth, climate change, and reforestation. Ecological Monographs. 69: 375–407.

Richardson, B.A.; Brunsfeld, S.J.; Klopfenstein, N.B. 2002. DNA from bird-dispersed seed and wind-disseminated pollen provides insights into postglacial colonization and population genetic structure of whitebark pine (*Pinus albicaulis*). Molecular Ecology. 11: 215–227.

Riggs, R.A.; Keane, R.E.; Cimon, N.; [et al.] 2015. Biomass and fire dynamics in a temperate forest-grassland mosaic: Integrating multi-species herbivory, climate, and fire with the FireBGCv2/GrazeBGC system. Ecological Modelling. 296: 57–78.

Rochefort, R.M.; Little, R.L.; Woodward, A.; [et al.]. 1994. Changes in sub-alpine tree distribution in western North America: A review of climatic and other causal factors. The Holocene. 4: 89–100.

Roe, A.L.; Amman, G.D. 1970. The mountain pine beetle in lodgepole pine forests. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 22 p.

Roe, G.H.; Baker, M.B. 2007. Why is climate sensitivity so unpredictable. Science. 318: 629–632.

Rollins, M.G. 2009. LANDFIRE: A nationally consistent vegetation, wildland fire, and fuel assessment. International Journal of Wildland Fire. 18: 235–249.

Rogers, P. 2002. Using forest health monitoring to assess aspen forest cover change in the southern Rockies ecoregion. Forest Ecology and Management. 155: 223–236.

Rogers, P.C.; Eisenberg, C.; St. Clair, S.B. 2013. Resilience in quaking aspen: Recent advances and future needs. Forest Ecology and Management. 299: 1–5.

Rogers, P.C.; Shepperd, W.D.; Bartos, D.L. 2007. Aspen in the Sierra Nevada: regional conservation of a continental species. Natural Areas Journal. 27: 183–193.

Roe, A.L.; Amman, G.D. 1970. The mountain pine beetle in lodgepole pine forests. Res. Pap. INT-RP-71. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 22 p.

Roe, G.H.; Baker, M.B. 2007. Why is climate sensitivity so unpredictable. Science. 318: 629–632.

Romme, W.H. 2005. The importance of multiscale spatial heterogeneity in wildland fire management and research. In: Lovett, G.M.; Jones, C.G.; Turner, M.G.; [et al.], eds. Ecosystem function in heterogeneous landscapes. New York: Springer: 253–266.

Romme, W.H.; Despain, D G. 1989. Historical perspective on the Yellowstone fires of 1988. BioScience. 39: 695–699.

Romme, W.H.; Turner, M.G. 1991. Implications of global climate change for biogeographic patterns in the greater Yellowstone ecosystem. Conservation Biology. 5: 373–386.

Romme, W.H.; Turner, M.G.; Gardner, R.H.; [et al.]. 1997. A rare episode of sexual reproduction in aspen (*Populus tremuloides* Michx.) following the 1988 Yellowstone fires. Natural Areas Journal. 17: 17–25.

Rood, S.B.; Braatne, J.H.; Hughes, F.M. 2003. Ecophysiology of riparian cottonwoods: Stream flow dependency, water relations and restoration. Tree Physiology. 23: 1113–1124.

Rose, N.A.; Burton, P.J. 2009. Using bioclimatic envelopes to identify temporal corridors in support of conservation planning in a changing climate. Forest Ecology and Management. 258: S64–S74.

Rustad, L.; Huntington, T.; Boone, R. 2000. Controls on soil respiration: Implications for climate change. Biogeochemistry. 48: 1–6.

Ryan, K.C. 1998. Analysis of the relative value of morphological variables in predicting fire-caused tree mortality. In: Viegas, D.X., ed. Proceedings from the III international conference on forest fire research and 14th conference on fire and forest meteorology. Coimbra, Portugal: Associacao para o Desenvolvimento da Aerodinamica Industrial: 1511–1526.

Ryan, K.C.; Reinhardt, E.D. 1988. Predicting postfire mortality of seven western conifers. Canadian Journal of Forest Research. 18: 1291–1297.

Ryan, M.G.; Gower, S.T.; Hubbard, R.M.; [et al.]. 1995. Woody tissue maintenance respiration of four conifers in contrasting climates. Oecologia. 101: 133–140.

Ryker, R.A.; Losensky, J. 1983. Ponderosa pine and Rocky Mountain Douglas-fir. Silvicultural systems for the major forest types of the United States. Agric. Handb. 445. Washington, DC: U.S. Department of Agriculture: 53–55.

Sala, A.; Peters, G.D.; McIntyre, L.R.; [et al.]. 2005. Physiological responses of ponderosa pine in western Montana to thinning, prescribed fire and burning season. Tree Physiology. 25: 339–348.

Safranyik, L.; Carroll, A.; Régnière, J.; [et al.]. 2010. Potential for range expansion of mountain pine beetle into the boreal forest of North America. The Canadian Entomologist. 142: 415–442.

Schauer, A.J.; Wade, B.K.; Sowell, J.B. 1998. Persistence of subalpine forest-meadow ecotones in the Gunnison Basin, Colorado. Great Basin Naturalist. 58: 273–281.

Schmidt, W.C. 1969. Seedbed treatments influence seedling development in western larch forests. Res. Note INT-RN-93. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 7 p.

Schmidt, W.C.; Fellin, D.G. 1973. Western spruce budworm damage affects form and height growth of western larch. Canadian Journal of Forest Research. 3: 17–26.

Schmidt, W.C.; Shearer, R.C. 1990. *Larix occidentalis* Nutt. western larch. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1 conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 348–353.

Schmidt, W.C.; Shearer, R.C. 1995. *Larix occidentalis*: A pioneer in the North American West. In: Schmidt, W.C.; McDonald, K.J., eds. Proceedings of an international symposium: Ecology and management of *Larix* forests: A look ahead; 1992 October 5–9; Whitefish, MT. Gen. Tech. Rep. INT-GTR-319. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 521 p.

Schmidt, W.C.; Shearer, R.C.; Roe, A.L. 1976. Ecology and silviculture of western larch forests. Tech. Bull. 1520. Washington, DC: U.S. Departmen of Agriculture, Forest Service. 99 p.

Schneider, S.H.; Semenov, S.; Patwardhan, A.; [et al.]. 2007. Assessing key vulnerabilities and the risk from climate change. In: Parry, M.L.; Canziani, O.F.; Palutikof, P.J.; [et al.], eds. Climate change 2007: Impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, UK: Cambridge University Press: 779–810.

Schoennagel, T.; Veblen, T.T.; Negron, J.F.; [et al.]. 2012. Effects of mountain pine beetle on fuels and expected fire behavior in lodgepole pine forests, Colorado, USA. PLoS ONE. 7: e30002.

Schoennagel, T.L.; Veblen, T.T.; Romme, W.H. 2004. The interaction of fire, fuels, and climate across Rocky Mountain landscapes. BioScience. 54: 651–672.

Schoettle, A.W.; Sniezko, R.A. 2007. Proactive intervention to sustain high-elevation pine ecosystems threatened by white pine blister rust. Journal of Forest Research. 12: 327–336.

Schowalter, T.; Hansen, E.; Molina, R.; [et al.]. 1997. Integrating the ecological roles of phytophagous insects, plant pathogens, and mycorrhizae in managed forests. In: Kohm, K.A.; Franklin, J.F., eds. Creating a forestry for the 21st century: The science of ecosystem management. Washington, DC: Island Press: 171–189.

Schrag, A.M.; Bunn, A.G.; Graumlich, L.J. 2007. Influence of bioclimatic variables on tree-line conifer distribution in the Greater Yellowstone Ecosystem: Implications for species of conservation concern. Journal of Biogeography. 35: 698–710.

Schumacher, S.; Reineking, B.; Sibold, J.; [et al.]. 2006. Modeling the impact of climate and vegetation on fire regimes in mountain landscapes. Landscape Ecology. 21: 539–554.

Schwandt, J.W. 2006. Whitebark pine in peril: a case for restoration. Report R1-06-28. Missoula, MT: U.S. Department of Agriculture, Forest Service, Northern Region. 20 p.

Scott, M.L.; Auble, G.T.; Friedman, J.M. 1997. Flood dependency of cottonwood establishment along the Missouri River, Montana, USA. Ecological Applications. 7: 677–690.

Seidl, R.; Spies, T.A.; Peterson, D.L.; [et al.]. 2016. Searching for resilience: Addressing the impacts of changing disturbance regimes on forest ecosystem services. Journal of Applied Ecology. 53: 120–129.

Severson, K.E.; Boldt, C.E. 1978. Cattle, wildlife, and riparian habitats in the western Dakotas. In: Shaver, J.C., ed. Management and use of northern plains rangeland; Proceedings of the regional rangeland symposium; Bismark, ND. Fargo, ND: North Dakota State University: 90–103.

Shafer, S.L.; Bartlein, P.J.; Thompson, R.S. 2001. Potential changes in the distributions of western north America tree and shrub taxa under future climate scenarios. Ecosystems. 4: 200–215.

Shafroth, P.B.; Stromberg, J.C.; Patten, D.T. 2002. Riparian vegetation response to altered disturbance and stress regimes. Ecological Applications. 12: 107–123.

Sharples, J.J. 2009. An overview of mountain meteorological effects relevant to fire behaviour and bushfire risk. International Journal of Wildland Fire. 18: 737–754.

Shearer, R.C. 1976. Early establishment of conifers following prescribed broadcast burning in western larch/Douglas-fir forests. In: Proceedings from the Tall Timbers Fire Ecology Conference No. 14. Tallahassee, FL: Tall Timbers Research Station: 481–500.

Shearer, R.C.; Schmidt, W.C. 1970. Natural regeneration in ponderosa pine forests of western Montana. Res. Pap. INT-RP-86. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 19 p.

Shepperd, W.D.; Bartos, D.L.; Mata, S.A. 2001. Above- and below-ground effects of aspen clonal regeneration and succession to conifers. Canadian Journal of Forest Research. 31: 739–745.

Shepperd, W.D.; Smith, F.W. 1993. The role of near-surface lateral roots in the life cycle of aspen in the central Rocky Mountains. Forest Ecology and Management. 61: 157–170.

Shinneman, D.J.; Baker, W.L.; Rogers, P.C.; [et al.]. 2013. Fire regimes of quaking aspen in the Mountain West. Forest Ecology and Management. 299: 22–34.

Sklar, F.H.; Costanza, R. 1991. The devlopment of dynamic spatial models for landscape ecology: A review and prognosis. In: Turner, M.G.; Gardner, R.H., eds. Quantitative methods in landscape ecology. New York: Springer-Verlag. 82: 239–288.

Smith, C.M.; Langor, D.W.; Myrholm, C.; [et al.]. 2013. Changes in white pine blister rust infection and mortality in limber pine over time. Canadian Journal of Forest Research. 43: 919–928.

Smith, M.D. 2011. The ecological role of climate extremes: Current understanding and future prospects. Journal of Ecology. 99: 651–655.

Smithwick, E.A.H.; Ryan, M.G.; Kashian, D.M.; [et al.]. 2009. Modeling the effects of fire and climate change on carbon and nitrogen storage in lodgepole pine (*Pinus contorta*) stands. Global Change Biology. 15: 535–548.

Sohngen, B.; Mendelsohn, R.; Sedjo, R. 2001. A global model of climate change impacts on timber markets. Journal of Agricultural and Resource Economics. 26: 326–343.

Soil Conservation Service. 1991. State soil geographic (STATSGO) data base: Data use information. Miscell. Publ. Num. 1492. Fort Worth, TX: U.S. Department of Agriculture, Natural Resources Conservation Service, National Soil Survey Center. 123 p.

Solomon, A.M.; West, D.C. 1993. Predicting afforestation success during climatic warming at the northern limit of forests. In: Alden, J.; Mastrantonio, J.L.; Odum, S., eds. Forest development in cold climates. New York: Springer: 167–188.

Soulé, P.; Knapp, P. 2013. Radial growth rates of two co-occurring coniferous trees in the Northern Rockies during the past century. Journal of Arid Environments. 94: 87–95.

Spittlehouse, D.L.; Stewart, R.B. 2004. Adaptation to climate change in forest management. British Columbia Journal of Ecosystems and Management. 4: 1–11.

St. Clair, J.B.; Howe, G.T. 2007. Genetic maladaptation of coastal Douglas-fir seedlings to future climates. Global Change Biology. 13: 1441–1454.

St. Clair, J.B.; Howe, G.T. 2011. Strategies for conserving forest genetic resources in the face of climate change. Turkish Journal of Botany. 35: 403–409.

Stainforth, D.A.; Aina, T.; Christensen, C.; [et al.]. 2005. Uncertainty in predictions of the climate response to rising levels of greenhouse gases. Nature. 433: 403–406.

Steele, R. 1990. *Pinus flexilis* James limber pine. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America: Volume 1 conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture: 348–353.

Steele, R.; Geier-Hayers, K. 1989. The Douglas-fir/ninebark habitat type in central Idaho: Succession and management. Gen. Tech. Rep. INT-GTR-252. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 65 p.

Steele, R.; Arno, S.F; Geier-Hayes, K. 1986. Wildfire patterns change in central Idaho's ponderosa pine-Douglas-fir forest. Western Journal of Applied Forestry. 1: 16–19.

Steele, R.; Cooper, S.V.; Ondov, D.M.; [et al.]. 1983. Forest habitat types of eastern Idaho-western Wyoming. Gen. Tech. Rep. INT-GTR-144. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 122 p.

Stephens, S.L.; Burrows, N.; Buyantuyev, A.; [et al.]. 2014. Temperate and boreal forest mega-fires: Characteristics and challenges. Frontiers in Ecology and the Environment. 12: 115–122.

Stewart, C.; Arno, S. 1997. Lodgepole pine landscape mosaics: Defining historic landscape patterns. Thesis. Missoula, MT: University of Montana. 76 p.

Stoehr, M.U. 2000. Seed production of western larch in seed-tree systems in the southern interior of British Columbia. Forest Ecology and Management. 130: 7–15.

Stout, D.L.; Sala, A. 2003. Xylem vulnerability to cavitation in *Pseudotsuga menziesii* and *Pinus ponderosa* from contrasting habitats. Tree Physiology. 23: 43–50.

Sturrock, R.N.; Frankel, S.J.; Brown, A.V.; [et al.]. 2010. Climate change and forest diseases. Plant Pathology. 60: 133–149.

Swanston, C.; Janowiak, M., eds. 2012. Forest Adaptation Resources: Climate change tools and approaches for land managers. Gen. Tech. Rep. NRS-GTR-87. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 121 p.

Swetnam, T.W.; Allen, C.D.; Betancourt, J.L. 1999. Applied historical ecology: Using the past to manage for the future. Ecological Applications. 9: 1189–1206.

Tarrant, R.F. 1953. Soil moisture and the distribution of lodgepole and ponderosa pine: A review of the literature. Res. Pap. No. 8. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 10 p.

Taylor, J.; Sturdevant, N. 1998. Limber pine mortality on the Lewis and Clark National Forest, Montana. Forest Health Protection Report 98-5. Missoula, MT: U.S. Department of Agriculture, Forest Service, Nothern Region: 1–7.

Temperli, C.; Bugmann, H.; Elkin, C. 2013. Cross-scale interactions among bark beetles, climate change, and wind disturbances: A landscape modeling approach. Ecological Monographs. 83: 383–402.

Thuiller, W.; Albert, C.; Araújo, M.B.; [et al.]. 2008. Predicting global change impacts on plant species' distributions: Future challenges. Perspectives in Plant Ecology, Evolution and Systematics. 9: 137–152.

Tomback, D.; Arno, S.F.; Keane, R.E. 2001. Whitebark pine communities: Ecology and restoration. Washington, DC: Island Press. 440 p.

Tomback, D.F. 1977. Foraging strategies of Clark's nutcracker. The Living Bird. 16: 123–161.

Tomback, D.F. 1982. Dispersal of whitebark pine seeds by Clark's nutcracker: A mutualism hypothesis. Journal of Animal Ecology. 51: 451–467.

Tomback, D.F. 1983. Nutcrackers and pines: Coevolution or coadaptation? In: Nitecki, M.H., ed. Chicago: University of Chicago Press: 179–223.

Tomback, D.F. 1989. The broken circle: Fire, birds and whitebark pine. In: Walsh, T., ed. Wilderness and wildfire. Miscel. Publ. 50. Missoula, MT: University of Montana, School of Forestry, Montana Forest and Range Experiment Station: 14–17.

Tomback, D.F. 1998. Clark's nutcracker (*Nucifraga columbiana*). The birds of North America. 331: 1–23.

Tomback, D.F.; Achuff, P. 2010. Blister rust and western forest biodiversity: Ecology, values and outlook for white pines. Forest Pathology. 40: 186–225.

Touzel, S.J. 2013. Mapping the distribution and abundance of western larch (*Larix occidentalis* Nutt.*)* with multi-temporal satellite imagery and gradient modeling. Thesis. Missoula, MT: University of Montana.

Turner, M. 1987. Landscape heterogeneity and disturbance. Ecological Studies. Volume 64. New York: Springer Verlag. 239 p.

Turner, M.G.; Romme, W.H.; Gardner, R.H.; [et al.]. 1993. A revised concept of landscape equilibrium: Disturbance and stability on scaled landscapes. Landscape Ecology. 8: 213–227.

Turner, M.G.; Romme, W.H.; Gardner, R.H. 1999. Prefire heterogeneity, fire severity, and early postfire plant reestablishment in subalpine forests of Yellowstone National Park, Wyoming. International Journal of Wildland Fire. 9: 21–36.

Urban, D.L.; Harmon, M.E.; Halpern, C.B. 1993. Potential response of Pacific Northwestern forests to climatic change, effects of stand age and initial composition. Climatic Change. 23: 247–266.

Uresk, D.W.; Boldt, C.E. 1986. Effect of cultural treatments on regeneration of native woodlands on the Northern Great Plains. The Prairie Naturalist. 18: 193–202.

USDA Forest Service [USDA FS]. 2006. Why is genetic diversity important? Volume 1. Why we care about genetics. Placerville, CA: U.S. Department of Agriculture, Forest Service, National Forest Genetics Lab. 2 p.

U.S. Fish and Wildlife Service [USFWS]. 2011. Endangered and threatened wildlife and plants: 12-month finding on a petition to list *Pinus albicaulis* as endangered or threatened with critical habitat. Federal Register. 76 (138): 42631–42654.

van Mantgem, P.J.; Stephenson, N.L. 2007. Apparent climatically induced increase of tree mortality rates in a temperate forest. Ecology Letters. 10: 909–916.

Veblen, T.T.; Hadley, K S.; Nel, E.M.; [et al.]. 1994. Disturbance regime and disturbance interactions in a Rocky Mountain subalpine forest. Journal of Ecology. 82: 125–135.

Villalba, R.; Veblen, T.T.; Ogden, J. 1994. Climatic influences on the growth of subalpine trees in the Colorado Front Range. Ecology. 75: 1450–1462.

Wadleigh, L.; Jenkins, M.J. 1996. Fire frequency and the vegetative mosaic of a spruce-fir forest in northern Utah. Great Basin Naturalist. 56: 28–37.

Walker, B.H. 1994. Landscape to regional-scale responses of terrestrial ecosystems to global change. Ambio. 23: 67–73.

Walker, R.F.; Geisinger, D.R.; Johnson, D.W.; [et al.]. 1995. Interactive effects of atmospheric CO_2 enrichment and soil N on growth and ectomycorrhizal colonization of ponderosa pine seedlings. Forest Science. 41: 491–500.

Wang, T.; Hamann, A.; Yanchuk, A.; [et al.]. 2006. Use of response functions in selecting lodgepole pine populations for future climates. Global Change Biology. 12: 2404–2416.

Waring, R.H.; Running, S.W. 1998. Forest ecosystems: Analysis at multiple scales. 2nd ed. San Diego, CA: Academic Press, Inc. 370 p.

Warwell, M.V.; Rehfeldt, G.E.; Crookston, N.L. 2007. Modeling contemporary climate profiles of whitebark pine (*Pinus albicaulis*) and predicting responses to global warming. In: Proceedings of the conference whitebark pine: A Pacific Coast perspective; 2006 August 27–31; Ashland, OR. R6-NR-FHP-2007-01. Ashland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region: 139–142.

Watling, J.I.; Romañach, S.S.; Bucklin, D.N.; [et al.]. 2012. Do bioclimate variables improve performance of climate envelope models? Ecological Modelling. 246: 79–85.

Weed, A.S.; Bentz, B.J.; Ayres, M.P.; [et al.]. 2015. Geographically variable response of *Dendroctonus ponderosae* to winter warming in the western United States. Landscape Ecology. 30: 1075–1093.

Wellner, C.A. 1970. Fire history in the northern Rocky Mountains. In: Proceedings, role of fire in the Intermountain West. Missoula, MT: Intermountain Fire Research Council: 42–64.

West, J.M.; Julius, S.H.; Kareiva, P.; [et al.]. 2009. US natural resources and climate change: Concepts and approaches for management adaptation. Environmental Management. 44: 1001–1021.

Westerling, A.L.; Hidalgo, H.G.; Cayan, D.R.; [et al.]. 2006. Warming and earlier spring increase in western US forest wildfire activity. Science. 313: 940–943.

White, C.A.; Olmsted, C.E.; Kay, C.E. 1998. Aspen, elk, and fire in the Rocky Mountain National Parks of North America. Wildlife Society Bulletin. 26: 449–462.

Whited, D.C.; Lorang, M.S.; Harner, M.J.; [et al.]. 2007. Climate, hydrologic disturbance, and succession: Drivers of floodplain pattern. Ecology. 88: 940–953.

Whitlock, C. 1993. Postglacial vegetation and climate of Grand Teton and southern Yellowstone National Parks. Ecological Monographs. 63: 173–198.

Whitlock, C. 2004. Forests, fire and climate. Nature. 432: 28–29.

Whitlock, C.; Bartlein, P.J. 1993. Spatial variations of holocene climatic change in the Yellowstone region. Quaternary Research. 39: 231–238.

Whitlock, C.; Sarna-Wojcicki, A.M.; Bartlein, P.J.; [et al.]. 2000. Environmental history and tephrostratigraphy at Carp Lake, southwestern Columbia basin, Washington, USA. Palaeogeography, Palaeoclimatology, Palaeoecology. 155: 7–29.

Whitlock, C.; Shafer, S.L.; Marlon, J. 2003. The role of climate and vegetation change in shaping past and future fire regimes in the northwestern US and the implications for ecosystem management. Forest Ecology and Management. 178: 5–21.

Wikars, L.O.; Schimmel, J. 2001. Immediate effects of fire-severity on soil invertebrates in cut and uncut pine forests. Forest Ecology and Management. 141: 189–200.

Williams, A.P.; Allen, C.D.; Millar, C.I.; [et al.]. 2010. Forest responses to increasing aridity and warmth in the southwestern United States. Proceedings of the National Academy of Sciences. 107: 21289–21294.

Woods, A.J.; Heppner, D.; Kope, H.H.; [et al.]. 2010. Forest health and climate change: A British Columbia perspective. The Forestry Chronicle. 86: 412–422.

Woodward, A.; Silsbee, D.G.; Schreiner, E.G.; [et al.]. 1994. Influence of climate on radial growth and cone production in subalpine fir (*Abies lasiocarpa*) and mountain hemlock (*Tsuga mertensiana).* Canadian Journal of Forest Research. 24: 1133–1143.

Worrall, J.J.; Rehfeldt, G.E.; Hamann, A.; [et al.]. 2013. Recent declines of *Populus tremuloides* in North America linked to climate. Forest Ecology and Management. 299: 35–51.

Wu, Z.; Dijkstra, P.; Koch, G.W.; [et al.]. 2011. Responses of terrestrial ecosystems to temperature and precipitation change: A meta-analysis of experimental manipulation. Global Change Biology. 17: 927–942.

Xu, C.; Gertner, G.Z.; Scheller, R.M. 2009. Uncertainties in the response of a forest landscape to global climatic change. Global Change Biology. 15: 116–131.

Zhang, J.; Marshall, J.D. 1994. Population differences in water-use efficiency of well-watered and water-stressed western larch seedlings. Canadian Journal of Forest Research. 24: 92–99.

Zhang, J.; Fins, L.; Marshall, J.D. 1994. Stable carbon isotope discrimination, photosynthetic gas exchange, and growth differences among western larch families. Tree Physiology. 14: 531–539.

Appendix 6—Vegetation Model Output, Vulnerability Assessment Descriptions, and Adaptation Options for Forest Vegetation in the Northern Rockies.

Appendix 6A—Dynamic Global Vegetation Model (MC2) Output for the Northern Rockies.

The following figures show output from the MC2 dynamic global vegetation model for the Northern Rockies region, including vegetation type distribution (figs. 6A.1–6A.4), carbon (6A.5, 6A.6), potential evapotranspiration (6A.7), and fire rotation (6A.8). See Chapter 6 for further information on the MC2 model and model simulation details.

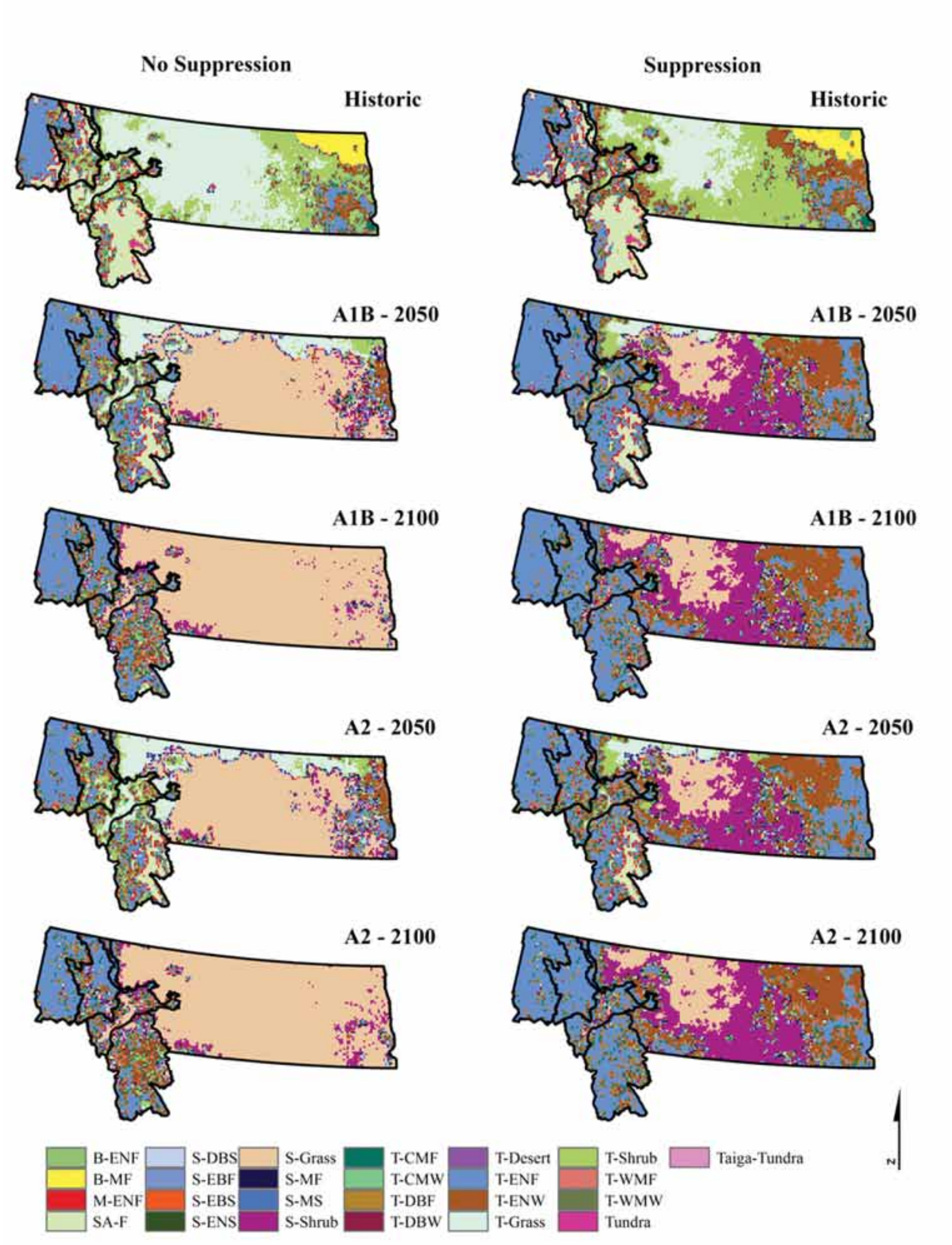

Figure 6A.1—Maps of MC2 vegetation type distributions for three time spans (historical, 2030–2050, and 2080–2100) and with and without fire suppression. Vegetation types are abbreviated as follows: B = boreal; M = maritime; S = subtropical; SA = subalpine; T = temperate; and ENF = evergreen needleleaf forest; ENW = evergreen needleleaf woodland; F = forest; MF = mixed forest; MW = mixed woodland; DBF = deciduous broadleaf forest; and DBW = deciduous broadleaf woodland.

2A

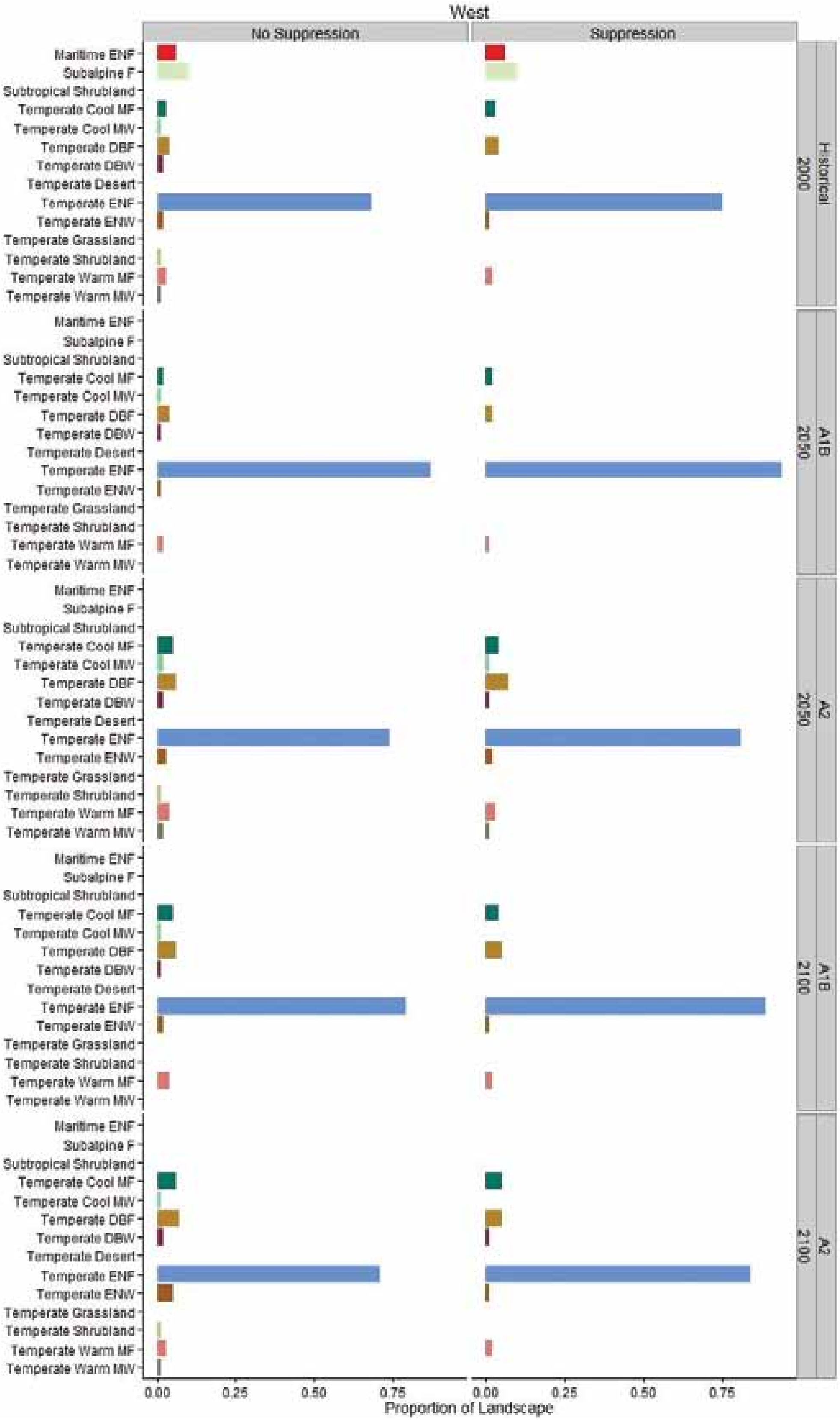

Figure 6A.2—MC2 Vegetation type distributions as proportion of the landscape for each subregion, where 2A = Western Rockies; 2B = Central Rockies; 2C = Eastern Rockies; 2D = Grassland; 2E = Greater Yellowstone Area; and vegetation type acronyms are: ENF = evergreen needleleaf forest; ENW = evergreen needleleaf woodland; F = forest; MF = mixed forest; MW = mixed woodland; DBF = deciduous broadleaf forest; and DBW = deciduous broadleaf woodland.

2B

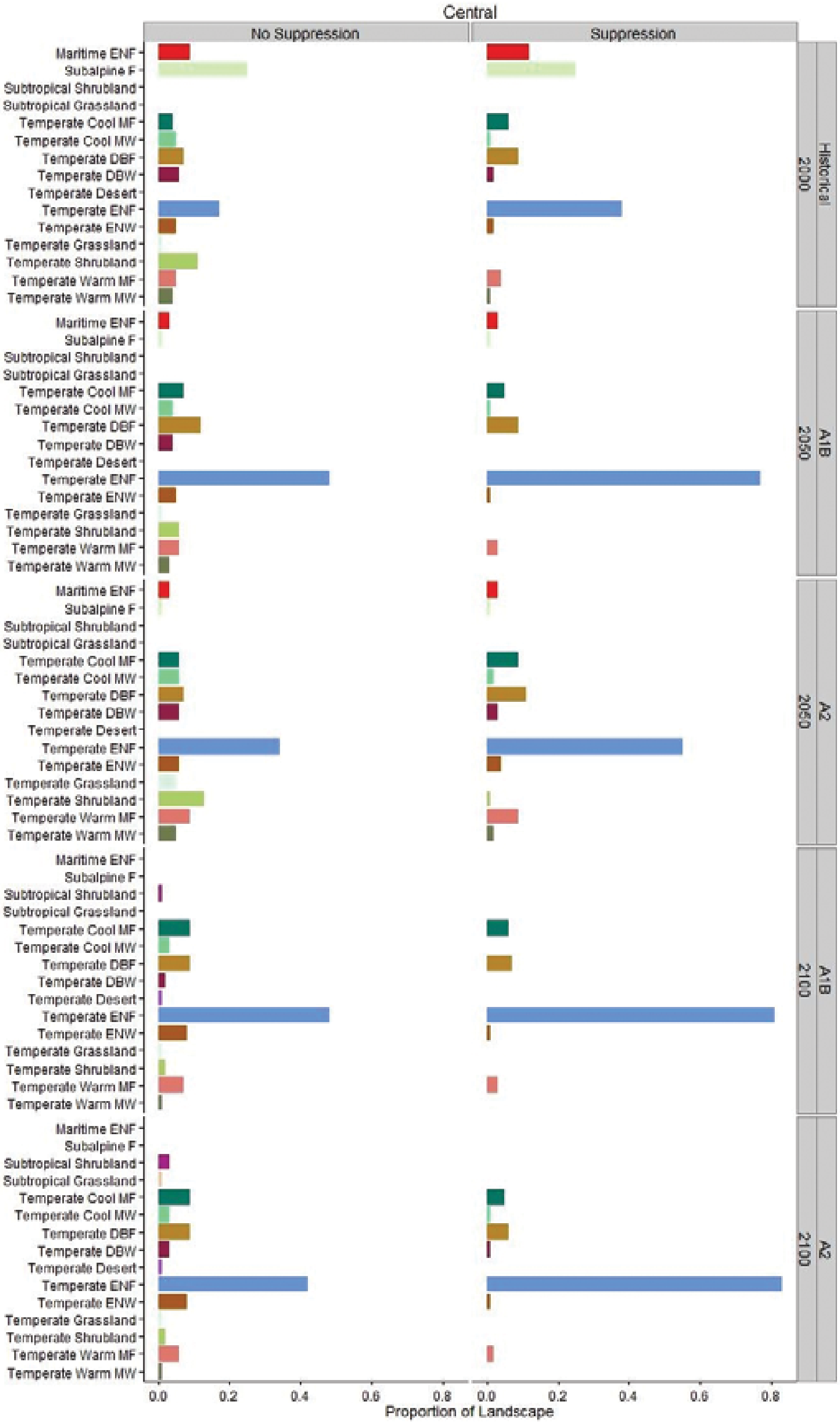

Figure 6A.2—*Continued.*

2C

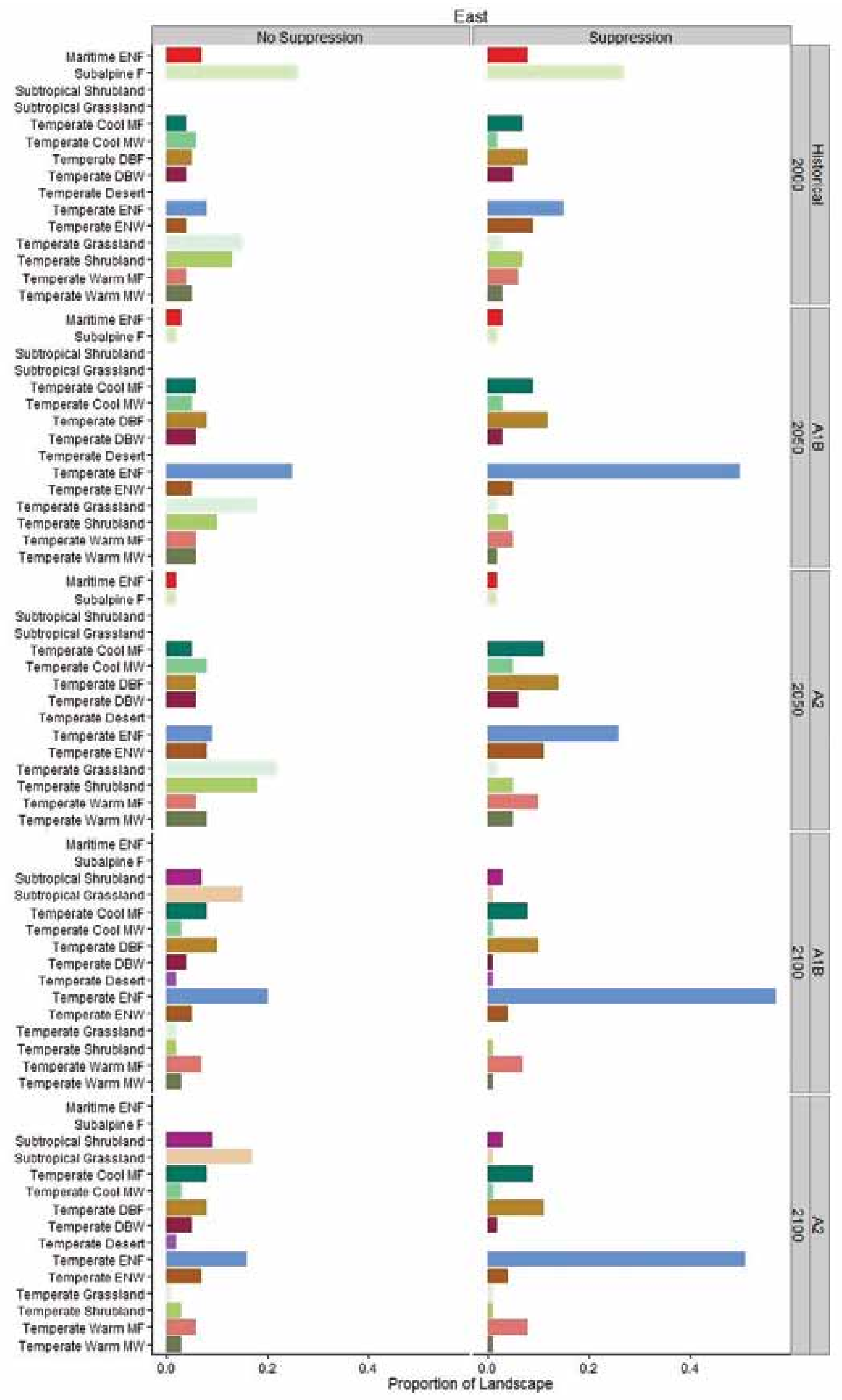

Figure 6A.2—*Continued.*

2D

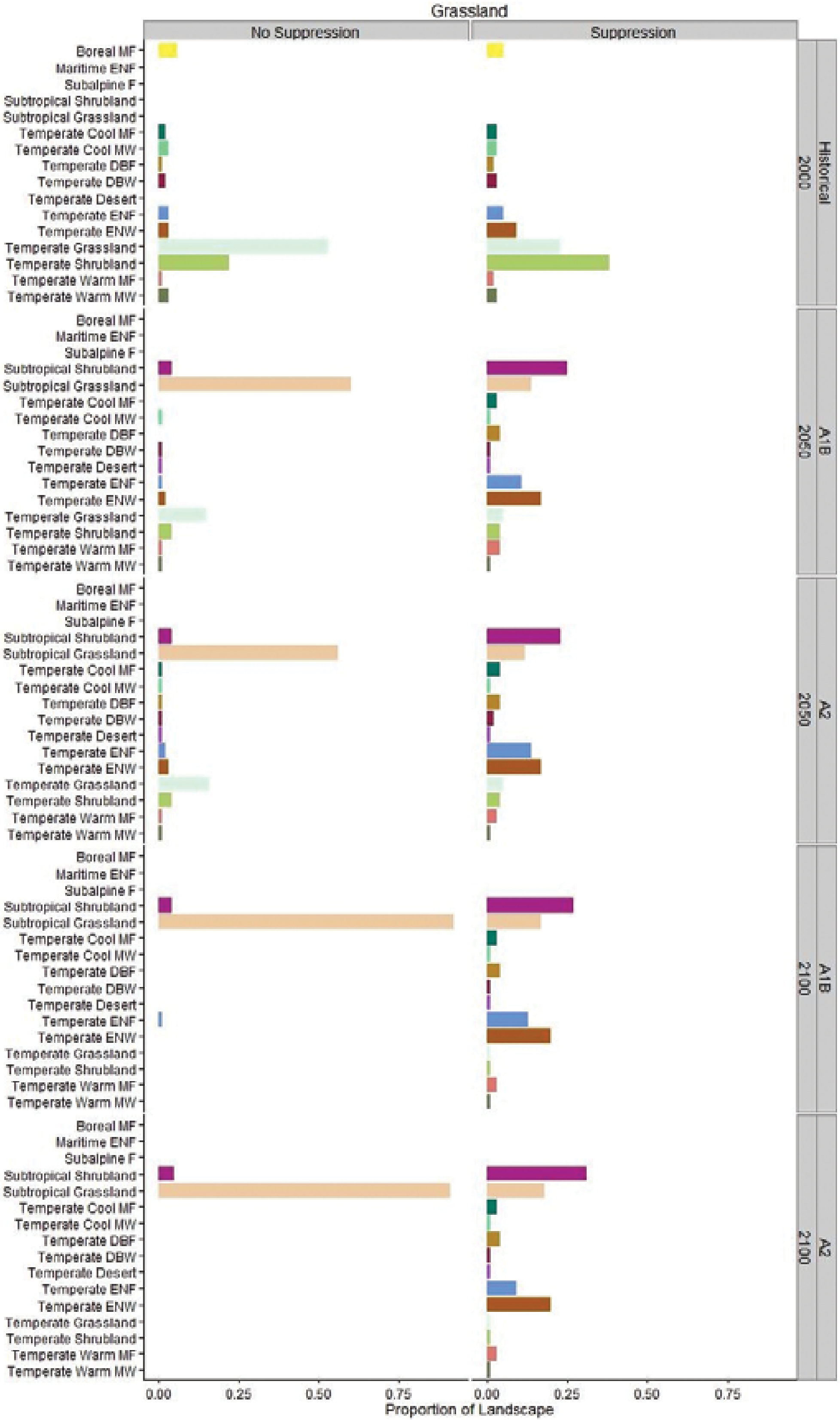

Figure 6A.2—*Continued.*

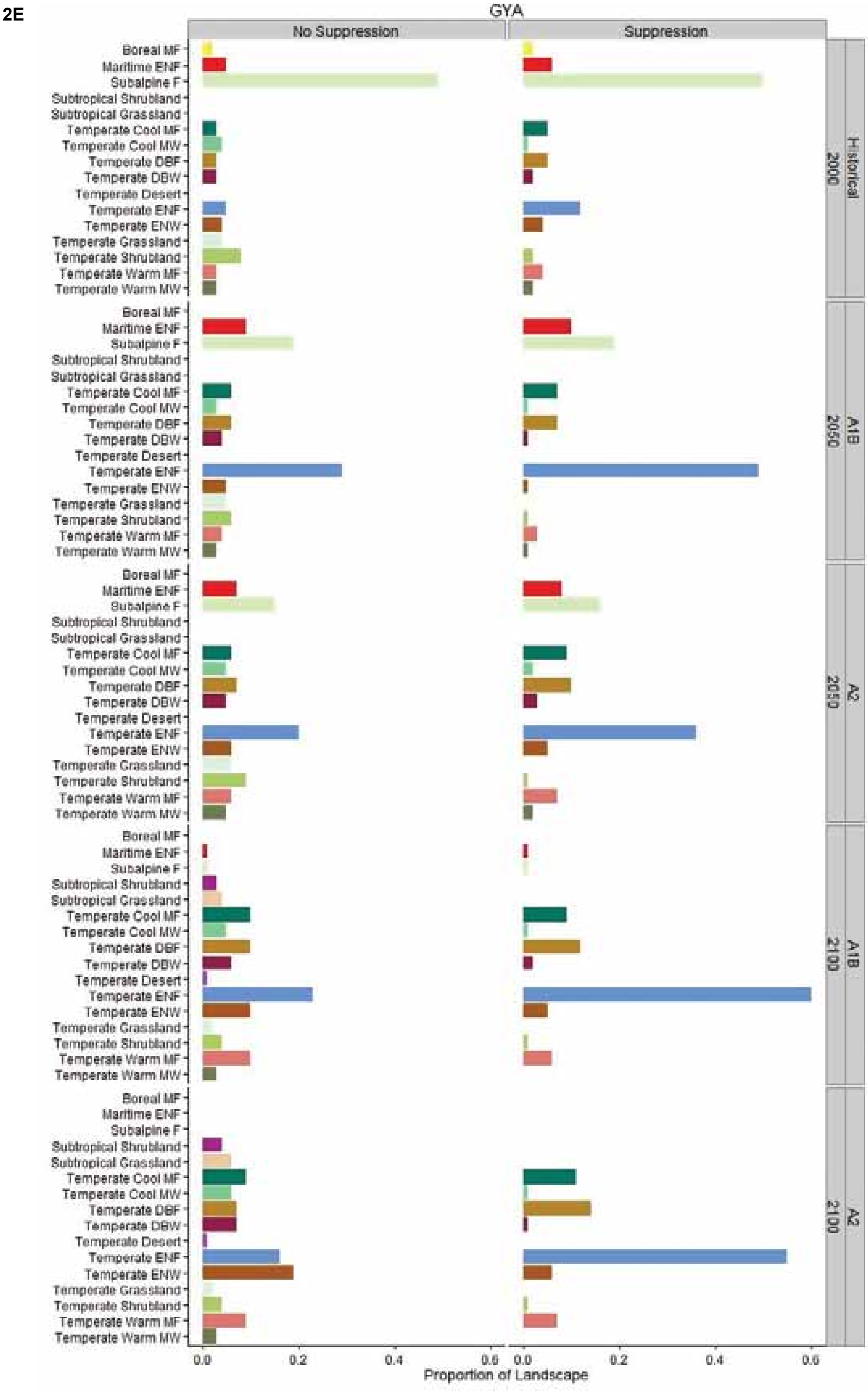

Figure 6A.2—*Continued.*

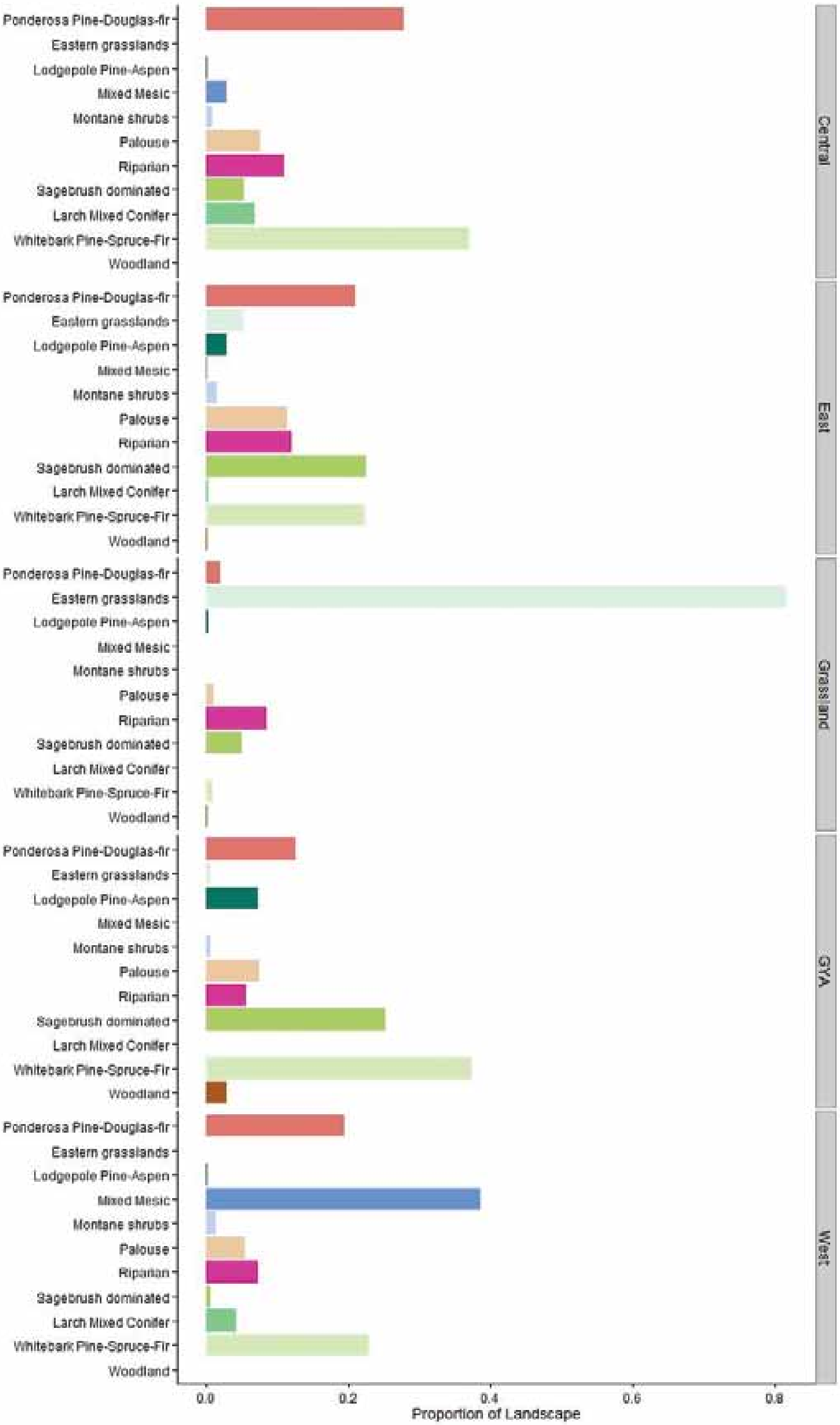

Figure 6A.3—Potential vegetation type distributions as proportion of the landscape for each subregion where vegetation types are abbreviated as follows: Ponderosa pine-Douglas-fir = dry ponderosa pine and Douglas-fir forests; eastern grasslands = eastern grasslands; lodgepole pine-aspen = lodgepole pine and aspen mixed conifer forests; mixed mesic = mixed mesic white pine, cedar, hemlock, grand fir forests; montane shrubs = montane shrubs; Palouse = Palouse (western grassland); riparian = riparian; sagebrush dominated = sagebrush-dominated systems; larch mixed conifer = western larch mixed conifer forests; whitebark pine-spruce-fir = whitebark pine-spruce-fir forests; woodland = woodland; and exotics = exotics.

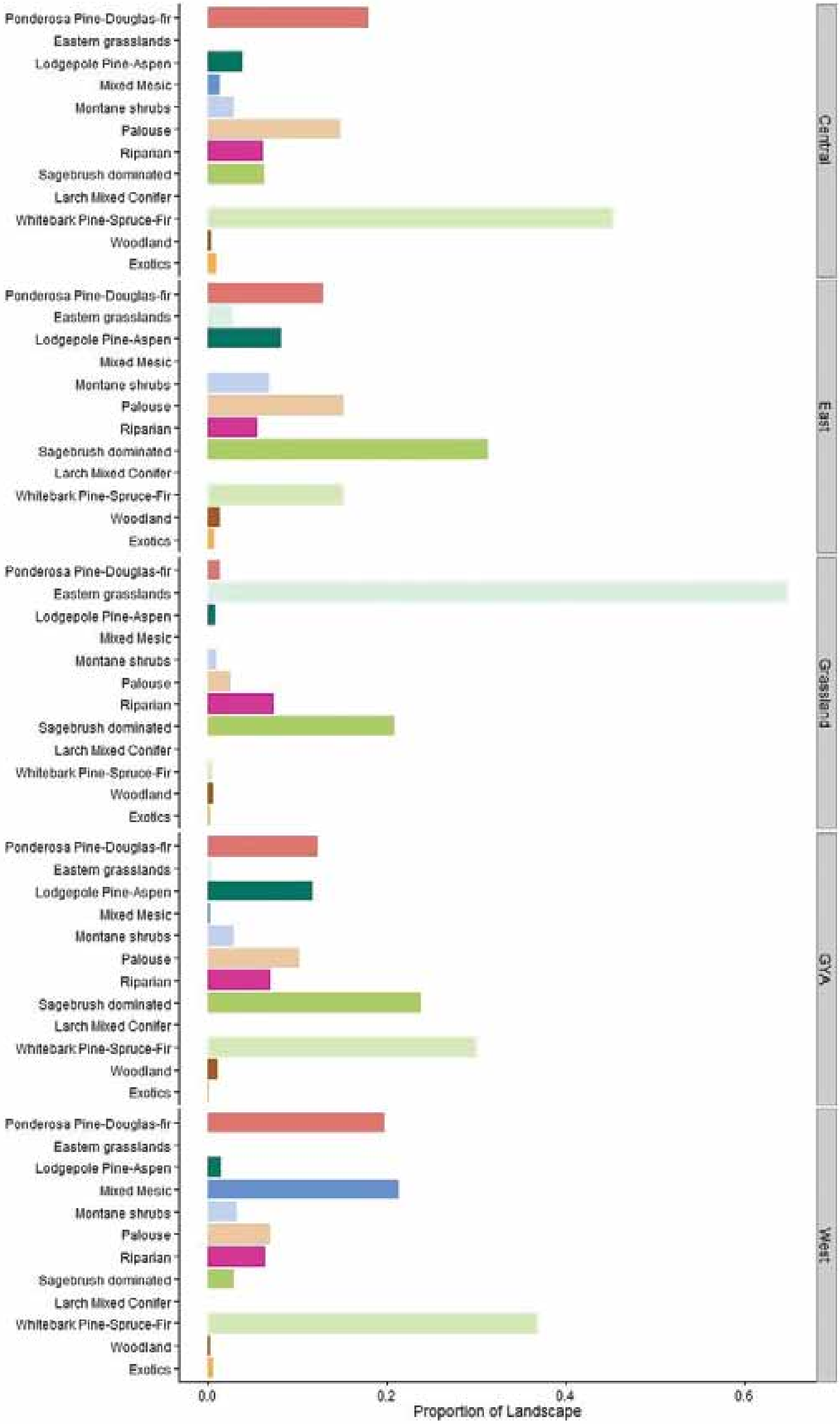

Figure 6A.4—Existing vegetation type distributions as proportions of the landscape for each subregion, where vegetation types are abbreviated as follows: Ponderosa pine-Douglas-fir = dry ponderosa pine and Douglas-fir forests; eastern grasslands = eastern grasslands; lodgepole pine-aspen = lodgepole pine and aspen mixed conifer forests; mixed mesic = mixed mesic white pine, cedar, hemlock, grand fir forests; montane shrubs = montane shrubs; Palouse = Palouse (western grassland); riparian = riparian; sagebrush dominated = sagebrush-dominated systems; larch mixed conifer = western larch mixed conifer forests; whitebark pine-spruce-fir = whitebark pine-spruce-fir forests; woodland = woodland; and exotics = exotics.

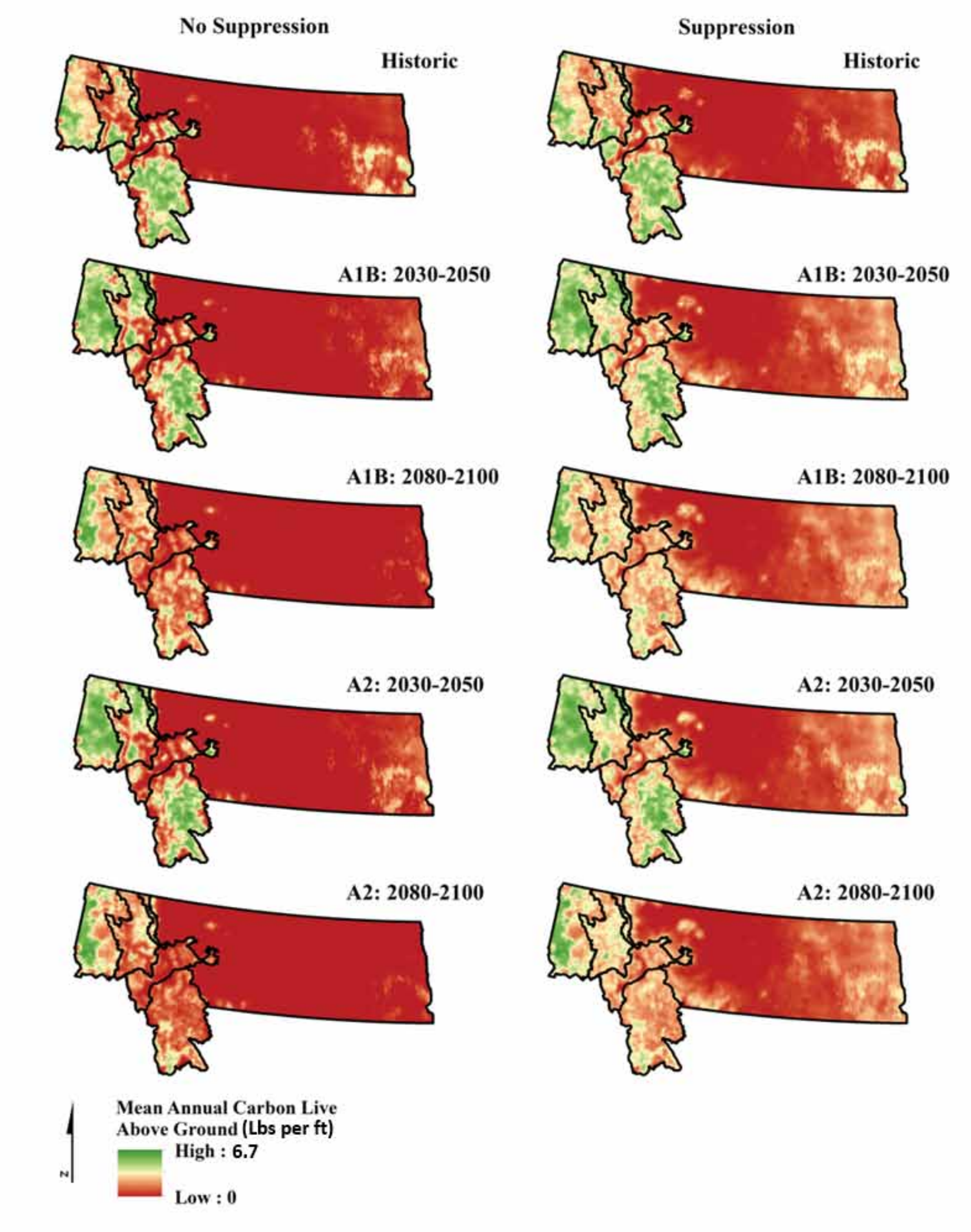

Figure 6A.5—Maps of aboveground live carbon averaged across each of three time spans (historical, 2030–2050, and 2080–2100) and with and without fire suppression, for the A1B and A2 emission scenarios.

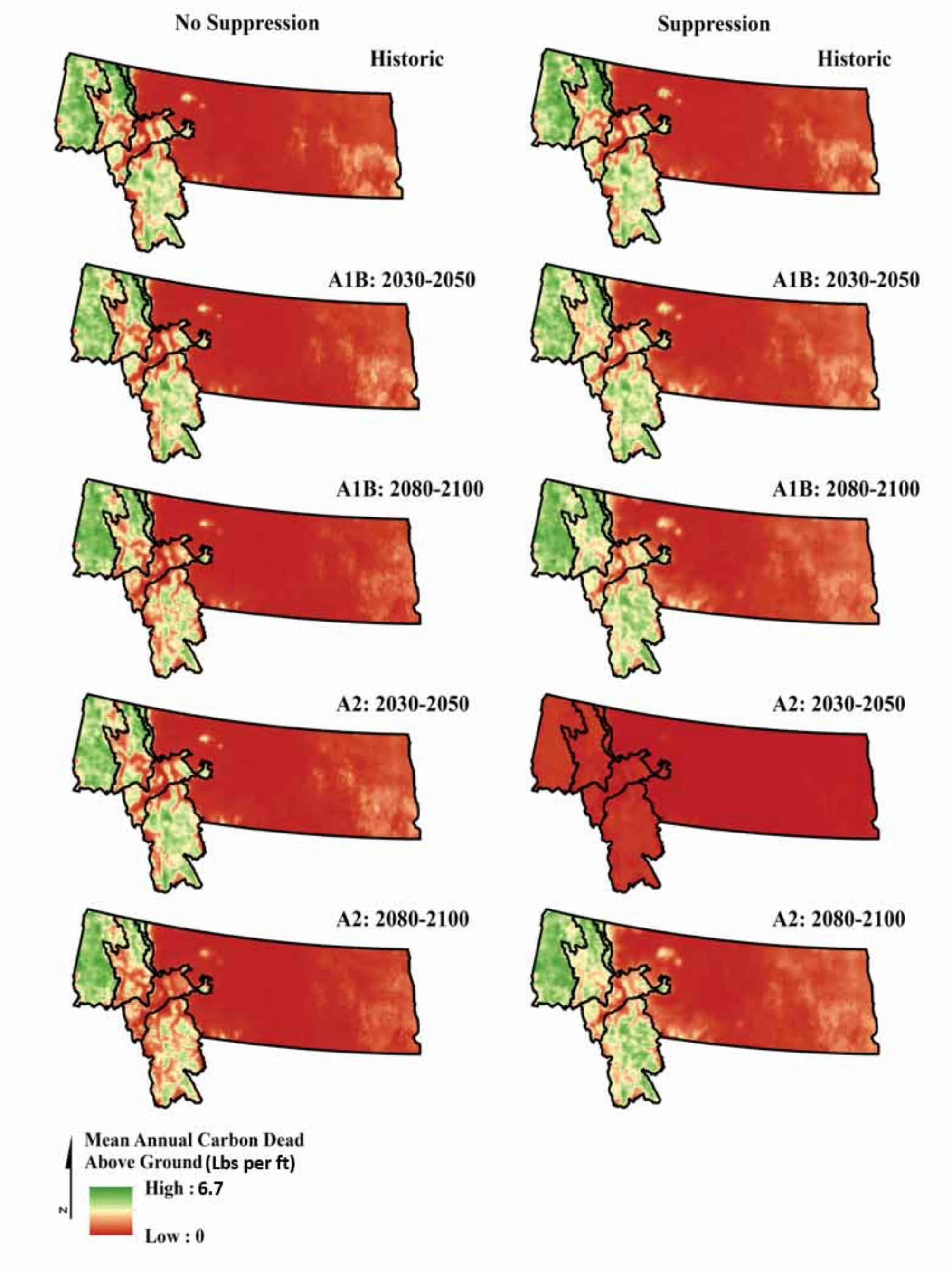

Figure 6A.6—Maps of aboveground dead carbon averaged across each of three time spans (historical, 2030–2050, and 2080–2100) and with and without fire suppression, for the A1B and A2 emission scenarios.

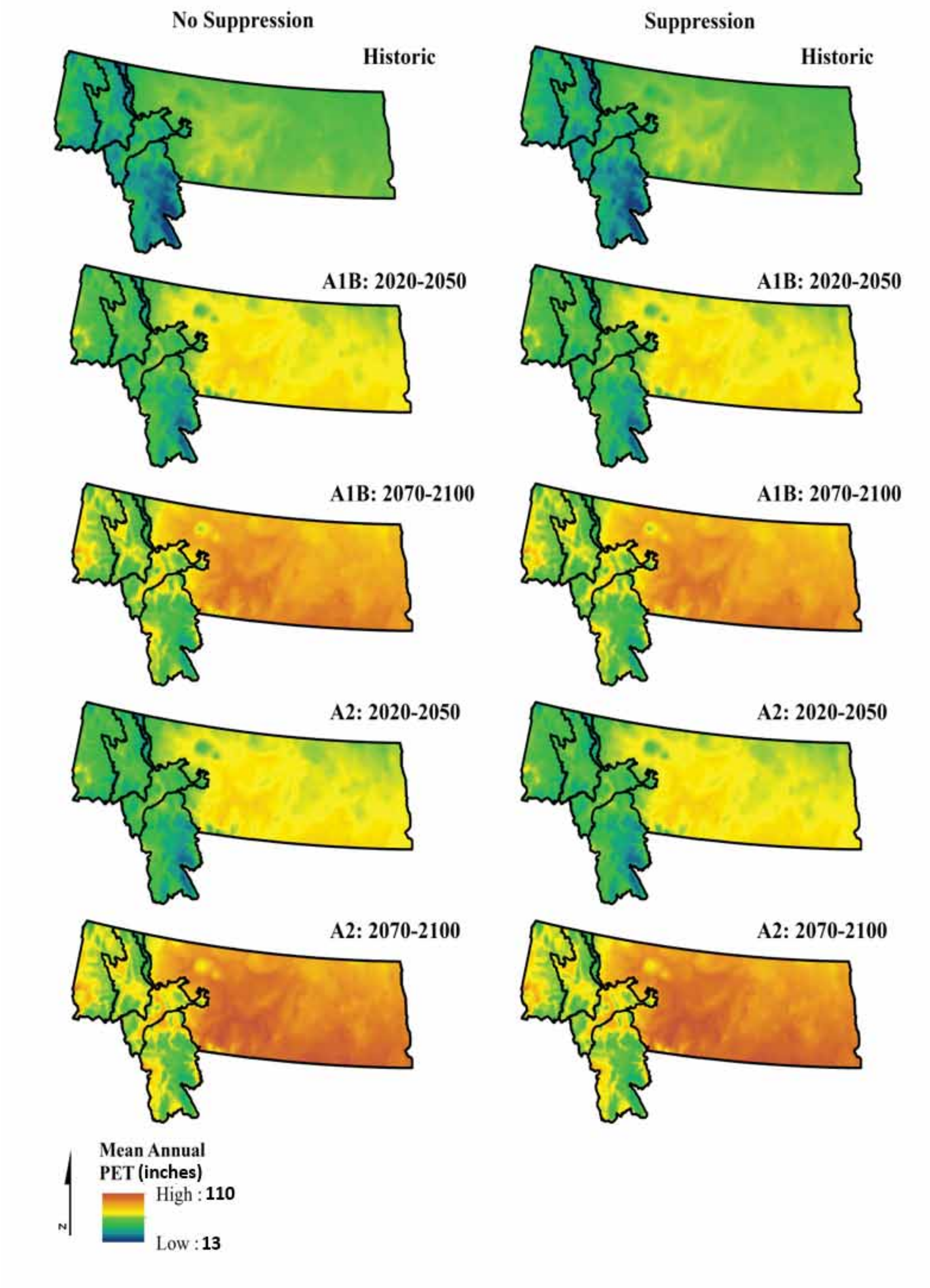

Figure 6A.7—Maps of potential evapotranspiration (PET) averaged across each of three time spans (historical, 2020–2050, and 2070–2100) and with and without fire suppression, for the A1B and A2 emission scenarios.

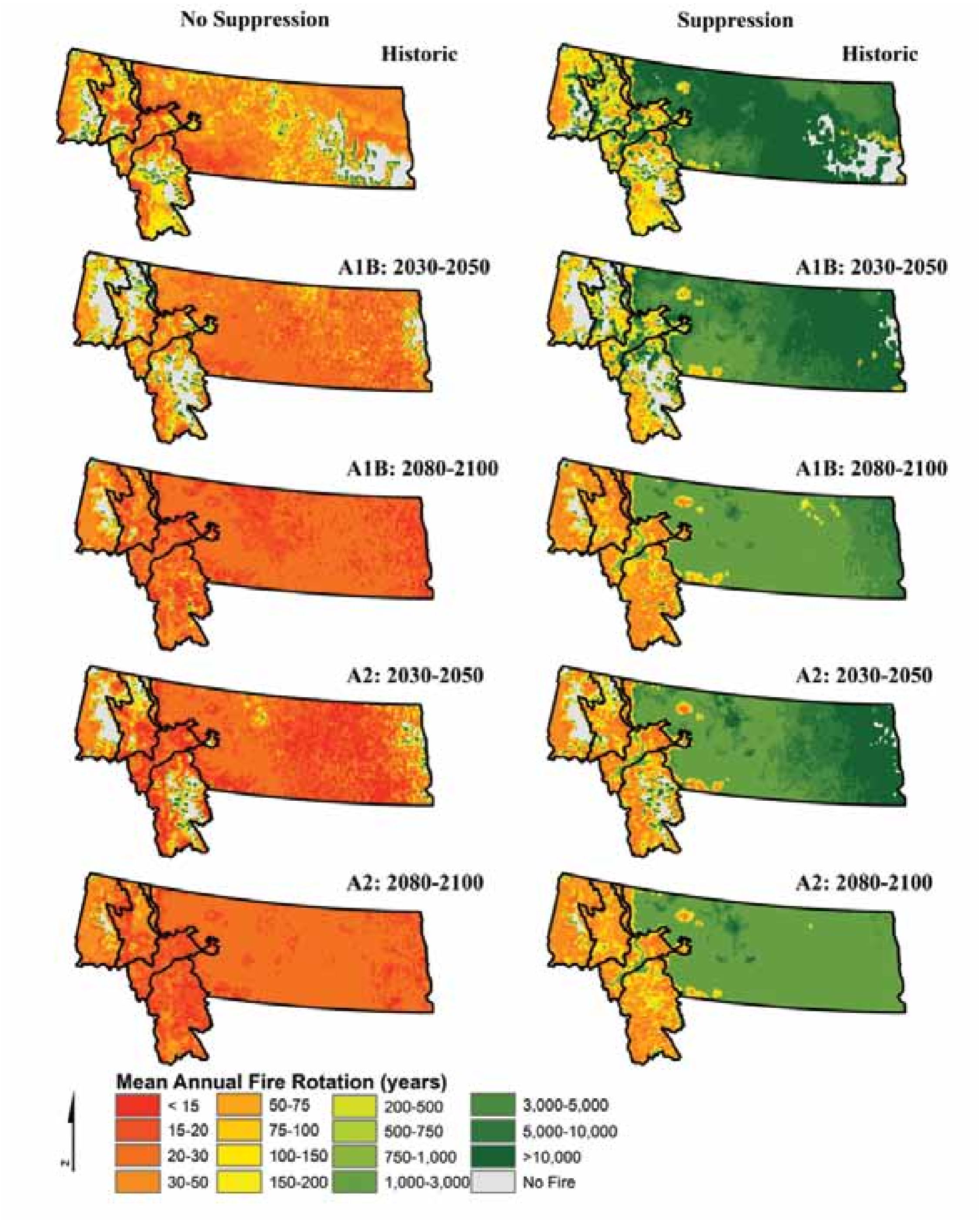

Figure 6A.8—Maps of annual fire rotation averaged across each of three time spans (historical, 2020–2050, 2080–2100) and with and without fire suppression, for the A1B and A2 emission scenarios.

Appendix 6B—Climate Change Vulnerability Assessments and Ratings for Tree Species, Forest Vegetation Types and Forest Resources of Concern.

The following tables describe climate change vulnerability assessments for tree species, forest vegetation types, and forest resource concerns (table 6B.1), and risk assessment and vulnerability ratings for the same species, vegetation types, and resources of concern (table 6B.2). See Chapter 6 for further discussion of climate change vulnerabilities for forest vegetation.

Table 6B.1—Vulnerability assessment for the three scales of analysis (species, vegetation type, resource concern) for vegetation of the Northern Rockies (NR).

Habitat, ecosystem function, or species	Broad-scale climate change effect	Current condition, existing stressors	Sensitivity to climatic variability and change	Expected effects of climate change
Species				
Limber pine	Warming temperatures, eastern portion of Northern Rockies anticipated increase in precipitation, less snow pack, variable precipitation during growing season.	Reduced abundance due to white pine blister rust infections, mountain pine beetle outbreaks, continued fire exclusion, and on wetter sites, limber pine dwarf mistletoe can cause mortality.	Shade intolerant, early-seral to pioneer species following fire or tree removal; difficulty in competing with other encroaching species on more productive sites; little to no reproduction at low tree densities (lack of effective pollination cloud) and those seeds that are produced have increased likelihood of inbreeding; minimum of 10 cone-bearing trees per acre is needed for dispersal by birds (i.e., corvids).	Increased growth; larger seed crops; increased seed dispersal into burned areas due to bird dispersal; lower seed germination due to warmer, drier conditions; loss of ectomycorrhizal associations, increased competition from wind-dispersed conifers; less blister rust infection due to higher temperatures and lower relative humidity disrupting the blister rust cycle; except in wave years; higher blister rust and dwarf mistletoe infections on eastside where precipitation is projected to increase; large and intense wildfires could threaten seed sources.
Ponderosa pine—var. *scopulorum*	Increasing temperatures; deeper and longer droughts; increasing fire severity and occurrence; shorter growing seasons and longer dormant seasons.	Higher than historical tree densities.	Generalist adaptive strategy; high phenotypic plasticity; early- to mid-seral species; moderate shade tolerance; well-adapted to drought.	Decreases in dwarf mistletoe and western gull rust damage; competitive capacity will increase; fire effects uncertain; more highly vulnerable to loss of disjunct and isolated populations, as compared to var. *ponderosa*; declining precipitation and variable spatial and temporal pattern may cause declines in regeneration, except in eastern portion of northern region where precipitation is expected to increase; increases in mountain pine beetle outbreaks; advancing competition, increasing western pine shoot borer occurrence; large and intense wildfires could threaten seed sources.
Ponderosa pine—var. *ponderosa*	Minor cold hardiness issues at lower elevations.	Precipitation pattern may favor regeneration; increases in pine beetle (western and mountain) *Lophodermium* sp. *Elytroderma, Armillaria* root rot, *Mycosphaerella pini*; extreme fire behavior; competition from grand fir and Douglas-fir.	Intermediate adaptive strategy at low to mid elevations; specialist adaptive strategy at higher elevations; moderate phenotypic plasticity; early- to mid-seral species; little shade tolerance; less adapted to drought.	Competitive capacity will increase; increasing fires may have both beneficial and detrimental effects (high uncertainty); dwarf mistletoe and western gull rust may decrease; loss of disjunct and isolated populations on edges of range; limited ectomycorrhizae availability at higher elevations; suitable substrates may not exist upslope, impedes both natural and artificial regeneration; large and intense wildfires could threaten seed sources.
Douglas-fir	Increase in temperature causing increase in soil moisture deficits and less available water especially at lower elevation dry sites.	Increase in tree density has increased risk of mortality from a large fire standpoint and will also limit regeneration and species distribution locally; root disease a major cause of mortality in northern Idaho and western Montana.	Sensitive to increasing temperatures and increasing soil moisture deficits, this will predispose Douglas-fir to other related mortality agents such as insect and disease.	At lower elevation southerly aspects expect ponderosa pine to be better able to cope with moisture deficits and disturbance such as fire, spruce bud worm; less seed source due fire size and due to cone production problems with spruce bud worm; mesic sites expect increase in mortality due to root disease; higher elevation southerly slopes may provide increased climate suitability for Douglas-fir; large and intense wildfires could threaten seed sources.

Table 6B.1—*Continued.*

Western larch	Increase in temperature causing increase in evaporative demand and soil moisture deficits.	Increase in forest density has increased mortality risk from a fire standpoint, especially existing large larch trees; past selective harvest and lack of fire have reduced species distribution and density, especially very large individuals.	Very sensitive to changes in temperature; spring frosts often reduce pollen, cone and seed production that leads to sporadic seed years; very sensitive to warm temperatures to establish regeneration on high energy, southerly slopes; rising temperatures and increasing soil moisture deficits will affect potential distribution; increasing amount of fire will likely benefit larch as long as it is not in overly dense forest conditions with poor vigor leading to loss of seed source after fire.	Range expansion and/or shift to more northerly aspects; reduction of size/age diversity of trees; larger fires may facilitate larch regeneration because the larger larch trees may provide seed on burned areas; increased drought on drier sites may exacerbate competition stress caused by invading shade tolerant species.
Western white pine	Warming temperatures; longer growing seasons; increased fire; increases in drought.	Local rates of blister rust infection may hold steady but remains highly susceptible to wave years of infection; increases in *Armillaria* root rot, mountain pine beetle (endemic—depends on white pine population size and tree size), *Lophodermium nitens*, *Lecanosticta aciocola* severe needle blight ongoing issue with cool-wet protracted springs; potential seed sources vulnerable to fire damage; critical thresholds for regeneration establishment may be possible resulting in ineffective pollen cloud, no seed production and higher probability of inbreeding.	Early spring frost and cold hardiness may not be physiologically in tune as individuals migrate upslope; moisture: lower elevation ecotones (<25″ per year precipitation threshold); moderate shade-tolerance; early- to mid-seral good competitive ability; up to 3 years seed viability in soil bank; generalist adaptive strategy; high phenotypic plasticity.	Increased growth; increased abundance with less grand fir provided there is a seed source or planting; less blister rust infection due to high temperatures and lower relative humidity except for wave years; if seed source exists, fire will increase growing space providing for regeneration potential; the species is dependent on ash cap on many sites, which may prevent migration to wetter/warmer sites; limited ectomycorrhizae availability at higher elevations; suitable substrate may not exist upslope so migration may not be expedited; both natural and artificial regeneration may occur with drought; pole blight may occur.

Quaking Aspen	Warming temperatures, decreasing snowpack, increase in severity and frequency of wildfires.	Although there are stable climax aspen communities, most aspen is a fire-maintained, early seral component of a forested community; stands are declining in number and size; stressors include competition with and shading by conifers, typically due to fire exclusion, domestic and native ungulate herbivory, and increasing temperature coupled with declining precipitation; reduction of soil moisture may cause severe water stress, which reduces aspen's ability to survive (e.g., sudden aspen decline) and to reproduce both vegetatively and by seed, thereby reducing genetic variability.	Sensitivity varies based on site characteristics, primarily soil moisture and solar radiation; it is mostly shade-intolerant; ubiquitous across most of North America; aspen is most persistent on sites with high solar radiation coupled with moist to wet soils; it does not tolerate extended drought; highly fire adapted and regenerates abundantly after stand-replacing fire, although it can regenerate in conifer dominated stands in some settings; it will persist or possibly increase with warmer temperatures as long as there is sufficient soil moisture; fringe communities may succumb to sudden aspen decline with long-term and severe water deficit, which will kill the roots.	Communities on warmer, drier sites could decrease due to water deficit; some stands may have significant mortality with little or no regeneration due to herbivory; sudden aspen decline has been associated with severe, prolonged drought, particularly in aspen stands that are on the fringe of the species' distribution (warmer and drier sites than those typically considered optimal for aspen persistence); fewer and smaller stands and of those that persist, there will be increased plant stress due to increased severity of summer droughts; increased fire frequency may likely favor aspen regeneration by removing shading conifers; younger stands (<40 years old) may be more resilient to drought and frequent fires could favor aspen on moister sites; severe fire may kill shallow root systems and eliminate aspen in some hotter and drier sites; growth may increase because photosynthetic rates appear to increase more in aspen than other tree species as atmospheric carbon increases, but this may be offset by increased drought stress and increased atmospheric ozone, which reduces photosynthesis and may increase susceptibility to insects and disease; higher herbivory (browsing) on regenerating stands is possible as adjacent upland vegetation senesces and desiccates earlier in the growing season; areas with mountain pine beetle-caused conifer mortality (especially in lodgepole pine) may release aspen and regenerate once the conifer canopy is removed; conifers use more water than aspen, so more aspen stands mean more streamflow; aspen forms natural fuel breaks that can be used effectively in fire management; pathological rotation is short-lived.
Grand fir	Increasing drought; longer growing seasons; increased fire and disease; highly variable weather.	Fire exclusion has increased grand fir in dry and mesic sites, but increased tree densities have also stressed fir trees, contributing to increased fuel loadings, higher root rot, and greater insect damage and mortality.	Late-seral, shade tolerant species highly susceptible to fire; usually exists in areas where competition and tree density is high so most trees are stressed; highly stressed trees are more susceptible to climatic fluctuations and trends.	On xeric sites, increased drought and longer growing seasons will exacerbate stress from competition, resulting in high mortality mainly from insects and disease; longer fire seasons and high fuel loadings from exclusion will also reduce this species; on mesic sites, longer growing seasons coupled with higher temperatures may increase growth rates and regeneration success, thereby increasing tree density and competition; fire will reduce grand fir dominance at across the landscape.
Western redcedar	Longer growing seasons; warmer temperatures; highly variable weather; enhanced warm and cold cycles; early frosts.	Root and butt diseases: low susceptibly to pathogens.	Generalist adaptive strategy; high phenotypic plasticity; high shade-tolerance; long-lived; seed production good; vegetative potential.	Increased growth rate; fire adds uncertainty; warm/cold cycle may facilitate red-belt and adversely affect western redcedar (early warming followed by early frost); cedar flagging from dry seasons (entire branch is shed, interferes with seed production); ash cap dependent, which may prevent it to migrate to wetter/warmer sites.

Table 6B.1—*Continued.*

Western hemlock	Longer growing seasons; warmer temperatures; highly variable weather; early frosts during the growing stage.	Confined to the moister portions of northern Rockies; susceptible to annosus root rot, *Echinodontium tinctorium*; High tolerance to Armillaria root rot.	Provisionally tends to be a generalist adaptive strategy; high shade tolerance; needs ample moisture; susceptible to spring frost; very good competitor; high seed producer; seed viability only lasts one year in soil bank; susceptible to acid rain; ash cap dependent.	May stay the same and potentially not change its current distribution; vulnerable to water deficits so declines are possible on the drier sites; since the species is ash cap dependent, migration may be retarded to wetter/warmer sites without ash cap soils.
Lodgepole pine	Increasing temperatures; longer droughty periods; increasing fire occurrence, frequency, and severity; increased productivity.	Advancing succession due to fire exclusion is contributing to declines in lodgepole pine in many areas; current increases in burn areas are creating many new lodgepole stands and some may become dense thickets; increased drought may exacerbate stress from other factors including competition, endemic insects and diseases, and wind; warming temperatures may heighten bark beetle activity resulting in more frequent and severe epidemics.	Shade-intolerant conifer that has a wide climatic amplitude in subalpine areas; exists on a wide variety of soil types and may be the only species to inhabit infertile and well drained sites; moderately drought tolerant; reproductive success depends on level of serotiny; well adapted to colonize post-burn environments; highly susceptible to bark beetles, especially when in stress from endogenous and exogenous factors such as competition, fire damage, and drought.	Longer drought periods and warmer temperatures may decrease growth and regeneration on the driest sites (lower elevation lodgepole stands); well adapted to increases in fire occurrence and size depending on level of serotiny, but it may be eliminated from sites where fires reburn stands before established seedlings and saplings become reproductively mature. In mesic subalpine sites, continued fire exclusion coupled with higher productivities will heighten competitive interactions and put more lodgepole pine into stress thereby increasing mortality, insect and disease vulnerability, canopy and surface fuels, and accelerating succession toward subalpine fir; conversely, increasing fire could expand lodgepole pine occurrence, even when fires are large and severe; increasing insect (i.e., bark beetles) outbreaks may further acceleration toward non-host, shade-tolerant species.
Subalpine fir	Increased disturbance frequency and severity; highly variable weather and climate; decreasing snowpacks; lengthening growing seasons.	Fire exclusion has increased abundance of this species on many subalpine and upper subalpine landscapes; many current stands have high densities and trees may be stressed from competitive interactions resulting in increasing susceptibility to disturbances; increasing drought could further exacerbate competitive stress and increase mortality.	Highly vulnerable to subtle changes in climate; shade-tolerant species that is an aggressive competitor in subalpine areas; uniquely adapted to quickly occupy gaps in subalpine forest canopies; relatively intolerant of drought; unable to mature when seasonal drought is common; not adapted to disturbance, especially fire, with high mortality even after low severity fires; frequent cone crops.	Longer growing seasons and reduced snowpacks will increase regenerative success, especially in those high-elevation areas where snow historically controlled regenerative success; higher productivity in subalpine forests may increase regeneration and species densities, eventually resulting in high competitive stress making these fir stands vulnerable to high mortality and therefore less resilient; declines of the species on drier sites may result from new drought regimes reducing regeneration success; increases of the species on moister sites will result from increased regeneration and competitive advantages; fir may gain in upper subalpine and timberline environments that are controlled by snow dynamics; subalpine fir could also increase as it replaces rust- and beetle-killed whitebark pine, yet whitebark pine can also act as a nurse crop to facilitate subalpine fir establishment; increased fire would decrease fir; the future of subalpine fir depends on both fire suppression levels and climatic responses.

Table 6B.1—*Continued.*

Engelmann spruce	Increased disturbance frequency and severity; highly variable weather and climate; decreasing snowpacks; lengthening growing seasons.	Spruce is usually associated with fir in the NR; it occurs as a minor to major component of many subalpine stands and only dominates in wetland or special land types; fire exclusion has increased abundance of this species on many subalpine and upper subalpine landscapes; many current stands have high densities and trees may be stressed from competitive interactions resulting in increasing susceptibility to disturbances; increasing drought could further exacerbate competitive stress and increase mortality.	Spruce is highly susceptible to changes in climate; it is not as an aggressive competitor and often is only a minor portion of a stand; it is highly vulnerable to drought; it can quickly regenerate in severely burned microsites providing there are seed sources; highly susceptible to windthrow and wind damage.	Losses of spruce in the drier portions of its range, especially in those seasonal moist sites that will now be dry; not well adapted to fire so major declines are expected in burned areas, but these declines may be offset by increased regeneration on burned areas with mineral soil substrates; continued suppression activities may maintain spruce on the landscape but it may be at lower levels due to increased drought; it may increase in the upper subalpine when snowpacks become consistently lower and soil becomes drier thereby allowing spruce to encroach into glades, meadows, and balds.
Whitebark pine	Warming temperatures, lower snowpacks, highly variable weather, increasing fires in both intensity and severity, increasing insect and disease outbreak frequencies and severities; increase in populations of mountain pine beetles through possible univoltism shifts.	Reduced abundance due to white pine blister rust infections, mountain pine beetle outbreaks, and continued fire exclusion; survival due to cold hardiness in seedlings and saplings in frost pockets and swales.	Inability to compete with encroaching conifers due to low growth rates, moderate shade intolerance, and seed dispersal characteristics. Little to no reproduction may occur once tree densities are low; long-lived species that lasts through climate epochs.	On many upper subalpine sites there may be increased growth, larger seed crops, increased seed dispersal into increasing areas being burned due to bird-mediated dispersal providing there are adequate seed sources; lower seed germination due to warmer, drier conditions; possible loss of ectomycorrhizal associations; lack of suitable substrates (suitable soils) as species moves upslope; increased competition from wind-dispersed, shade tolerant conifers with fire exclusion; less infection due to higher temperatures and lower relative humidity, disrupting the highly variable blister rust cycle except in wave years; increased fire may provide caching habitat for nutcrackers that allow decades of development in the absence of competition from fir and spruce; losses in whitebark pine due to increased beetles, rust, and fire; may be offset by increases in growth, cone crops, and abundant regeneration in burned areas, but management actions are needed to augment natural regeneration by planting and direct seedling rust-resistant pine.

Table 6B.1—*Continued.*

Alpine larch	Warming temperatures; longer growing seasons; smaller snowpacks; less summer water; possible summer droughts; increased fire.	Found in moist upper subalpine cove sites with abundant above and belowground moisture; exists in mixed stands of whitebark pine and sometimes subalpine fir; can form extensive stands in sub-irrigated upper subalpine areas.	Very shade intolerant; intergrades with western larch; while this species is quite sensitive to shifts in climate, it may initially increase because it may colonize upper subalpine non-forest sites quicker than other species, but the newly established individuals may be unable to survive to maturity as drought and temperatures increase, climates become more variable, and fires increase in the upper subalpine; susceptible to damage from fire.	Increased growth rates; greater ability to populate upper subalpine and treeline ecotones because of wind-aided seed dispersal and evergreen regenerative properties; increased fire may reduce many stands that historically were too wet to burn; on mesic sites, enhanced growing environment may increase competition from other more competitive, shade-tolerant conifers; lack of whitebark pine seed caching because of depressed cone crops may favor alpine larch dominance in areas that currently lack trees (e.g., treeline, subalpine balds, meadows, glades); on xeric sites, lack of summer groundwater may contribute to higher water stress and lower growth rates; years with deep droughts may kill established regeneration; alpine larch may decline in those areas with lower water availability and declining groundwater flow.
Green ash	Warming temperatures, decreasing snowpack, increase in severity and frequency of wildfires.	Domestic and native ungulate herbivory has affected both structure and composition of these communities.	Green ash has broad ecological amplitude and can survive droughty conditions, but persists optimally in moist sites; as soil moisture declines, marginal sites may become less favorable for regeneration and survival of young trees; there will probably be increased vegetative regeneration and decreased production of seedlings following fire—fire often kills green ash seed on or near the soil surface, restricting seedling recruitment to surviving seed producing trees.	Green ash may benefit from increased temperatures; seedling growth may increase with increasing soil temperatures; after increased fires, green ash has both root crown and epicormic sprouts, and both are typical following disturbances such as fire; fire may be very important in woody draws and riparian areas of the Great Plains; woody draws are long and narrow, so even though they are more moist than surrounding uplands, they can burn during frequent fires in surrounding grasslands. Low-severity fires might promote regeneration by thinning stands and stimulating sprouting (the primary response to fire); browsing pressure will likely increase with increased drought, as upland grasses and forbs desiccate and senesce earlier, or are replaced by invasive, less palatable species.

Table 6B.1—*Continued.*

Cottonwood	Warming temperatures; decreasing snowpack; increase in severity and frequency of wildfires.	There has been a reduction in area due to conversion and development of floodplains; composition and structure of cottonwood forests have been altered due to changes in flow regimes; structural alteration (typically simplification) of the channel (e.g., levees, bank-armoring structures) has likely contributed to channel widening, or channel incision and loss of floodplain interaction; nonnative trees that are more drought tolerant are already present along rivers and streams in eastern Montana; increased drought stress will likely favor these species over cottonwood; additional stressors include roads, along with domestic and native ungulate browsing (particularly on young cottonwoods).	All species of cottonwood require saturated, but aerobic substrates and full sunlight to germinate and persist; any alteration of hydrologic flow regime (e.g., timing, magnitude and duration) will affect floodplain interaction and plant available water that may reduce recruitment and establishment of seedlings (cottonwoods regenerate primarily by seed); decreased streamflows and floodplain interaction may result in a shift in streamside vegetation to upland species, along with reduced growth and regeneration, and increased mortality of cottonwood; because cottonwoods are shade intolerant (require full sunlight), any conifers that establish on the drier fluvial surfaces will grow tall enough to eventually shade out the cottonwoods; as snowpacks decline and melt earlier, peakflows will be reduced and variation in discharge will decline, leading to a loss of various fluvial (depositional) surfaces along the stream, on which cottonwood germinate; the system becomes less complex; there may be fewer recruitment events; in addition, there may be a shift in timing of peakflows to earlier in the season, before cottonwood seed is viable for germination, resulting in both decreased germination and establishment of young cottonwoods; increased demand for water (additional diversions, reservoir expansions) and increased browsing pressure (adjacent upland vegetation senesces and desiccates earlier in the growing season) may also lead to a decline in cottonwood; sizes of cottonwood forests may decrease as these fluvial surfaces are less frequently inundated; there may be little to no recruitment of young cottonwoods.	Timing of flooding is critical to germination success, and establishment of young (sapling, pole) cottonwoods both diminished; this will vary based on winter snowpack, and amount and timing of snowmelt (and associated peakflows); seedlings establish on moist to wet bare mineral soil, typically on stream bars, in full sunlight; as the snowpack declines and melts earlier, there will be reduced, more stabilized flows (loss of extreme high and/or low flows) and/or a shift in timing of peakflows to earlier in the season, before cottonwood seed is viable for germination); with earlier peakflows and less discharge, germination success is diminished; increased demand for water (additional diversions, reservoir expansions) and increased browsing pressure (as adjacent upland vegetation senesces and desiccates earlier in the growing season) will also likely lead to a decline in cottonwood.

Table 6B.1—*Continued.*

Vegetation types				
Dry ponderosa pine and Douglas-fir forests	Increase in temperature, causing increase in soil moisture deficits and less available water especially at lower elevation dry sites.	Increased density has put at risk increased mortality from a fire-severity standpoint, root disease a major cause of mortality in northern Idaho and western Montana.	Sensitive to increasing temperatures and increasing soil moisture deficits; this will predispose Douglas-fir to other related mortality agents such as insects and disease; this may give ponderosa pine an advantage on these settings	At lower elevation southerly aspects, expect ponderosa pine to be better able to cope with moisture deficits and disturbance such as fire, spruce budworm; less seed source due to fire size and cone production problems with spruce budworm; mesic sites expect increase in mortality due to root disease; higher elevation southerly slopes may provide increased climate suitability for Douglas-fir, whereas ponderosa pine will be favored at lower elevations; patches size will increase due to severe fire if density reductions are not implemented.
Western larch mixed-conifer forests	Increase in temperatures will likely rearrange current species associations on higher-energy slope positions	Larch forests have been reduced in extent due to successional effect from fire suppression, and preferential harvest; forest density increases have been substantial, and it now exists in uncharacteristic dense forest conditions in many areas. In northern Idaho, forest density (and productivity) was a bit denser due to warm mesic climate and deep ash capped soils. In areas once dominated in northern Idaho by western larch, western white pine, and ponderosa pine, they are now dominated by mixed grand fir and moist-site Douglas-fir forests. The spatial pattern of forest structure has been homogenized in many areas, leading to a continuous horizontal fuel profile atypical of historic landscape structure in many areas once typical of mixed-severity regime.	Larch is sensitive to changes in temperature; very sensitive to warm temperatures to establish regeneration on high energy, southerly slopes; rising temperatures and increasing soil moisture deficits will affect potential distribution and pattern of larch forests especially on high energy aspects. Increasing amount of fire will likely benefit larch as long as it is not in overly dense forest conditions with poor vigor with continuous horizontal fuel profile across landscapes.	Larch is highly vulnerable to increase in temperature and uncharacteristic fires in dense forest settings; the loss of large tree structure and larch regeneration success could be a major effect due to the current lack of heterogeneity and patch densities; this is especially true for high-energy topographic locations; increases in soil moisture deficits could retract the range of western large to more northerly slopes with deep soils; cone production could be positively affected with increasing temperatures, which could cause cone maturation to be earlier and could mean earlier fire seasons may still be timed with cone production and seed cast after fire. Adaptation ability for cone production and seeding distance and regeneration ability may be reduced if connectivity is reduced due to very large and more frequent severe fires occurring. High forest density and low landscape pattern diversity compared to HRV may put at risk medium, large, and old-growth larch stands due to increasing moisture deficits and stand replacing fire. Simplification of within- and between-patch structure due to increased fire severity and size could lead to loss of diversity and loss of important wildlife habitat such as cavity nesting habitat for birds and mammals. In northern Idaho, the change in species composition of the forest to more intolerant to shade species has resulted in a forest much more susceptible to wide spread root disease mortality. These areas involve millions of acres on which less carbon sequestration will take place given the relatively novel species composition of today's forests. Given the likely increase in soil moisture deficits in the future, root disease effects are not likely to reduce; lower tree densities of reproductively mature larch and loss of connectivity between populations will increase inbreeding depression as a result of disruption of an effective pollen cloud.

Table 6B.1—*Continued.*

Lodgepole pine and aspen mixed conifer forests	Increasing temperatures; longer droughty periods; increasing fire occurrence, frequency, and severity; increased productivity.	Many stands of this type are succeeding to subalpine fir-spruce due to fire exclusion; aspen has been declining due to lack of fire and increasing drought; healthiest areas are in wilderness.	This type is more sensitive to management actions than climate in that continued fire exclusion will ensure their decline; this type thrives with fire and will even survive insect and disease outbreaks if fire is present on the landscape.	This cover type could expand in the future with increasing fires and warming of the subalpine; disturbances may eliminate competing conifers and facilitate serotiny-aided lodgepole pine regeneration; aspen may decline on the drier parts of its range, but could increase and make major advances into the subalpine as fires burn competing conifers and temperatures moderate creating favorable climates; if fires are too frequent, this cover type may be replaced by semi-permanent shrub-herb, but as long as fire return intervals are greater than the reproductive age, lodgepole and aspen should prevail; as fires increase, more areas in this type will be early seral, creating more heterogeneous landscapes with more patches of pine and fir mixed with aspen.
Mixed mesic white pine, cedar, hemlock grand fir forests	Increasing temperatures; more drought; more fires.	This type is limited to the northwestern portions of the Northern Rockies and much of this area has had management activities (harvest); western white pine occurrence has been severely reduced by blister rust but has shown increases in some portions of this type; unharvested stands are becoming more dense creating conditions that favor rot, insects, and disease damage.	This type includes a fire-tolerant, fast-growing, early-seral species (western white pine) with a collection of late-seral, shade-tolerant, highly competitive species so the type may actually not change in coverage as fires facilitate conversion to pine on these sites.	Western white pine may become a more dominant feature of this type; declines in cedar and hemlock are possible; moisture changes are probably not limiting on these sites as much as the dependence on ash cap soils; major gains in the type are probably not possible because of the limited distribution of ash cap soils in some areas and the decrease in moisture outside of the current type's range. Similar to other types, the distribution of seral types may be more heterogeneous due to fire, but the long period of fire exclusion might foster atypical high-severity fires that might burn entire landscapes in some areas, and this might result in homogeneous pine stands providing sufficient rust resistance and seed sources.
Whitebark pine-spruce-fir forests	Declining snowpacks; increasing fire; increasing temperatures.	This type is probably increasing from effective fire exclusion; losses in whitebark pine are successionally replaced by fir-spruce; the low-elevation spruce-fir types are becoming more dense and crowded.	This type might not be as sensitive as other more xeric sites to direct climate change impacts, because there is abundant water, and predicted increases in both regeneration and growth may actually increase its climate resilience; increasing fires may cause a shift to more early seral communities and if whitebark pine populations were not experiencing rust outbreaks, these early seral communities would probably be dominated by whitebark pine.	This type may contract in the future due to several interacting factors; whitebark pine will continue to decline due to rust and beetle outbreaks, spruce-fir forest may decline due to increased fire and reduced soil water; this site could be replaced by lodgepole-aspen in drier locations; if agencies plant and conduct restoration activities, whitebark pine could make major gains into the increasing burned areas, thereby replacing spruce-fir and limiting the contraction of this type; low-elevation spruce-fir stands are probably going to move towards the western larch/mixed conifer type because of prolonged droughts and increasing temperatures, especially after fires.

Table 6B.1—*Continued.*

Resource concerns				
Landscape heterogeneity	Increased productivity causing accelerated succession; increases in disturbance frequency and extent; highly variable drought intensity and extent; migration of species to new habitats; changes in magnitude, season, and variability of water availability.	Ninety years of fire exclusion coupled with past management activities (e.g., grazing) has reduced landscape heterogeneity.	Landscape heterogeneity is highly susceptible to subtle shifts in climate because it is the reflection of the interaction of vegetation dynamics with disturbance regimes, topography, and land use. Small changes in climate may facilitate large changes in disturbances or vegetation dynamics causing new landscape mosaics.	Increased fire across most of the NR may both increase and decrease landscape heterogeneity. Wildfires and wildland fire use fires may create patchworks of fire severity types across burned areas that will increase heterogeneity and therefore landscape resilience, but some fires may burn fire-excluded landscapes with high severities causing atypical large patches of high plant mortality that may decrease heterogeneity. While large, severely burned patches occurred in historical fires, the frequency and size of these patches may be different today. The highly variable species migration rates into areas with new climates may increase heterogeneity, but the rapidly changing climates may only facilitate generalist species thereby decreasing heterogeneity.
Timber production	Increase in temperature, causing increase in soil moisture deficits and less available water that larch need, will shift species composition to root disease prone species particularly Douglas-fir and grand fir. Higher temperatures likely to extend fire season and to reduce forest inventory on areas suitable for timber production. Some increase in productivity may occur at mid to higher elevations; however, increase in fire may reduce timber production opportunities.	Composition shift causing reduced productivity is likely in the western portion of the region on root disease-prone sites and in southerly exposures. Risk of uncharacteristic fire severity very high due to uncharacteristic high forest density which will reduce timber production opportunities, especially in dry forest areas.	Sensitivity is high in northern Idaho and southerly exposures due to increasing moisture deficits and increase in uncharacteristic disturbance such as severity and extent of fire and root disease.	Expect some increase in production at mid and higher elevations due to warming temperatures. This could be offset overall by losses due to root disease and increase in fire severity across the areas suitable for timber productions. Less production (sequestration) anticipated in northern Idaho if current species compositions are not changed.

Table 6B.1—*Continued.*

Carbon sequestration	Increased fire; increasing drought; productivity gains and losses.	Past policies of fire exclusion have created late-seral landscapes that sequester little carbon; past timber activities may have created younger stands that sequester more carbon; increasing disturbance (fire, insect, disease) has caused short-term losses in carbon sequestration.	Carbon sequestration is very sensitive to climate change impacts on vegetation and disturbance. Rates of carbon sequestration are going to largely be dependent on the rate of burning in the future and the gains and losses of productivity; productivity gains and losses need to be evaluated at large spatial and temporal scales to understand future carbon dynamics.	Fire exclusion will tend to push most ecosystems into later successional stages where sequestration rates are minimal; burning from controlled and uncontrolled wildfires and prescribed burning will cause short-term losses but the high productivity of the developing early-seral stands may increase sequestration for decades; sites that were historically dry will probably experience decreases in production and carbon sequestration in the future, and mesic sites with abundant water (e.g., subalpine, upper subalpine) may experience increases in productivity.

Table 6B.2—Risk assessment and vulnerability ratings for species, vegetation types, and resources of concern.

Habitat, ecosystem function, or species	Adaptive capacity	Exposure	Risk assessment magnitude of effects	Risk assessment likelihood of effects	Northern Rockies vulnerability ranking
Species					
Limber pine	Intermediate adaptive strategy largely driving by timing of pollen cloud dispersal (elevational effect); highly adapted to populating the burned areas predicted for the future due both to wind and corvid-mediated dispersal; poor competitor on more productive sites, if future fires are larger and more severe, there will be less competition from other subalpine conifers; possesses moderate genetic variation (capacity) in blister rust resistance; major gene resistance to blister rust has not been identified in several studies of interior populations, warmer temperatures favor expansion of alternate host species (currant, lousewort and Indian paintbrush); little to no opportunity to hybridize with western white pine due to non-overlapping species distributions, cannot hybridize with whitebark pine; very high risk of loss of disjunct and isolated populations due to genetic drift, ineffective pollen cloud, and substrate availability.	High	Moderate	Moderate	15
Ponderosa pine var. *scopulorum*	Generalist adaptive strategy; high phenotypic plasticity; better adapted to drought.	Moderate	Low	Low	17
Ponderosa pine var. *ponderosa*	Intermediate adaptive strategy at low to mid elevations Specialist adaptive strategy at higher elevations; moderate phenotypic plasticity; less adapted to drought.	Moderate	Moderate	Moderate	16
Douglas-fir	Specialist adaptive strategy at low to mid elevations, generalist adaptive strategy at higher elevations; no opportunity of hybridizing with coastal Douglas-fir subspecies because distributions do not overlap; highly adaptive to a large range of moisture and temperature gradients. In moist forest settings, Douglas-fir is limited to a relatively short-lived seral species due to the influence of two root diseases; With warming temperatures and a possible decrease in summer moisture drought conditions, Rocky Mountain Douglas-fir may increase along with an increase in associated stressors; vulnerable to uncharacteristic fire behavior and severity due to increased densities; increase in susceptibility to Douglas-fir bark beetle mortality uncertain but probably an increased activity; moderate change in species distribution expected away from driest margins. High potential for natural regeneration failure due to reduced seed source from large wildfires and difficult micro climate especially on southerly exposures at lower elevations with increasing moisture deficits expected. On moist sites (mixed mesic forest), increases in root disease mortality due to increasing moisture stress on sites where western white pine, ponderosa pine and larch occurred historically. Less carbon sequestration expected in Douglas-fir in those forest setting; there is a high likelihood of change in local distribution due to moisture deficits and fire severity and a high probability of carbon sequestration in northern Idaho if Douglas-fir remains a dominant species in mixed mesic forests.	High	High	High	5

Table 6B.2—*Continued.*

Western larch	Intermediate adaptive strategy, low capacity to regenerate with increasing moisture deficits on higher energy slopes. Larch is adapted to warm-moist and cool-moist settings. It does best on northerly cool aspects. It is a prolific light seed-cone producer but in sporadic years. High adaptive capacity on lower-energy slopes; wind disperses seed further than many associates, so it can take advantage of newly opened areas due to harvest or fire as an early seral species; larch has few insect and disease stressors unlike associates and is adapted to fairly frequent mixed-severity fire; it can be a dominant, long-lived seral tree species on lower-energy slopes; potential serious reductions in extent of larch on higher energy-slope positions.	High	High	Very High	4
Western white pine	Generalist adaptive strategy, high phenotypic plasticity; cold hardiness may influence distribution; lack of abundance may influence its role in the moist forest ecosystem; especially adapted to future climates in the mesic regions of the NR; there could be major expansions into historical ranges and the subalpine as rust resistance increases in western white pine populations.	Moderate	High	High	3
Quaking aspen	Aspen has the widest distribution of all trees in this report; it is circumpolar across multiple continents; it is highly susceptible to many insect and diseases; with a warmer climate, gypsy moth may have greater impacts and cause possible mortality; aspen distribution may shift upslope or to northeast (cooler, moister) aspects if drought and repeated fire causes mortality on the warmer, drier sites. Riparian aspen communities will likely persist or increase in extent, particularly if the sites remain moist throughout the growing season and increased fire burns the riparian zone, killing conifers. Fire will favor aspen, but prolonged drought will cause mortality. Younger aged stands (<40 years) may be more resilient to drought; long-distance dispersal by light seed may enhance its ability to colonize recently burned areas; this will vary based on winter snowpack, and amount and time of melt (and associated peakflows) but only if there is sufficient moisture.	Moderate	Moderate	High	14
Grand fir	Generalist adaptive strategy, only differences are racial (blue and green race); increases in disease, insects, and fire may reduce populations.	Low	Moderate	Moderate	8
Western redcedar	Generalist adaptive strategy; high phenotypic plasticity; cold/warm cycles early in spring or winter may cause red belt.	Moderate	Moderate	Moderate	6
Western hemlock	Inferred or putative generalist adaptive strategy; information limited concerning western hemlock; susceptible to early frost.	Moderate	Moderate	Moderate	7
Lodgepole pine	Specialist adaptive strategy; especially adapted to occupy post-burn landscapes that may be more common in the future; highly susceptible to increasing bark beetle outbreaks, especially on landscapes dominated by mature individuals. Varying levels of serotiny allow the species to occupy new upper subalpine environments while also regenerating after fire; its intolerance of deep droughts may reduce its capacity along the xeric edges of its current range; Heterogeneity across broad landscapes may mitigate adverse impacts from fire and mountain pine beetles.	Moderate	Moderate	Moderate	11

Table 6B.2—*Continued.*

Subalpine fir	Generalist adaptive strategy; increasing fire will reduce subalpine fir populations to historical levels; fire exclusion may foster subalpine fir encroachment into larch, lodgepole pine, and whitebark pine late seral stands; increasing subalpine temperatures may increase fir growth and accelerate succession toward fir-dominated stands, however, as competition increases, warmer climate may facilitate increased mortality from insects and disease as trees become more stressed from high stem densities.	Low	Moderate	Moderate	10
Engelmann spruce	Intermediate adaptive strategy with opportunities to hybridize with white spruce; hybrids may be more suited to future climates, and hybridization is another key driver in speciation.	Low	Moderate	Moderate	9
Mountain hemlock	Increasing fire will reduce populations, especially where trees are smaller or in dense stands; potential to move to higher elevations; sensitive to low soil moisture in drier locations. Hybridizes with western hemlock to some extent. Should be able to persist across the landscape, perhaps in more scattered patches.	Low	Moderate	Moderate	12
Whitebark pine	Highly adapted to populating the greater burned areas predicted for the future due to bird-mediated dispersal; if future fires are larger, more severe, there will be less competition from other subalpine conifers; ability to survive fire better than its competitors; moderately shade tolerant so it can exist in competition with limited cone crops; delayed germination adaptation may mitigate warmer, drier conditions; possesses moderate to high genetic variation (capacity) in adaptive traits (blister rust resistance, late winter cold hardiness and drought tolerance), as well as phenotypic plasticity to respond to climate change. Warmer temperatures favor expansion of alternate host species (currant, lousewort and Indian paintbrush). No opportunity to hybridize with another stone pine and cannot cross with western white or limber pine where species distributions overlap; high risk and loss of disjunct and isolated populations; more drought tolerant than its associates; long distance bird dispersal will increase regeneration potential as more of the landscape burns.	High	High	High	2
Alpine larch	Its specific habitat requirements may make it difficult for alpine larch to remain on the landscape over the long term; short-term gains in alpine larch encroachment in upper subalpine and treeline glades and meadows may be lost in those years with deep drought; effective, long-term establishment of alpine larch may depend on the ability of seed dispersal to find those areas with sufficient moisture to maintain the species.	High	High	High	1

Table 6B.2—*Continued.*

Green ash	Since green ash communities are already fire adapted (most associated species display some fire tolerance and/or postfire sprouting ability), increased fire will likely not affect most of the moister communities. However, those communities associated with either ephemeral drainages (e.g., woody draws) or moist upland microsites (e.g., northeast-facing depressions) may experience more drought stress as snowpack declines and melts sooner, and regeneration may decrease, eventually resulting in loss of those communities.	Moderate	Moderate	High	18
Cottonwood	Plains cottonwood may be more persistent due to greater plant available soil water in the unsaturated zone (as a result of finer textured soils). Black and narrowleaf cottonwood typically occur in coarser substrate, which will become more droughty as flows are lower and recede earlier than in the past, or are attenuated due to diversions. Seedling and sapling mortality may increase in these species. Plains cottonwood regeneration occurs with episodic flooding, whereas black and narrowleaf cottonwood typically regenerate with 1–3 year bankfull flow return intervals; therefore, plains cottonwood will likely be more adapted to irregular flows (in timing, magnitude and duration) that may occur with climate change.	Moderate	Moderate	Moderate to High	13
Vegetation Types					
Dry ponderosa pine and Douglas-fir forests	Douglas-fir highly adaptive to a large range of moisture and temperature gradients. Ponderosa pine adapted to settings that are moisture limited and can grow well where moisture is less limited such in association currently with grand fir; exposure of Douglas-fir to increasing moisture deficits may change composition to more ponderosa pine; increasing moisture deficits will give ponderosa pine the advantage on dry forest settings due to fire, insect and disease.	High	High	High	3
Western larch mixed conifer forests	Larch forests thrive on northerly cool aspects locations in the Region; it is a prolific light seed cone producer but in sporadic years, may seed longer distances than many associates, so can take advance of newly opened areas due to harvest or fire as an early-seral species; larch has few insect and disease stressors unlike associates, and is adapted to fairly frequent fire; it can out-dominate long lived seral tree species, and due to the large diameters it can attain, is important to many cavity nesters as habitat when it reaches ages over 200 years in patches large enough and connected enough to provide functional habitat; exposure on southerly aspects; warming temperatures and possible decreased summer moisture and associated drought may decrease distribution of western larch, which could mean that it will retreat to low-energy northerly slope settings; distribution of larch in patches on high-energy slopes would be reduced significantly; serious reduction in extent of larch in patches on higher-energy slope positions in the long run.	High	High	Very High	2

Table 6B.2—*Continued*.

Lodgepole pine and aspen mixed conifer forests	This type has the capacity to absorb climate changes and either remain constant or expand into the upper subalpine; losses in aspen due to drought may be offset by gains in lodgepole pine, especially after fire; there may be long-term migrations of this type to higher elevation areas with increasing disturbance.	High	Moderate	High	4
Mixed mesic white pine, cedar, hemlock grand fir forests	This type may have the capacity to remain intact with changing climates; it may not be able to expand due to ash cap; increasing fire will favor western white pine, and fire exclusion will favor the shade-tolerant species; drier sites may see grand fir becoming more common than cedar or hemlock.	Low	Moderate	Low	5
Whitebark pine-spruce-fir forests	This type may have the capacity to respond favorably to changes in climate, but depressed populations of whitebark pine coupled with increasing fire may result in short-term losses of this type; however, if rust-resistant whitebark pine are planted and restoration activities are implemented, whitebark pine can easily dominate on these sites, especially if fires are large and severe, and whitebark pine may be able to make advances into the treeline; continued fire exclusion will probably aid in keeping this type somewhat static, and it may encroach on lower treeline sites if no fires are allowed	High	High	High	1
Resource Concerns					
Landscape heterogeneity	Because heterogeneity is an expression of disturbance, vegetation, and climate interactions, it depends on other factors to determine its adaptive capacity.	High	Moderate	High	NA
Timber production	Productivity could increase at higher elevation sites. Productivity in northern Idaho will likely decrease on southerly aspects due to root disease reducing productivity of alternate species unless western larch, ponderosa pine and western white pine are aggressively restored; high exposure due to species composition changes and risk to increased disturbance.	High	Moderate to high in north Idaho	High in north Idaho	NA
Carbon sequestration	All ecosystems have an inherent capability to store carbon, and the rate and capacity of carbon storage depends on plant productivity and disturbance; maximum levels of productivity depend on climate, whereas the instantaneous levels of productivity depend on successional stage or time since disturbance.Modeling studies have shown that many areas will increase productivity and increase sequestration rate and magnitude; the delicate balance between disturbance and climate coupled with land management will dictate where sequestration will increase and where it will decrease; it is important to know that over the long term (centuries), sequestration is near zero (disturbance and respiration losses are balanced by productivity gains).	High	High	Moderate	NA

Appendix 6C—Adaptation Options for Forest Vegetation in the Northern Rockies.

The following tables describe climate change sensitivities and adaptation strategies and tactics for forest vegetation, developed in a series of workshops as a part of the Northern Rockies Adaptation Partnership. Tables are organized by subregion within the Northern Rockies. See Chapter 6 for summary tables and discussion of adaptation options for forest vegetation.

Table 6C.1—Adaptation options that address climate change effects on forest vegetation in the Central Rockies subregion.

Sensitivity to climatic variability and change: Aspen will be lost with increasing drought and lack of disturbance.

Adaptation strategy/approach: Increase aspen populations.

Strategy objective: Increase aspen populations.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**	**Specific tactic – D**
Tactic	Actively decrease impact of ungulate browsing through an increase in predation of ungulates.	Increase fire disturbance.	Develop techniques to successfully artificially regenerate aspen; improve drought tolerance with genetic selection of drought-tolerant mother trees and identify and target areas most likely to successfully regenerate or provide favorable planting habitat.	Protect existing aspen populations and regeneration treatments from browsing.
Tactic effectiveness (risks)	High where ungulates concentrate			
Implementation urgency				
Where can tactics be applied? (geographic)				
Opportunities for implementation				
Cost				
Barriers to implementation				

Table 6C.2—Adaptation options that address climate change effects on forest vegetation in the Central Rockies subregion.

Sensitivity to climatic variability and change: Known unknowns and unknown unknowns.			
Adaptation strategy/approach: Increase knowledge and implement adaptive management.			
Strategy objective: Increase knowledge.			
	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Monitor blister rust resistance within planted white pine stands, and try to understand the relationship between infection rates and climatic/weather drivers like fog.	Identify other resource management goals (not directly related to stand structure and composition) that may modify management strategies for forest vegetation, such as water yield, snow retention, and wildlife habitat.	Support the adaptive management research framework through a partnership between Region 1 and Rocky Mountain Research Station to evaluate management treatments and how they contribute to resiliency. Develop monitoring framework that will be consistently be implemented so that long-term change can be captured.
Tactic effectiveness (risks)			
Implementation urgency			
Where can tactics be applied? (geographic)			
Opportunities for implementation			
Cost			
Barriers to implementation			

Table 6C.3—Adaptation options that address climate change effects on forest vegetation in the Central Rockies subregion.

Sensitivity to climatic variability and change: Potential shifts in lodgepole pine ecosystems with changing climate.

Adaptation strategy/approach: Promote resilience by maintaining age-size class composition at the stand and landscape level.

Strategy objective: Maintain lodgepole pine on the landscape.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Identify areas appropriate for wildfire use and increase flexibility in how we manage fire; emphasize modified suppression and resource benefit fire; allow moderate season fires to burn.	Proactively treat stands with prescribed fire to reduce fire and insect mortality and increase individual tree vigor.	Implement silvicultural mechanical treatments.
Tactic effectiveness (risks)	Other unintended consequences could lower it from high to moderate.	Highly variable depending on scale, district expertise, and landscape position.	Highly variable depending on scale.
Implementation urgency	High/near term		
Where can tactics be applied? (geographic)	Implement Wildland Fire Decision Support System forest wide		
Opportunities for implementation	Highly dependent on other resource concerns and political and social issues.		
Cost			
Barriers to implementation	Integration between forest and fire team	Prescribed fire air quality restrictions limit what we can do; other resource constraints (e.g., lynx)	Other resource constraints (e.g., thermal cover, lynx habitat, marten and wolverine)

Table 6C.4—Adaptation options that address climate change effects on forest vegetation in the Central Rockies subregion.

Sensitivity to climatic variability and change: Homogenization of the ponderosa forest type across the landscape resulting in: increase in density causing associated increase in risk of severe stand replacing fires; increased risk of mortality from drought; loss of ponderosa pine on stressful sites; loss of large ponderosa pine and large ponderosa snag recruitment; and reduced water yield.

Adaptation strategy/approach: Decrease the density within ponderosa pine-Douglas-fir stands, and increase structural.

Strategy Objective: Develop resilience to fire, moisture and stress.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Reduce stand density with thinning, prescribed fire, and wildland fire use, with density and structural goals based on past and predicted future conditions.	Promote age class and structural diversity across the landscape, through regeneration harvest, thinning, prescribed fire, and wildland fire use.	Monitor establishment, survival and development of ponderosa by age class and in different conditions (e.g., aspect, heat load and soil moisture) using Forest Inventory and Analysis data and project-level stocking exams.
Tactic effectiveness (risks)	Thinning = high, prescribed fire = high, wildland fire use = moderate	High where implemented	High
Implementation urgency	Near-term in wildland urban interface areas. Mid term elsewhere	Near-term in wildland urban interface areas. Mid term elsewhere	Long term
Where can tactics be applied? (geographic)	Ponderosa pine – Douglas-fir forests	Harvesting and thinning on non-reserved lands; prescribed fire and managed wildfire where approved.	Ponderosa pine – Douglas-fir forests
Opportunities for implementation	Work with collaborators on private and state land to include the wildland urban interface; work with other organizations to get funding and do work on private lands.	Work with collaborators on private and state land to include the wildland urban interface; work with other organizations to get funding and do work on private lands.	Forest Inventory and Analysis; restoration and resiliency report; forest regeneration reports; adapting Common Stand Exam protocols.
Cost	Varies by treatment	Varies by treatment	Inexpensive
Barriers to implementation		Some: cost, litigation	None

Table 6C.5—Adaptation options that address climate change effects on forest vegetation in the Central Rockies subregion.

Sensitivity to climatic variability and change: Changing moisture regimes with changing climate.

Adaptation strategy/approach: Replace plant association group-habitat typing with an index based on biophysical variables.

Strategy objective: Increase knowledge; plant associations are not expected to remain the same under climate change.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Identify a set of biophysical predictors related to habitat types, site productivity, vegetation composition, and structure. Possible predictors include landform, soil depth, texture, type (specifically ash cap soil), actual evapotranspiration, potential evapotranspiration and water balance deficit.	Predict site productivity based on biophysical predictors; make concept operationally implementable so it can be used to aid in planting decisions, and aid understanding of long-term effects of management and long-term goals for a site.	Project into the future based on climate change models.
Tactic effectiveness (risks)	Unknown	Unknown	Unknown
Implementation urgency	Near term	Near term	Near term
Where can tactics be applied? (geographic)			
Opportunities for implementation			
Cost			
Barriers to implementation			

Table 6C.6—Adaptation options that address climate change effects on forest vegetation in the Central Rockies subregion.

Sensitivity to climatic variability and change: Subalpine forest shifts as a result of climate change.		
Adaptation strategy/approach: Monitor and detect change through measurement of seedling survival, species composition and mortality of mature trees.		
Strategy objective: Increase knowledge on climate-induced subalpine forest shifts.		
	Specific tactic – A	**Specific tactic – B**
Tactic	Install and analyze additional plots where trend information is needed over time; more intensified grid plots could also be installed.	Initiate the use of Forest Inventory and Analysis plot information.
Tactic effectiveness (risks)	High	Moderate to low, depending on how fast changes are occurring
Implementation urgency	High—need to know now	Already part of regional strategy
Where can tactics be applied? (geographic)	Target areas of risk where we expect to see changes, focusing on disturbance areas; in especially sensitive forests such as whitebark pine	Large enough geographic area that information can be statistically meaningful
Opportunities for implementation	Very few because of lack of funding, but protocols and analysis tools already established	Already paid for, and existing expertise and knowledge are available
Cost	High based on information needs; many plots needed	Minimal
Barriers to implementation	Need training and oversight of data collection; need to have consistent, replicable protocol in order to be able to detect change; lack of immediate and long term funding	Difficult to get data updates in a timely manner; getting non-Forest Service data is difficult

Table 6C.6 (cont.)—Adaptation options that address climate change effects on forest vegetation in the Central Rockies subregion.

Sensitivity to climatic variability and change: Subalpine forest shifts as a result of climate change.

Adaptation Strategy / Approach: Monitor and detect change through measurement of seedling survival, species composition and mortality of mature trees.

Strategy Objective: Increase knowledge on climate-induced subalpine forest shifts.

	Specific Tactic – C	**Specific Tactic – D**
Tactic	Expand reforestation monitoring to include additional monitoring and different electronic formats.	Implement pre- and post-treatment monitoring over time to be able to determine if treatments are meeting objectives.
Tactic effectiveness (risks)	High	High
Implementation urgency	High	Moderate
Where can tactics be applied? (geographic)	Everywhere we are planting sensitive vegetation types	Treated areas
Opportunities for implementation	Often doing it anyway, and methodology is already in place; need to make sure we are collecting right data in the right electronic format	Existing systems
Cost	Low unless data collection becomes arduous	Moderate to high; depends on number of plots, length of time to monitor, what data is being collected; less expensive than intensifying grid across landscape.
Barriers to implementation	Need to establish protocols, conduct training, identify a location for data storage and determine who would do the analysis; lack of immediate and long-term funding	Lack of immediate and long-term funding; lack of training and oversight; lack of treatments to monitor

Table 6C.7—Adaptation options that address climate change effects on forest vegetation in the Central Rockies subregion.

Sensitivity to climatic variability and change: Subalpine forest shifts as a result of climate change.

Adaptation strategy/approach: Promote resilience of subalpine forests to climate-induced shifts.

Strategy objective: Create conditions that will be resilient to perturbations.

	Specific tactic – A	**Specific tactic – B**
Tactic	Identify areas appropriate for managed wildfire; increase the flexibility of how we manage fire and de-emphasize suppression.	Artificially regenerate seedlings with higher levels of rust resistance than the natural population.
Tactic effectiveness (risks)	Other unintended consequences could lower it from high to moderate	High at site scale; low across large landscapes
Implementation urgency	High, near term	High, near term
Where can tactics be applied? (geographic)	Highly dependent on other resource concerns and political and social issues; problematic in areas with mixed ownership	Have process established to develop rust-resistant seedlings; coordinate with tree improvement and nursery to continue to improve rust-resistant sources
Opportunities for implementation	All lands: Federal, State, communities, universities	All lands: Federal, State, communities, universities
Cost	Inexpensive	Moderately expensive; cost of seedlings high
Barriers to implementation	Integration between forest and fire team; sociopolitical environment and backlash; lack of public support; decline in air quality	Access; cost; unable to plant in wilderness or research natural areas

Table 6C.7 (cont.)—Adaptation options that address climate change effects on forest vegetation in the Central subregion.

Sensitivity to climatic variability and change: Subalpine forest shifts as a result of climate change.

Adaptation Strategy / Approach: Promote resilience of subalpine forests to climate-induced shifts.

Strategy Objective: Create conditions that will be resilient to perturbations.

	Specific Tactic – C	**Specific Tactic – D**	**Specific Tactic – E**
Tactic	Proactively treat with prescribed burn or mechanical thinning to reduce the risk of fire and insect mortality; increase individual tree vigor to establish the desired species composition and establish appropriate patch sizes.	Implement a broad review of most current literature, develop a regional library, and implement a climate change information system.	Educate the public on the need to increase resilience and develop a cohesive broader message.
Tactic effectiveness (risks)	Highly variable depending on scale, district expertise, and landscape position	High if used	Low
Implementation urgency	Moderate-mid term	High, near term	Low, long term
Where can tactics be applied? (geographic)		All	National
Opportunities for implementation	All lands: Federal, State, communities, universities		National
Cost	Low to high depending on the combination of treatment, objectives, and local conditions		
Barriers to implementation	This type is not far from historic conditions, so there is not as much reason to restore; more complex and difficult to access; window for treatment is narrow		

Table 6C.8—Adaptation options that address climate change effects on forest vegetation in the Central Rockies subregion.

Sensitivity to climatic variability and change: Homogenization of the larch forest type across the landscape, resulting in increased density and associated increase risk of severe stand-replacing fires, increased risk of mortality from drought, loss of western larch on stressful sites, loss of large larch, large larch snag recruitment across the landscape, and reduced water yield.

Adaptation strategy/approach: Decrease density within stands, and increase structural diversity across the landscape.

Strategy objective: Develop resilience to fire and moisture stress.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Reduce density with thinning, prescribed fire, and wildland fire use.	Promote age class and structural diversity across the landscape, through regeneration harvest, thinning, prescribed fire and managed wildfire.	Monitor establishment and survival of western larch by age class across different aspects/heat load/soil moisture. Use Forest Inventory and Analysis data to capture regeneration success by topographical position.
Tactic effectiveness (risks)	Thinning = high; prescribed fire = moderate; wildland fire use = low when first implemented, moderate over time	High where implemented	High
Implementation urgency			
Where can tactics be applied? (geographic)			
Opportunities for implementation			
Cost			
Barriers to implementation			

Table 6C.8 (cont.)—Adaptation options that address climate change effects on forest vegetation in the Central subregion.

Sensitivity to climatic variability and change: Homogenization of the larch forest type across the landscape resulting in increased density and associated increase risk of severe stand-replacing fires, increased risk of mortality from drought, loss of western larch on stressful sites, loss of large larch, large larch snag recruitment across the landscape, and reduced water yield.

Adaptation Strategy / Approach: Decrease density within stands, and increase structural diversity across the landscape.

Strategy Objective: Develop resilience to fire and moisture stress.

	Specific Tactic – D	**Specific Tactic – E**
Tactic	Prioritize management for larch on landscape facets where monitoring indicates it is going to persist (e.g., on north aspects, but not southern aspects, or by habitat types).	Maintain and promote large diameter western larch across the landscape, so that large diameter snags, larch seed sources, and wildlife habitats are also maintained.
Tactic effectiveness (risks)	High	High if A and B are implemented
Implementation urgency		
Where can tactics be applied? (geographic)		
Opportunities for implementation		
Cost		
Barriers to implementation		

Table 6C.9—Adaptation options that address climate change effects on forest vegetation in the Central Rockies subregion.

Sensitivity to climatic variability and change: High density of mixed-mesic white pine, cedar, hemlock, grand fir forests stands, leading to lower carbon sequestration because of lower density of some species (western larch, ponderosa pine and western white pine) and increased shade-tolerant species. High densities increase susceptibility to wildfire. Heart rot and root rot in cedar and hemlock decreases carbon sequestration and timber values.

Adaptation strategy/approach: Promote diversity in species composition and structure at multiple spatial scales (within stands and across the landscape).

Strategy objective: Promote resilience and increase carbon sequestration.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**	**Specific tactic – D**	**Specific tactic – E**
Tactic	Conduct regeneration harvest and planting to promote shade-intolerant tree species (western larch, ponderosa pine, white pine, birch) and rust-resistant white pine.	Promote western white pine.	Conduct density management thinning to retain shade-intolerant trees and develop large trees across the landscape; prune younger western white pine to promote resistance to blister rust.	Increase knowledge on how the different tree densities and species compositions within this group function as habitats and how the mosaic of stands influences connectivity for wildlife.	Develop a site-related prioritization approach for implementing management tactics.
Tactic effectiveness (risks)	High	High	High	Unknown	High
Implementation urgency	Mid-term				
Where can tactics be applied? (geographic)	Within this vegetation type				
Opportunities for implementation					
Cost					
Barriers to implementation					

Table 6C.10—Adaptation options that address climate change effects on forest vegetation in the Central Rockies subregion.

Sensitivity to climatic variability and change: Whitebark pine mortality from increased mountain pine beetle outbreaks, fire, and blister rust and shifts/loss of site conditions that support whitebark pine.

Adaptation strategy/approach: Restore whitebark pine.

Strategy objective: Increase rust-resistant regeneration and cone-bearing trees.

	Specific Tactic – A	**Specific Tactic – B**
Tactic	Assess condition; save seed sources; grow rust-resistant seedlings; collect seed; plant seedlings; monitor activities; prioritize treatment; and conduct research (mirror Keane range-wide restoration strategy).	Identify sites that are likely to not be as heavily affected (refugia), and focus on those sites for restoration.
Tactic effectiveness (risks)		
Implementation urgency		
Where can tactics be applied? (geographic)		
Opportunities for implementation		
Cost	Cost of planting is high	
Barriers to implementation	The majority of whitebark pine is in wilderness or other protected areas, or in roadless areas with limited access. Thus, the ability to treat directly is limited. Need comprehensive long-term planning and commitment at multiple levels.	

Table 6C.11—Adaptation options that address climate change effects on forest vegetation in the Eastern Rockies subregion.

Sensitivity to climatic variability and change: Increased frequency of disturbance, drought, browsing, and livestock trampling may reduce vigor and extent of aspen, even given the upward trend of aspen in pine forests that have extensive beetle-caused mortality.

Adaptation strategy/approach: Maintain and restore/promote the health and vigor of clones (specific key stressors include repeat disturbance, conifer competition, browse/trampling).

Strategy objective: Maintain the current population trend, promote resilience, reduce stressors/threats, and increase knowledge.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Actively manage ungulates during regeneration phase through slash barriers, increase in hunting pressure, grazing management, and salt block placement.	Implement conifer removal on multiple scales and through hand and commercial treatments.	Monitor extent and condition of aspen clones to determine the effect of disturbance frequency on aspen survival, effects of sudden aspen decline, and incidence of new clones; potentially implement strategic protection measures for specific clones.
Tactic effectiveness (risks)	High	Highest	Moderate to high
Implementation urgency	Near term	Near term	Near term
Where can tactics be applied? (geographic)	Wherever aspen regeneration prescriptions are applied	Wherever aspen occurs outside of protected areas	Wherever aspen occurs
Opportunities for implementation	Rocky Mountain Elk Foundation, permittees	Rocky Mountain Elk Foundation	Rocky Mountain Elk Foundation
Cost	Varies by tool Expensive – fencing Inexpensive – grazing management	Inexpensive	Moderate
Barriers to implementation	Some – litigation, scale and timing to be effective (staffing capacity), priority, permittees	Some – litigation, scale and timing to be effective (staffing capacity), priority	None

Table 6C.12—Adaptation options that address climate change effects on forest vegetation in the Eastern Rockies subregion.

Sensitivity to climatic variability and change: Ability of Douglas-fir to develop stands with large, old-growth structure in face of fire, drought, beetles, spruce budworm and limited cone production.

Adaptation strategy/approach: Establish age-class and structural diversity to promote landscape heterogeneity, which could provide appropriate amounts and distribution of large tree structure.

Strategy objective: Promote resilience, and reduce stressors and threats.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**	**Specific tactic – D**
Tactic	Manage stand-scale disturbance through thinning, prescribed fire, and managed wildfire to accelerate the development of large tree structure.	Manage landscape disturbance to achieve heterogeneity in structural diversity.	Identify critical stands with high value and large diameter for direct protection from fire and insects; protect using pesticides, stand manipulation, fire protection, and windthrow monitoring.	Improve cone production in key seed collection areas through fertilizer, pesticides and stand manipulation.
Tactic effectiveness (risks)	High – thinning Moderate – fire	Moderate–high	High	Moderate
Implementation urgency	Near–mid term	Near–mid term	Near term	Near term
Where can tactics be applied? (geographic)	Throughout species range	Throughout species range	Throughout species range	Throughout species range
Opportunities for implementation	Forest plan; restoration committees	Forest plan; restoration committees	Coordinate with wildlife priorities	Inland Empire Cooperatives
Cost	Inexpensive	Inexpensive	Inexpensive	Inexpensive–moderate
Barriers to implementation	Some – litigation, scale and timing to be effective (staffing capacity), priority	Some – litigation, scale and timing to be effective (staffing capacity), priority	None	None

Table 6C.13—Adaptation options that address climate change effects on forest vegetation in the Eastern Rockies subregion.

Sensitivity to climatic variability and change: Limited ability of limber pine to persist and regenerate in the face of frequent fires, blister rust, and mountain pine beetle; challenges include uncertainties in genetic variation and geographically disjunct populations.

Adaptation strategy/approach: Maintain viable populations of limber pine in suitable environments.

Strategy objective: Promote resilience, reduce impacts of stressors/threat, increase knowledge, engage coordination.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Address information gaps on current location, potential habitat and stand condition through coordination with other groups.	Address genetic data gaps by establishing a breeding program that includes rust-resistance screening, seed zones and drought tolerance through coordination with other groups.	Conduct stand treatments to promote resilience over a wide geographic area; protect remaining trees with pesticides and stand thinning, and establish a planting program to achieve scattered, mature individuals.
Tactic effectiveness (risks)	High	High	High at local stand scale Moderate at population level
Implementation urgency	Near term	Near term	Near term: improve resilience of existing stands Mid term: planting (answer genetic questions first)
Where can tactics be applied? (geographic)	Forest plans, Region 1 broad-scale monitoring strategy	Inland Empire Cooperatives	Forest plans, restoration committees
Opportunities for implementation	All lands: Federal, State, communities, universities	All lands: Federal, State, communities, universities	All lands: Federal, State, communities, universities
Cost	Moderate	Moderate	Inexpensive
Barriers to implementation	None	None	Some – litigation, scale and timing to be effective (staffing capacity), priority

Table 6C.14—Adaptation options that address climate change effects on forest vegetation in the Eastern Rockies subregion.

Sensitivity to climatic variability and change: Small, fragmented populations of eastside ponderosa pine are vulnerable to loss of regeneration capability following repeated disturbance, especially in ecotone sites.

Adaptation strategy/approach: Maintain genetic diversity, population viability, increase resilience, and promote big-tree structure.

Strategy objective: Promote resilience, reduce stressors/threats, and facilitate transition.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Manage stand-scale disturbance through thinning, prescribed fire, and managed wildfire to accelerate the development of large tree structure and facilitate the transition to a savanna ecotone.	Develop seed collection and reforestation strategies to address potential loss of genetic variability due to disjunct populations and shortened mean return interval of severe disturbances.	Identify critical stands with high value and large diameter for direct protection from fire and insects; protect with pesticides, stand manipulation and fire protection.
Tactic effectiveness (risks)	High	High	High
Implementation urgency	Near term	Near term	Near term
Where can tactics be applied? (geographic)	Low-elevation ecotones; suitable ponderosa pine sites	Low-elevation ecotones; suitable ponderosa pine sites	Low-elevation ecotones; suitable ponderosa pine sites
Opportunities for implementation	Forest management; coordination with fire management	Forest management; Inland Empire Cooperatives	Forest management; wildlife habitat priorities
Cost	Inexpensive	Inexpensive-moderate	Inexpensive
Barriers to implementation	Some – litigation, scale and timing to be effective (staffing capacity), priority	None	Some – litigation, scale and timing to be effective (staffing capacity), priority

Table 6C.15—Adaptation options that address climate change effects on forest vegetation in the Greater Yellowstone Area subregion.

Sensitivity to climatic variability and change: Climate change forces us to work across boundaries. Stressors cross boundaries. The public expects coordination.

Adaptation strategy/approach: Work across jurisdictions at larger scales.

Strategy objective: Encourage coordination.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Align budgets/priorities for program of work for neighboring lands (e.g., constrained budget forces cooperation for invasive species strategies).	Communicate about adjacent projects and coordinate on the ground; develop memorandum of understanding for adjacent projects.	Preserve roads, trails, and access despite increased fire and flood events.
Tactic effectiveness (risks)	High (if we succeed)	High	High
Implementation urgency	Near term	Near term	Near term
Where can tactics be applied? (geographic)	Greater Yellowstone Area	Greater Yellowstone Area	Greater Yellowstone Area
Opportunities for implementation	All lands: Federal, State, communities, universities	All lands: Federal, State, communities, universities	All lands: Federal, State, communities, universities
Cost	Inexpensive	Inexpensive	Moderately expensive
Barriers to implementation	Major barriers: politics	Some barriers: politics, time, lack of priority, distance, travel budget	Major barriers: shrinking budget, staffing

Table 6C.16—Adaptation options that address climate change effects on forest vegetation in the Greater Yellowstone Area subregion.

Sensitivity to climatic variability and change: Increased fire and disease, and interactions with the wildland-urban interface.

Adaptation strategy/approach: Increase the resilience of forest stands to disturbance, and protect people from potential risks of increased fire.

Strategy objective: Manage fires in a way to protect the wildland-urban interface.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**	**Specific tactic – D**
Tactic	Create buffer zones between fire and residential development, and implement thinning.	Influence development zoning in high risk areas with local government and insurance companies.	Increase forest diversity through heterogeneity of species composition, age class and structure.	Promote education and communication about responsible land owner tactics.
Tactic effectiveness (risks)	Moderate – intense fires cross boundaries, having unintended impacts on resources and cultural landscapes	High (if we can do it)	Moderate – large fires, climate change	Moderate
Implementation urgency				
Where can tactics be applied? (geographic)				
Opportunities for implementation				
Cost				
Barriers to implementation	Land designation			

Table 6C.17—Adaptation options that address climate change effects on forest vegetation in the Greater Yellowstone Area subregion.

Sensitivity to climatic variability and change: Increased impacts from fire, insects, disease, climate, and increased competition.

Adaptation strategy/approach: Promote resilient whitebark pine communities.

Strategy objective: Maintain whitebark pine on the landscape.

	Specific tactic – A	Specific tactic – B	Specific tactic – C	Specific tactic – D
Tactic	Actively remove other dominant tree species, such as lodgepole pine, spruce, and fir.	Prioritize management for populations with high survival potential.	Implement a variety of management strategies/options, such as control sites, fire management, and possibly planting at lower elevations.	Replant screened, grafted, genetically-selected seedlings to promote blister rust resistance.
Tactic effectiveness (risks)	Unknown – Not sure if it will work. See barriers.	Moderate – uncertainty, models not considering all factors	Moderate – see barriers	Moderate – see barriers
Implementation urgency	Near term	Near term to long term – more information needed	Near term	Near term
Where can tactics be applied? (geographic)	Where we have access; road, management designation	Greater Yellowstone Area	Greater Yellowstone Area	Where we have access and permission
Opportunities for implementation	Overall whitebark pine strategies, investment strategy niches, Greater Yellowstone Coordinating Committee (GYCC)	Overall whitebark pine strategies, investment strategy niches, GYCC	Overall whitebark pine strategies, investment strategy niches, GYCC	We already have the trees and selected planting locations
Cost	Moderately expensive	Inexpensive	Expensive	Moderately expensive
Barriers to implementation	Major barriers: limited by wilderness designations, road access, litigation, compliance	Major barriers: limited by wilderness designations, road access, litigation, cost	Major barriers: limited by wilderness designations, road access, litigation, cost	Some barriers: ideal places, access, cost

Table 6C.18—Adaptation options that address climate change effects on forest vegetation in the Western Rockies subregion.

Sensitivity to climatic variability and change: Increased impacts from moisture stress and fire.			
Adaptation strategy/approach: Actively reduce density and maintain low densities.			
Strategy objective: Promote forest resilience through density management.			
	Specific tactic – A	**Specific tactic – B**	**Specific Tactic – C**
Tactic	Implement pre-commercial and commercial thinning.	Conduct prescribed fires.	Influence stand density with planting.
Tactic effectiveness (risks)	High	Moderate – depends on stand conditions	High
Implementation urgency	Near term	Near term	Near term
Where can tactics be applied? (geographic)	Be cautious with root disease-sensitive species and sites	Applied to maintain fire-tolerant conifer species and maintaining structure	Apply where opportunities for thinning are negligible due to species or stand condition
Opportunities for implementation			
Cost	Inexpensive	Inexpensive	Inexpensive
Barriers to implementation	Some, barriers greater for Federal lands. There are thinning restrictions due to lynx.	Some, barriers greater for Federal lands	Some, barriers greater for Federal lands

Table 6C.19—Adaptation options that address climate change effects on forest vegetation in the Western Rockies subregion.

Sensitivity to climatic variability and change: Increased frequency and scale of disturbances such as fire and drought.

Adaptation strategy/approach: Promote disturbance-resilient species, such as ponderosa pine, western larch, western white pine, Douglas-fir, and lodgepole pine.

Strategy objective: Promote forest vegetation resilience to disturbance.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Promote thinning to favor disturbance-resilient species.	Promote planting of disturbance-resilient species.	Promote resilient species with prescribed fire and/or natural fire use.
Tactic effectiveness (risks)	High	High	Moderate
Implementation urgency	Near term	Near term	Near term
Where can tactics be applied? (geographic)	Western larch and western white pine on moist sites, ponderosa pine on dry sites, Douglas-fir on extremely dry sites, and lodgepole pine on harsh, difficult-to-regenerate sites	Western larch and western white pine on moist sites, ponderosa pine on dry sites, Douglas-fir on extremely dry sites, and lodgepole pine on harsh, difficult-to-regenerate sites	Western larch and western white pine on moist sites, ponderosa pine on dry sites, Douglas-fir on extremely dry sites, and lodgepole pine on harsh, difficult-to-regenerate sites
Opportunities for implementation	Any place management is feasible and allowed	Any place management is feasible and allowed	Any place management is feasible and allowed
Cost	Inexpensive	Inexpensive	Inexpensive
Barriers to implementation	Scale will vary because of budgetary limitations	Scale will vary because of budgetary limitations	Scale will vary because of budgetary limitations. Natural fire may not occur where or when manageable.

Table 6C.20—Adaptation options that address climate change effects on forest vegetation in the Western Rockies subregion.

Sensitivity to climatic variability and change: Increasing disturbance, moisture deficits, stress due to densities, and longer growing seasons.

Adaptation strategy/approach: Increase knowledge on current issues, strategies, and future goals.

Strategy objective: Increase knowledge for agency land managers and stakeholders.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Address definition of connectivity, including species guilds and scale, and identify existing and historical conditions in order to develop desired conditions.	Share knowledge on tree species migration at fine and broad scales, and track regeneration success, species distribution at the fine scale, species transfer zones, habitat type, and soil type.	Share knowledge on fire regimes in spruce-fir forests and increase collaboration with the Rocky Mountain Research Station.
Tactic effectiveness (risks)	High	High	High
Implementation urgency	Near term	Near term	Near term
Where can tactics be applied? (geographic)	Landscape	Landscape	Landscape
Opportunities for implementation	All lands: Federal, State, communities, and universities	All lands: Federal, State, communities, and universities	All lands: Federal, State, communities, and universities
Cost	Inexpensive	Inexpensive	Moderately expensive
Barriers to implementation	Funding, lack of understanding	Funding, lack of understanding	Funding, lack of understanding

Table 6C.21—Adaptation options that address climate change effects on forest vegetation in the Western Rockies subregion.

Sensitivity to climatic variability and change: Increasing disturbance, moisture deficits, stress due to densities, and longer growing seasons.

Adaptation strategy/approach: Increase knowledge on current issues, strategies, and future goals.

Strategy objective: Increase knowledge for agency land managers and stakeholders.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Address how to manage conservation areas, such as research natural areas and roadless wilderness. Promote the important components within these areas and desired conditions.	Communicate the accurate mapping of important species such as whitebark pine.	Address the desired conditions at the landscape level and also for important wildlife habitat by tackling the "how much is enough?" question.
Tactic effectiveness (risks)	Moderate	High	High
Implementation urgency	Mid term	Near term	Near term
Where can tactics be applied? (geographic)	Program level	Program and national forest level	Program and national forest level
Opportunities for implementation	Rocky Mountain Research Station, The Nature Conservancy, Wilderness Society, universities, tribes, conservation partners	Universities, conservation partners, state heritage programs	Rocky Mountain Research Station, universities, tribes, conservation partners
Cost	Moderate	Moderate	Moderate
Barriers to implementation	Time	Time	Time

Table 6C.22—Adaptation options that address climate change effects on forest vegetation in the Western Rockies subregion.

Sensitivity to climatic variability and change: Increasing disturbance, moisture deficits, stress due to densities, and longer growing seasons.		
Adaptation strategy/approach: Increase knowledge on current issues, strategies, and future goals.		
Strategy objective: Increase knowledge for agency land managers and stakeholders.		
	Specific tactic – A	**Specific tactic – B**
Tactic	Increase awareness on the role of root disease in carbon sequestration by publishing root disease data.	Communicate the need for improved integration between wildlife managers and forest ecologists, and between research and management. Conduct an annual meeting with Rocky Mountain Research Station on management needs for research.
Tactic effectiveness (risks)		High
Implementation urgency		Near term
Where can tactics be applied? (geographic)		Program/Region 1 and Rocky Mountain Research Station
Opportunities for implementation		Program/Region 1 and Rocky Mountain Research Station
Cost		Inexpensive
Barriers to implementation		Lack of time and impetus to do it

Table 6C.23—Adaptation options that address climate change effects on forest vegetation in the Western Rockies subregion.

Sensitivity to climatic variability and change: Increasing disturbance, moisture deficits, stress due to densities, and longer growing seasons.

Adaptation strategy/spproach: Promote resilience in forest vegetation.

Strategy objective: Promote resilience by reducing stressors and facilitating transitions.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Reduce forest densities to increase fire resilience in lodgepole pine and whitebark pine sites with pre-commercial thinning.	Reduce forest densities to increase fire resilience in lodgepole pine and whitebark pine sites with commercial thinning.	Promote appropriate density conditions and landscape heterogeneity within mapped lynx habitat in spruce-fir forests.
Tactic effectiveness (risks)	High	High	Moderate
Implementation urgency	Near term	Near term	Mid term
Where can tactics be applied? (geographic)	Higher elevations	Higher elevations	Higher elevations
Opportunities for implementation	Whitebark Pine Ecosystem Foundation, Forest Health Protection, programmatic National Environmental Policy Act work, stewardship contracting	Whitebark Pine Ecosystem Foundation, Forest Health Protection, programmatic National Environmental Policy Act work	Rocky Mountain Research Station, State fish and game, U.S. Fish and Wildlife Service, fire management
Cost	Moderate	Moderate	Moderate
Barriers to implementation	Endangered species (lynx), lack of available funding, access and short operating season, management limitations within roadless wilderness, and potential litigation	Endangered species (lynx), lack of available funding, access and short operating season, management limitations within roadless wilderness, and potential litigation	Knowledge gaps in desired conditions

Table 6C.24—Adaptation options that address climate change effects on forest vegetation in the Western Rockies subregion.

Sensitivity to climatic variability and change: Increasing disturbance, moisture deficits, stress due to densities, and longer growing seasons.

Adaptation strategy/approach: Promote resilience in forest vegetation.

Strategy objective: Promote resilience by reducing stressors and facilitating transitions.

	Specific tactic – D	**Specific tactic – E**	**Specific tactic – F**
Tactic	Promote legacy trees of western larch, Douglas-fir, western white pine, Engelmann spruce, whitebark pine, and alpine larch.	Establish seed collection and seed bank needs for alpine larch, whitebark pine, and high-elevation western larch. Coordinate with other land managers/owners on post-wildfire reforestation and planting of whitebark pine.	Promote landscape heterogeneity and create connectivity at multiple scales through strategic placement of treatment units. Increase patch size/treatments and cross-ownership boundary coordination.
Tactic effectiveness (risks)	High	High	Moderate
Implementation urgency	Near term	Near term	Mid term
Where can tactics be applied? (geographic)	Within dense spruce-fir-lodgepole pine sites	Where these species occur	Landscape
Opportunities for implementation	National forest plan desired conditions and project	Whitebark Pine Ecosystem Foundation, volunteers (e.g., backcountry horseman and stakeholders), Forest Health Protection	National forest plan desired conditions and project
Cost	Inexpensive, can be done in conjunction with project	Expensive because of remote location/poor access, but volunteers can help reduce costs	Moderately expensive
Barriers to implementation	None other than lack of these trees in some areas and windthrow risk	Budget (to reduce this barrier, solicit partners); wilderness/roadless policy limitations	Some barriers, knowledge gaps in desired conditions.

Table 6C.25—Adaptation options that address climate change effects on forest vegetation in the Western Rockies subregion.

Sensitivity to climatic variability and change: Increasing disturbance, moisture deficits, stress due to densities, and longer growing seasons.

Adaptation strategy/approach: Promote resilience in forest vegetation.

Strategy objective: Promote resilience by reducing stressors and facilitating transitions.

	Specific tactic – G	**Specific tactic – H**	**Specific tactic – I**
Tactic	Promote rapid response and assessment for postfire reforestation.	Implement a triage approach to soil moisture conditions (e.g., loess soil mapping) for prioritizing areas for western white pine and whitebark pine.	Protect high value trees (plus other trees, cone producers, and rare species) via management actions, including pheromones and seed orchards.
Tactic effectiveness (risks)	High	High	High
Implementation urgency	Near term	Near term	Near term
Where can tactics be applied? (geographic)	Throughout zone	Throughout zone	Throughout zone
Opportunities for implementation	Burned area emergency response	Intermountain Forest Tree Nutrition Cooperative, Natural Resources Conservation Service, Rocky Mountain Research Station	Whitebark Pine Ecosystem Foundation, volunteers, Forest Health Protection
Cost	Moderate	Moderate	Moderate
Barriers to implementation	Burned Area Emergency Response does not fund planting; commitment from forests; lack of strategy	Funding; lack of understanding of utility by forests	Funding

Table 6C.26—Adaptation options that address climate change effects on forest vegetation in the Western Rockies subregion.

Sensitivity to climatic variability and change: Accelerated root disease mortality due to climate stressors.

Adaptation strategy/approach: Reduce dominance of root disease sensitive species (e.g., Douglas-fir and grand fir) on root disease-prone sites.

Strategy Objective: Facilitate transition.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Regenerate and plant with species less susceptible to root disease.	Thin out root disease-susceptible species where less root disease-susceptible species are abundant.	Conduct a hot prescribed burn, followed by a re-burn.
Tactic effectiveness (risks)	High	Moderate	Unknown
Implementation urgency	Near term	Near term	Near term
Where can tactics be applied? (geographic)	Root disease-prone sites with current root disease losses and where management is allowed	Root disease-prone sites; in areas where management is allowed; where adequate, less susceptible species are present and current losses are low	Root disease prone sites and in areas where management is allowed
Opportunities for implementation			
Cost	Inexpensive	Inexpensive	Inexpensive
Barriers to implementation	Where losses are severe and timber value may not cover cost of site preparation	May not be highest priority	5-year regeneration requirement

Table 6C.27—Adaptation options that address climate change effects on forest vegetation in the Western Rockies subregion.

Sensitivity to climatic variability and change: Sites with limited species and genetic diversity are more likely to be impacted by climate change and climate-related stressors.

Adaptation strategy/approach: Work across jurisdictions at larger spatial scales.

Strategy objective: Promote resilience.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Plant potential microsites with a mix of species to push the envelope and hedge bets.	Maintain species diversity during thinning.	Interplant to supplement natural regeneration and genetic diversity.
Tactic effectiveness (risks)	Moderate	High	Moderate
Implementation urgency	Near term	Near term	Near term
Where can tactics be applied? (geographic)	In sufficiently large openings (from harvest, fire, or in understocked stands) on sites most at risk for drought stress	Dense or overstocked stands on sites more at risk for drought stress and where sufficient diversity exists for thinning.	In sufficiently large openings (from harvest, fire, or in understocked stands) on sites most at risk for drought stress.
Opportunities for implementation			
Cost	Inexpensive	Inexpensive	Inexpensive
Barriers to implementation	Openings are lacking because of limited timber harvest and fire suppression.	Insufficient funding. Thinning restrictions due to lynx.	Openings are lacking because of limited timber harvest and fire suppression.

Table 6C.28—Adaptation options that address climate change effects on forest vegetation in the Western Rockies subregion.

Sensitivity to climatic variability and change: Increasing moisture demands and drought stress in moisture-demanding species (western hemlock and western redcedar) on upland sites.

Adaptation strategy/approach: Minimize the effects to the stand from the affected species (western hemlock and western redcedar).

Strategy objective: Promote resilience and facilitate transition.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Implement pre-commercial thinning to limit dominance of these species on drought-prone sites.	Encourage regeneration harvest and planting with a more diverse species mix.	Preserve road and trail access despite increased fire and flood events.
Tactic effectiveness (risks)	Moderate	High	Moderate
Implementation urgency	Near term	Near term	Mid term
Where can tactics be applied? (geographic)			
Opportunities for implementation			
Cost			
Barriers to implementation			

Table 6C.29—Adaptation options that address climate change effects on forest vegetation in the Western Rockies subregion.

Sensitivity to climatic variability and change: Western white pine is resilient but underrepresented.

Adaptation strategy/approach: Increase presence of western white pine.

Strategy objective: Promote resilience and facilitate transition.

	Specific tactic – A	**Specific tactic – B**	**Specific tactic – C**
Tactic	Implement planting of blister rust-resistant western white pine.	Implement blister rust pruning.	Retain existing natural white pine when thinning, harvesting, and burning.
Tactic effectiveness (risks)	High	High	Thinning/harvest – high Burning – moderate
Implementation urgency	Near term	Near term	Near term
Where can tactics be applied? (geographic)	Western redcedar, western hemlock and moist grand fir habitat types (and on moister microsites of drier habitat types)	Stands with moderate infection, but low infection above pruning height; may be critical if thinning	Where western white pine is present
Opportunities for implementation			
Cost	Inexpensive	Inexpensive	Inexpensive
Barriers to implementation	Openings are lacking because of limited timber harvest and fire suppression.	Insufficient funds and access	More difficult on steep slopes, where fire is needed for site preparation