Chapter 5
Effects of Climate Change on Forest Vegetation in the Northern Rockies

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Abstract Increasing air temperature, through its influence on soil moisture, is expected to cause gradual changes in the abundance and distribution of tree, shrub, and grass species throughout the Northern Rockies, with drought tolerant species becoming more competitive. The earliest changes will be at ecotones between life-forms (e.g., upper and lower treelines). Ecological disturbance, including wildfire and insect outbreaks, will be the primary facilitator of vegetation change, and future forest landscapes may be dominated by younger age classes and smaller trees. High-elevation forests will be especially vulnerable if disturbance frequency

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increases significantly. Increased abundance and distribution of non-native plant species, as well as the legacy of past land uses, create additional stress for regeneration of native forest species.

Most strategies for conserving native tree, shrub, and grassland systems focus on increasing resilience to chronic low soil moisture, and to more frequent and extensive ecological disturbance. These strategies generally include managing landscapes to reduce the severity and patch size of disturbances, encouraging fire to play a more natural role, and protecting refugia where fire-sensitive species can persist. Increasing species, genetic, and landscape diversity (spatial pattern, structure) is an important "hedge your bets" strategy that will reduce the risk of major forest loss. Adaptation tactics include using silvicultural prescriptions (especially stand density management) and fuel treatments to reduce fuel continuity, reducing populations of nonnative species, potentially using multiple genotypes in reforestation, and revising grazing policies and practices. Rare and disjunct species and communities (e.g., whitebark pine, quaking aspen) require adaptation strategies and tactics focused on encouraging regeneration, preventing damage from disturbance, and establishing refugia.

Keywords Forest productivity • Climate change vulnerabilities • Adaptation strategies and tactics • Conifer forests • Ponderosa pine • Whitebark pine • Lodgepole pine • Grand fir • Douglas-fir • Western white pine • Western red cedar • Green ash • Cottonwood • Limber pine

5.1 Introduction

Climate change will affect vegetation assemblages in the Northern Rockies *directly* through altered vegetation growth, mortality, and regeneration, and *indirectly* through changes in disturbance regimes and interactions with altered ecosystem processes (e.g., hydrology, snow dynamics, nonnative species). Some species may be in danger of decreased abundance, whereas others may expand their range. New vegetation communities may form, and historical vegetation complexes may shift to other locations or become rare.

Here we assess the effects of climate change on forest vegetation, based on species autecology, disturbance regimes, current conditions, and modeling. We focus on important Northern Rockies forest tree species and the vegetation types in Fig. 5.1, inferring the vulnerabilities of each species and vegetation type from information found in the literature. Vulnerability is considered with respect to heterogeneous landscapes, including both vegetation disturbance and land-use history.

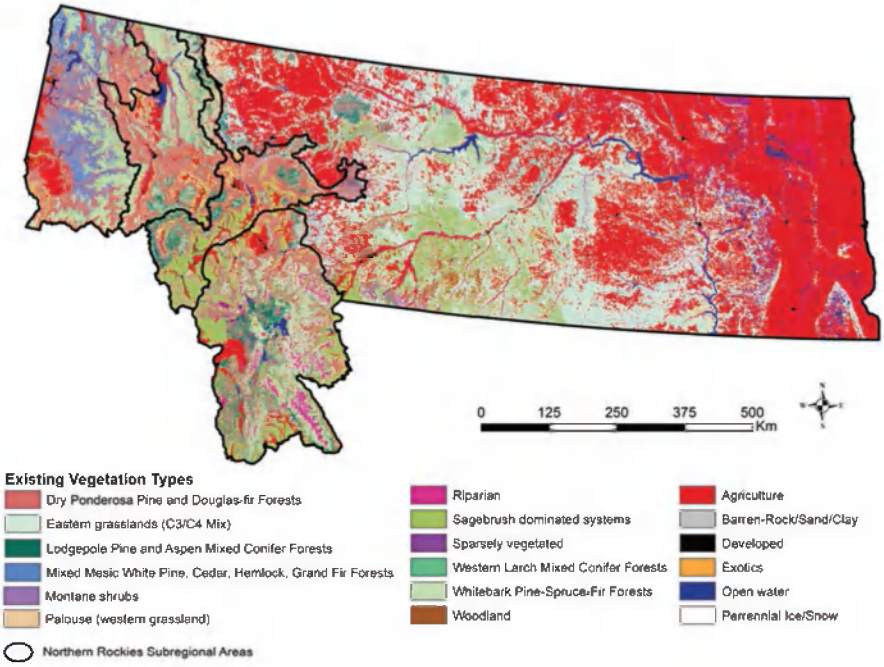

Fig. 5.1 Current vegetation types for the five Northern Rockies subregions. The map was created from the LANDFIRE Existing Vegetation Type map by aggregating National Vegetation Classification Standard vegetation types into a set of vegetation types relevant at coarse spatial scales

5.1.1 Climate Change Assessment Techniques

Past efforts to project the effects of climate change on ecosystem processes have primarily used four techniques (Clark et al. 2001; Schumacher et al. 2006; Joyce et al. 2014). *Expert opinion* involves experts in the fields of climate change, ecology, and vegetation dynamics qualitatively assessing the effects of various climate change scenarios on vegetation. *Field assessment* involves sampling or remote sensing to monitor vegetation change as the climate warms. Although field assessment techniques are the most reliable and useful, they are often intractable because of the large areas and long time periods for which sampling is needed to detect changes. S*tatistical analysis* can be used to create empirical models that project climate change response, including projections of habitat, range, or occupational shifts of tree species from climate warming using species distribution models (e.g., Iverson and Prasad 2002). This type of model is inherently flawed, because it relates contemporary species occurrence to current climate, resulting in predictions of *potential* species habitat, not species distribution, and does not include interacting ecological processes (e.g., reproduction, tree growth, competitive interactions, disturbance) (Iverson and McKenzie 2013). *Modeling* to assess climate-mediated

vegetation responses is the most effective technique, using projected future climate as inputs into ecological models to simulate climate change effects and interactions (Keane et al. 2004). Models focused on large spatial scales (100–1000 km^2) are best suited for projecting climate change effects, because most ecosystem processes operate and most management decisions are made at large scales (Cushman et al. 2007; McKenzie et al. 2014).

A mechanistic, process-driven simulation approach is needed to emphasize physical drivers of vegetation dynamics directly related to climate, which makes model design complex, with many species characteristics and disturbance factors (Lawler et al. 2006). Ecosystem models that accurately project climate change effects must simulate disturbances, vegetation, climate, and their interactions across multiple spatial scales, but few models simulate ecosystem processes with the mechanistic detail needed to realistically represent important interactions (Keane et al. 2015b; Riggs et al. 2015). A fully mechanistic approach 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).

5.1.2 Forest Vegetation Responses to Climate

The effects of climate change on forest vegetation will be driven primarily by altered disturbance regimes, and secondarily through shifts in regeneration, growth, and mortality (Flannigan et al. 2009; Temperli et al. 2013). Trees will respond to reduced water availability, higher temperatures, and changes in growing season in different ways, but because trees are stationary organisms, altered vegetation composition and structure will be the result of changes in plant processes and responses to disturbance.

Several modes of plant function will determine fine-scale response to climate change (Joyce and Birdsey 2000). *Productivity* may increase in some locations because of increasing temperatures and longer growing seasons (especially at higher elevation), but decrease in others where soil moisture decreases (especially at lower elevation). The window of successful *seedling establishment* will change (Ibañez et al. 2007), and increasing drought and high temperatures may narrow the time for effective regeneration in low-elevation forests and widen the window in high-elevation forests. *Tree mortality* can be caused by temperature or moisture stress, as well as late growing-season frosts and high winds (Joyce et al. 2014). *Phenology* may be disrupted in a warmer climate, with some plants experiencing damage or mortality when phenological cues and events are mistimed with new climates (e.g., flowering during dry portions of the growing season). Finally, *genetic limitations* of species or trees may affect their response to climate change (e.g., species restricted to a narrow range of habitat conditions may become maladapted) (Hamrick 2004; St. Clair and Howe 2007) (Table 5.1).

Table 5.1 Comparison of attributes characterizing adaptive strategies for tree species (After Rehfeldt 1994)

	Adaptive strategy	
Attributes	Specialist	Generalist
Factors 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	Mostly among populations	Mostly within populations

Direct effects of temperature on plant growth may increase photosynthesis and respiration. If projected temperatures exceed photosynthetic optima (especially at low elevation), then plant growth might suffer, whereas some trees at high elevation may have photosynthetic gains. Respiration also increases with temperature, 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 may increase water-use efficiency (and growth) in some conifer species, potentially compensating for lower water availability. Longer growing seasons and a more variable climate may affect dormancy regulation, bud burst, and early growth (Chmura et al. 2011). Warmer temperatures may reduce growing-season frosts in mountain valleys, thereby allowing cold-susceptible species, such as ponderosa pine (*Pinus ponderosa*), to exist in habitats currently occupied by other species. Snowmelt provides much of the water used by trees in mountain forests, so amount and duration of snowpack will greatly influence regeneration and growth patterns, typically having a negative effect at low elevation and often a positive effect at high elevation (Peterson and Peterson 2001).

Human land-use activities may overwhelm climate change effects in some cases. For example, decades of fire exclusion have resulted in increased tree regeneration and denser canopies in dry forests, coupled with accumulation of fuels (Keane et al. 2002). Because these conditions create competition for water, light, and nutrients, trees in fire-excluded forests are often stressed, making them susceptible to mortality from secondary stressors, such as drought, insect outbreaks, and fire. Most tree species are long lived and genetically diverse, so they can survive wide fluctuations of weather, but interacting drought and modified disturbance regimes will probably play a major role in the future distribution and abundance of forest species (Allen et al. 2010). Most plants have slow migration rates, often depending more on regenerative organs (e.g., sprouting) than seed dispersal. The potential for tree species to migrate may differ among different mountain ranges, depending on local biophysical conditions.

Genetic diversity helps species adapt to changing environments, colonize new areas, and occupy new ecological niches (Ledig and Kitzmiller 1992) (Table 5.1).

Species and populations vulnerable to climate change are typically rare, 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 elevation limit of their distribution, and populations threatened by habitat loss, fire, insects, or disease (Spittlehouse and Stewart 2003; St. Clair and Howe 2007). Fragmentation is a critical issue for plant populations because isolation and a small number of individuals can promote inbreeding and loss of genetic diversity (Broadhurst et al. 2008).

5.1.3 Biotic and Abiotic Disturbances

Most changes in vegetation will be facilitated through responses to disturbance or to stress complexes in which multiple factors interact to modify ecosystem structure and function (McKenzie et al. 2009; Iverson and McKenzie 2013; Keane et al. 2015a). Fire exclusion in the Northern Rockies since the 1920s has disrupted annual occurrence, spatial extent, and cumulative area burned by wildfires, resulting in increased surface fuel loads, tree densities, and ladder fuels, especially in low-elevation, dry conifer forests. If drought increases as expected, area burned will increase significantly (McKenzie et al. 2011; Peterson et al. 2014). Reduced snowpack and drier fuels could also make high-elevation forests more susceptible to increasing fire occurrence (Miller et al. 2009).

Insect activity and outbreaks are also affected by climate and will dictate future forest composition and structure. Mountain pine beetle (*Dendroctonus ponderosae*) is an aggressive and economically important insect responsible for high tree mortality across large areas (Logan et al. 2003), and warming temperatures have directly influenced bark beetle-caused tree mortality in much of western North America (Safranyik et al. 2010). Future mortality will depend on spatial distribution of live host trees, heterogeneity of future landscapes, and ability of beetle populations to adapt to changing conditions.

5.2 Climate Change Effects on Tree Species

Climate change effects on trees species in the Northern Rockies were inferred based on autecology, disturbance interactions, and current and historical conditions (Table 5.2). Most information is from published literature, tempered with professional experience. Primary sources of autecological information are Minore (1979), Burns and Honkala (1990), Bollenbacher (2012), and Devine et al. (2012). MC2 model output was used to evaluate climate change effects on important species and vegetation types (Figs. 5.2 and 5.3). The literature is sometimes inconsistent on the response of tree species to climate change, reflecting considerable uncertainty about

Table 5.2 Climate change vulnerability ratings for tree species in the Northern Rockies (including its five subregions); ratings are ordinal, with 1 being the most vulnerable. Species not included in a subregion are indicated by a dash

		Subregion				
Species	Northern Rockies	Western Rockies	Central Rockies	Eastern Rockies	Greater Yellowstone Area	Grassland
Alpine larch	1	2	1	–	–	–
Whitebark pine	2	1	2	1	1	–
Western white pine	3	5	3	–	–	–
Western larch	4	6	4	–	–	–
Douglas-fir	5	8	8	2	2	1
Western redcedar	6	4	5	–	–	–
Western hemlock	7	3	6	–	–	–
Grand fir	8	7	7	–	–	–
Engelmann spruce	9	9	11	3	4	5
Subalpine fir	10	10	12	4	5	6
Lodgepole pine	11	11	10	5	6	7
Mountain hemlock	12	3	9	–	–	–
Cottonwood	13	12	13	6	3	2
Aspen	14	13	14	8	7	3
Limber pine	15	–	15	7	8	4
Ponderosa pine-west	16	14	16	–	–	–
Ponderosa pine-east	17	–	–	9	9	8
Green ash	18	–	–	10	10	9

projections. In addition, the amount of climate change matters. Most climate change studies project minimal changes after moderate warming (B1, B2, A1B, RCP 4.5 scenarios), but major species shifts under extreme emission scenarios (A1, A2, RCP 8.5 scenarios). The time frame used affects the magnitude of response, with most studies projecting much greater changes in vegetation after the mid-twenty-first century.

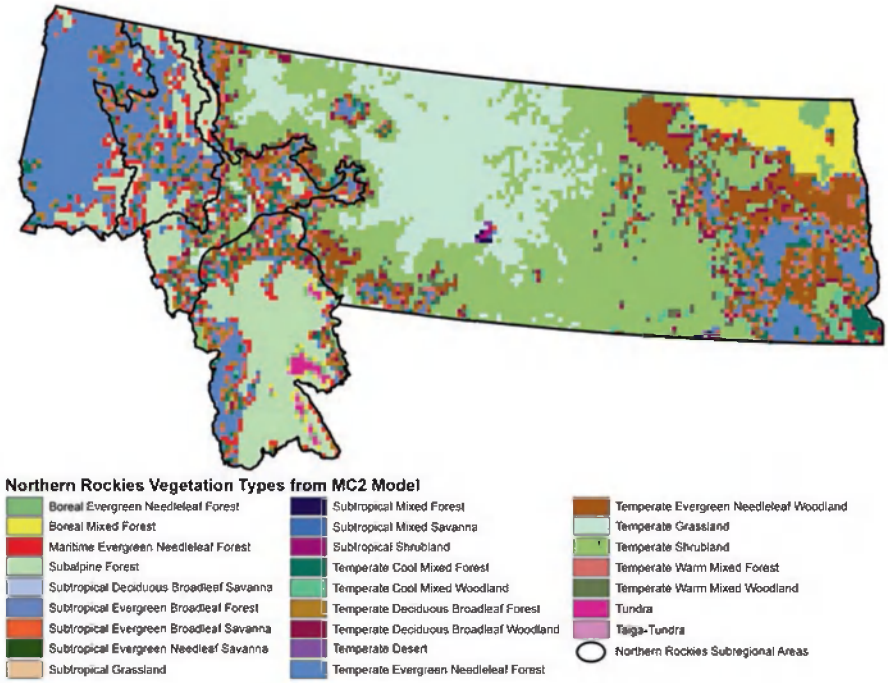

Fig. 5.2 Vegetation types used in the MC2 model within the five Northern Rockies subregions. These vegetation types are represented in the model output in Fig. 5.3

*5.2.1 Ponderosa Pine (*Pinus ponderosa*)*

Ponderosa pine is a shade-intolerant, drought-adapted species in low-elevation, dry forests of the Northern Rockies (Minore 1979). Ponderosa pine is a "drought avoider" that 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). Seedlings 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 Schimidt 1970). As a seral species, ponderosa pine is often associated with Douglas-fir (*Pseudotsuga menziesii*), lodgepole pine (*Pinus contorta* var. *latifolia*), grand fir (*Abies grandis*), and western larch (*Larix occidentalis*). Ponderosa pine is highly resistant to fire, more resistant than nearly all of its competitors (Ryan and Reinhardt 1988), which historically allowed it to maintain dominance over large areas that burned frequently.

Ponderosa pine is expected to tolerate increasing temperatures and droughts with only moderate difficulty. As a "drought avoider," it can close stomata at low soil-water potential, allowing it to persist in low-elevation sites (Stout and Sala 2003). Three studies have projected an expansion of the range of ponderosa pine in a warmer climate (Hansen et al. 2001; Nitschke and Innes 2008; Morales et al. 2015).

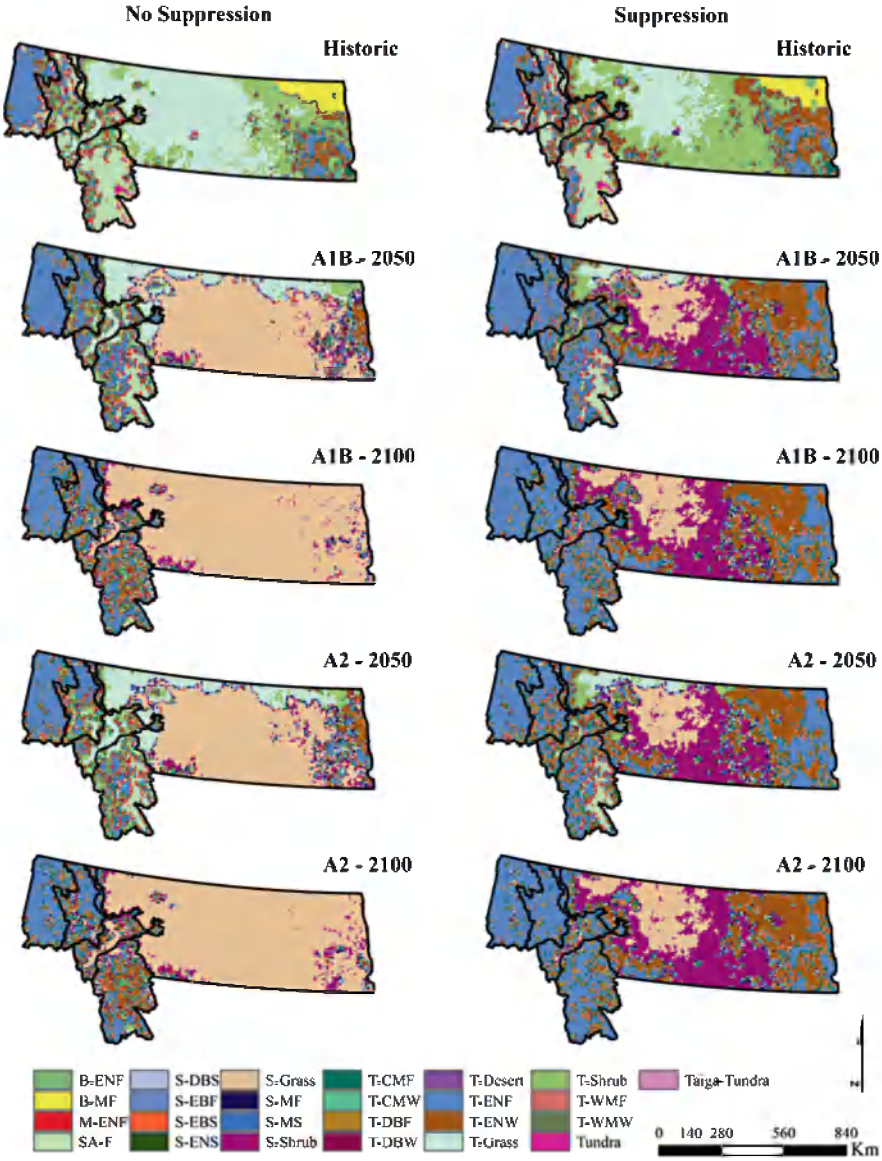

Fig. 5.3 Maps of MC2 vegetation type distributions with and without fire suppression (the two columns) for two emission scenarios (A1B, A2) and three time periods (historical, 2050, 2100) (each row) for five Northern Rockies subregions (outlined in bold on each map). Vegetation types are: *B* boreal, *M* maritime, *SA* subalpine, *S* subtropical, *T* temperate, *ENF* evergreen needleleaf forest, *ENW* evergreen needleleaf woodland, *F* forest, *MF* mixed forest, *MW* mixed woodland, *DBF* deciduous broadleaf forest, *DBW* deciduous broadleaf woodland (not all vegetation types in the legend are represented in the map)

There may also be opportunities for this species to move to higher elevations, based on competitiveness in dry soils (Gray and Hamann 2013). Increases in mountain pine beetle and other insects, advancing competition resulting from fire exclusion, 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 survive, and mature individuals will not become established. Increasing fire severity and occurrence could also eliminate old trees that provide seed sources for populating future burns.

5.2.2 *Douglas-Fir (*Pseudotsuga menziesii*)*

Douglas-fir is a major component of lower elevation and mixed-conifer forests in the Northern Rockies (only Rocky Mountain Douglas-fir [var. *glauca*] is found here]). It is an early-seral species in moist habitats with western larch, western white pine (*Pinus monticola*), grand fir, western redcedar (*Thuja plicata*), and western hemlock (*Tsuga heterophylla*), and is late seral in drier habitats with ponderosa pine, juniper, and quaking aspen (*Populus tremuloides*). Douglas-fir is a "drought tolerator," keeping stomata open to extract soil water at low soil-water potentials, thereby subjecting it to xylem cavitation and potentially death in extreme drought (Stout and Sala 2003; Sala et al. 2005). Western spruce budworm (*Choristoneura occidentalis*), Douglas-fir tussock moth (*Orgyia pseudotsugata*), and Douglas-fir beetle (*Dendroctonus pseudotsugae*) are prominent insects that affect this species. Thick bark, thick foliar buds, and deep main roots confer high resistance to wildfire.

Douglas-fir is expected to have low to moderate vulnerability to climate change. Recent modeling results project no change to significant increases in the range of this species in a warmer climate (Morales et al. 2015), although it is possible that it will decrease in drier portions of its range (Nitschke and Innes 2008). Growth will probably decrease somewhat in a warmer climate, both in the Northern Rockies and the rest of the western United States (Restaino et al. 2016). Increased wildfire, coupled with adverse effects of fire exclusion, could cause mortality in Douglas-fir in stands with high fuel loadings. If fires increase, they may be so frequent that Douglas-fir seedlings cannot become established and become mature trees. Recent surveys show significant increases in Douglas-fir seedling mortality in response to increasing drought and high temperature, which may become more common in the future.

5.2.3 *Western Larch (*Larix occidentalis*)*

Western larch grows in moist, cool environments in valley bottoms, benches, and northeast-facing mountain slopes. Larch has low water-use efficiency compared to other conifers in the Northern Rockies, explaining its absence on xeric sites. Early

autumn cold snaps affect seedling and sapling survival (Rehfeldt 1995), and drought affects mid- to late-season survival. Larch is a long-lived, shade intolerant, early-seral species, growing fast with tall, open crowns and outcompeting other species (Milner 1992). It is moderately drought tolerant and can survive seasonal drought, but performs poorly when droughts last more than 2 years. Douglas-fir is the most common associate, but many other species can be found with larch. Frequent, low-intensity wildfire historically maintained dominance of larch, which is very tolerant of fire (Ryan and Reinhardt 1988). Extensive logging removed many of the large larch that could have survived fire, and fire exclusion has eliminated the burned mineral soil seedbeds where western larch can regenerate.

Western larch may be susceptible to future changes in climate because of its narrow distribution in the Northern Rockies and its uncertain association with wildfire. Modeling studies suggest that larch may be susceptible to a warmer climate, with potentially large constrictions of its range in some locations (Fins and Steeb 1986; Nitschke and Innes 2008; Aston 2010; Morales et al. 2015). Western larch may migrate to more northerly and higher areas in the Northern Rockies, but not without surviving major fires (Gray and Hamann 2013). Increasing fires may help return western larch to the Northern Rockies landscape, but this would require significant assistance from planting. Continued fire exclusion will probably result in continuing declines of western larch, because increased competition will reduce vigor, making trees more susceptible to insects and pathogens, and fuel loadings will propagate crown fires, causing high larch mortality (Keane et al. 1996).

*5.2.4 Western White Pine (***Pinus monticola***)*

Western white pine grows at mid elevations in the Northern Rockies, often in steep topography along moist creek bottoms, lower benches, and north aspects. Intermediate in shade tolerance, it is usually an early-seral species (Minore 1979), attaining dominance in a stand only following wildfire or through silvicultural systems that favor it. Once established, western white pine grows best in full sunlight. Seedlings have low drought tolerance, and seedling mortality in the first growing season is attributed to high surface temperatures and low soil moisture. White pine is tolerant of cold when dormant. Seed germination requires 20–120 days of cold, and occurs following snowmelt, typically on mineral soil (Graham 1990). Mature trees are relatively tolerant of wildfire, especially where they have high, open crowns. The abundance and distribution of white pine is currently restricted because of removal through logging over the past century.

Western white pine may be reasonably well adapted to higher temperature in wetter portions of the Northern Rockies (Loehman et al. 2011). Its fast growth rate and ability to survive fire provide ecological resilience, and it disperses seeds heavily into burned areas, providing an effective means of regeneration with high fire occurrence. However, in much of its range, white pine will be susceptible to declines from interacting effects of fire exclusion, white pine blister rust, and rapid succes-

sion to shade tolerant conifer communities. White pine blister rust is a huge stressor, because white pine has not yet developed the genetic capacity to overcome this disease (Fins et al. 2002). Therefore, even if wild fire increases opportunities for regeneration, there may be few residual trees to provide the necessary seed source. 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 Without a comprehensive restoration program, this species may never again be dominant in the Northern Rockies.

5.2.5 *Grand Fir (*Abies grandis*)*

Grand fir is found on a wide variety of sites in the Northern Rockies, including stream bottoms and valley and mountain slopes (Foiles et al. 1990). It is either an early-seral or late-seral species depending on site moisture (Ferguson and Johnson 1996), often found with Douglas-fir, ponderosa pine, western redcedar, western hemlock, and other species. Grand fir is shade tolerant, but is relatively intolerant of drought. It has low frost tolerance but can tolerate seasonally fluctuating water tables. It 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). It is susceptible to heart rot and decay, especially armillaria root rot (*Armillaria* spp.) and annosus root disease (*Heteribasidion annosum*), and is attacked by numerous insects (Foiles et al. 1990). Fire exclusion has greatly increased grand fir on both dry and mesic sites, but increased tree densities have also stressed grand fir trees, contributing to increased fuel loadings, and higher damage and mortality from root rot and insects.

On xeric sites, increased drought and longer growing seasons will exacerbate stress for grand fir, with increased competition and potentially high mortality from insects and disease. Modeling studies have projected major declines in this species by the end of the twenty-first century (Nitschke and Innes 2008; Coops and Waring 2011). However, increased productivity may lead to expanded grand fir populations on sites with moderate moisture (Urban et al. 1993; Aston 2010). On mesic sites where grand fir is seral to western redcedar and western hemlock, longer growing seasons coupled with higher temperatures may increase growth rates and regeneration success. Longer fire seasons and high fuel loadings will potentially rearrange grand fir communities across the Northern Rockies, reducing grand fir dominance at both large and small scales. Although many grand fir forests are stressed from high tree densities, the species will probably tolerate changes in climate and remain on the landscape at levels similar to historical conditions.

5.2.6 *Western Redcedar (*Thuja plicata*)*

Western redcedar is a component of mesic forests in the Northern Rockies, occupying wet ravines and poorly-drained depressions, often as a riparian species. Shade tolerance is high, and it is often present in all stages of forest succession. It is associated with grand fir, western white pine, western hemlock, western larch, and ponderosa pine, occurring in pure stands only where fire has been excluded for a long time, or where fire has been used to maintain redcedar dominance. It regenerates best on disturbed mineral soil, although scorched soil is not beneficial for regeneration, and seedlings survive best in partial shade. It is not resistant to drought or frost, and can be damaged by freezing temperatures in late spring and early autumn. Western redcedar is not severely affected by most insects and pathogens (Minore 1990), and is moderately fire tolerant when mature.

Western redcedar may retain its current range in a warmer climate, and productivity may increase in cooler, wetter locations (Hamann and Wang 2006; Aston 2010). Although warmer conditions may benefit redcedar in some locations, drier conditions would likely reduce its distribution and productivity in dry to mesic sites, especially if it becomes more susceptible to insect attacks (Woods et al. 2010). Warming could also result in a loss of chilling required for western redcedar growth and reproduction (Nitschke and Innes 2008). The potential effects of disturbance on redcedar are unclear. Fire can maintain redcedar communities if it burns at low severities and kills only seedlings and saplings, but high-severity wildfires can eliminate seed sources. Continued fire exclusion may maintain current western redcedar distributions, but without fuel treatments, crown fires may be sufficiently common to cause extensive mortality. Western redcedar is often associated with ash cap soils, and the potential of redcedar to migrate to non-ash soils under new climates may be limited.

5.2.7 *Western Hemlock (*Tsuga heterophylla*)*

Western hemlock is found in mild, humid climates and in environments with abundant soil moisture throughout the growing season, typically found in association with western redcedar, grand fir, Douglas-fir, western larch, western white pine, lodgepole pine, and ponderosa pine. Where soils are relatively dry in summer, hemlock is found primarily on north aspects and in moist drainages and other locations where water is available. This species is very shade tolerant and usually considered a late-seral species, although it is often common at all stages of stand development. Hemlock is highly susceptible to drought during the growing season (Baumgartner et al. 1994). It can germinate on a variety of organic and mineral seedbeds, and its seedlings are highly susceptible to frost. Many root and bole pathogens cause significant damage and mortality in western hemlock. It is very susceptible to fire damage because of its shallow roots and thin bark, and its relatively shallow root system

makes it susceptible to wind throw. Most stands in the Northern Rockies that contain western hemlock have become denser over the past century, with the hemlock component increasing in the overstory and understory, a condition often leading to reduced vigor.

Increased drought and area burned are expected to reduce abundance and distribution of hemlock, especially in drier locations. Several studies have projected contractions in western hemlock distribution. For example, Hansen et al. (2001) simulated major contractions in western hemlock range, and Shafer et al. (2001) reported that western hemlock may decrease in range because chilling requirements for the seeds will not be met. Keane et al. (1996) simulated losses of western hemlock and redcedar under moderate climate warming in Glacier National Park, mostly as a result of severe fires. Other studies project both a decrease and increase in western hemlock in a warmer climate (Urban et al. 1993; Cumming and Burton 1996; Hamann and Wang 2006). It is possible that western hemlock will maintain most of its current range in the future, although it may not have the diversity in growth habit to allow it to expand its range into higher-elevation sites as temperatures warm.

5.2.8 *Lodgepole Pine (*Pinus contorta var. latifolia*)*

Lodgepole pine has the widest range of environmental tolerance of any conifer in North America (Lotan and Critchfield 1990) and is found in a broad range of soils and local climatic conditions in the Northern Rockies. Shade intolerant and relatively tolerant of both drought and cold temperatures, this species grows in nearly pure stands as well as in association with several other conifer species. The presence of cone serotiny in most populations allows lodgepole pine to reproduce prolifically following wildfire, and seedlings can survive diverse microsite and soil conditions, although drought is a common cause of mortality in first-year seedlings. Fire plays a critical role in lodgepole pine forest succession, and many current forests originated from stand-replacement fires. Mature trees have moderate tolerance to fire and can survive light burns. Mountain pine beetle also plays a significant role in the dynamics of lodgepole pine ecosystems, as evidenced by recent large outbreaks in the Northern Rockies and much of western North America, which have resulted from a combination of increased temperature and an abundance of low-vigor stands (Carroll et al. 2003).

Longer droughts and warmer temperature in lower-elevation sites may reduce lodgepole pine growth and regeneration, with a possible transition to other tree species (Chhin et al. 2008; Nigh 2014). The results of different modeling studies are equivocal about the future distribution of this species in a warmer climate, but given that lodgepole pine is a generalist 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 over long time periods. In the subalpine zone where seasonal drought is not a problem, moderate warming may

increase lodgepole pine productivity (Johnstone and Chapin 2003; Wang et al. 2006; Aston 2010) and possibly distribution. The ultimate fate of lodgepole pine will depend on frequency and extent of wildfire (Smithwick et al. 2009). Populations with serotiny can be expected to respond well to future fire, but very frequent fire could eliminate younger stands. In addition, mountain pine beetle outbreaks will probably have a major influence on lodgepole pine abundance and distribution in a warmer climate (Creeden et al. 2014). As with fire, the frequency and extent of beetle-caused mortality will dictate future stand conditions (Logan and Powell 2001). In summary, lodgepole pine distribution may both expand and contract depending on location, but the species is expected to persist in the Northern Rockies as long as fire remains on the landscape.

*5.2.9 Limber Pine (*Pinus flexilis*)*

Limber pine is a shade-intolerant, early-seral species in the Northern Rockies (Steele 1990). Limber pine has difficulty competing with other species on more productive mesic sites and is often succeeded by Douglas-fir and subalpine fir. Reproduction is often very low for this slow-growing, long-lived species whose seeds are dispersed by rodents and Clark's nutcracker (Lanner 1980). Limber pine is tolerant of drought and can become established and grow in arid environments. The broad niche occupied by limber pine indicates this species has a generalist adaptive strategy. However, it is experiencing significant damage from white pine blister rust and mountain pine beetle in some locations (Taylor and Sturdevant 1998; Jackson et al. 2010).

Warming temperatures and decreasing 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, more seeds per cone, wider seed dispersal, and greater resistance to disease. Warm temperature could cause drier soils, especially for seed germination and seedling growth. Disturbance interactions will affect limber pine dynamics in a warmer climate. Increased wildfire may limit its encroachment into grasslands in areas where grazing is low. Warmer, drier conditions may also reduce blister rust infection by disrupting the pathogen life cycle, especially during the late summer when infection occurs.

*5.2.10 Subalpine Fir (*Abies lasiocarpa*)*

Subalpine fir occupies lower valleys to the upper subalpine zone in the Northern Rockies, often associated with grand fir, western larch, Douglas-fir, western redcedar, and western white pine at lower elevations (Pfister et al. 1977) and with lodgepole pine, Engelmann spruce, whitebark pine, alpine larch, and mountain hemlock at higher elevations (Arno 2001). Fir tolerates shade, but cannot tolerate prolonged drought,

especially in the seedling stage. Reproduction tends to occur in pulses relative to periodic seed crops and the occurrence of favorable weather for germination and establishment (Alexander et al. 1990). Fir is highly susceptible to fire damage because of thin bark, dense foliage, and shallow roots (Ryan and Reinhardt 1988), and even low-severity fires can cause high mortality. Several insects and pathogenic fungi damage this species, especially in older, low-vigor stands. Abundance of subalpine fir has increased in some Northern Rockies landscapes (Keane et al. 1994), increasing stress from competitive interactions and causing at least some mortality during dry periods.

With a diverse range throughout the Northern Rockies, subalpine fir could expand its range into the treeline, become more productive in colder portions of its current range, and decline in growth and extent in warmer, drier portions of its current range. Model output ranges from large losses of subalpine fir (Hamann and Wang 2006) to minimal change in its distribution (Bell et al. 2014). Most paleo-reconstructions in the Holocene show that subalpine fir was dominant during cold periods and declined during warm periods (Whitlock 1993, 2004; Brunelle et al. 2005). The future of subalpine fir will depend on the degree of warming and frequency and extent of disturbance, especially wildfire. Increased fire would reduce subalpine fir dominance faster and more extensively than direct climate effects. This species may shift across the high mountain landscape, with gains balancing losses (caused directly by changes in climate). However, future increases in fire, disease, and insects may limit its abundance. Because fir is an aggressive competitor, gains through advanced succession in the upper subalpine zone may balance or exceed losses from fire, drought, and pathogens in the lower subalpine zone.

*5.2.11 Engelmann Spruce (***Picea engelmannii***)*

Engelmann spruce is a major component of high-elevation forests in the Northern Rockies, and although it commonly occurs with subalpine fir, it is also associated with many other conifer species. Spruce is shade tolerant and cold tolerant, but is intolerant of low soil moisture and prolonged drought (Alexander and Shepperd 1990). Seedlings are very intolerant of high temperatures and low soil moisture. Spruce is very susceptible to fire injury and mortality, although some mature trees can survive fire (Bigler et al. 2005), thus providing a post-fire seed source. Spruce beetle (*Dendroctonus rufipennis*) and western spruce budworm are serious stressors, usually attacking older, low-vigor trees. Logging and fire have reduced spruce in some lower-elevation areas in the Northern Rockies.

In a warmer climate, some losses of Engelmann spruce may occur in drier portions of its range, especially in seasonally moist sites. Mortality events in Engelmann spruce over the last 20 years have been attributed to prolonged drought, presumably related to changing climate (Liang et al. 2015), and warm, dry weather has been associated with periods of low growth (Alberto et al. 2013). Most modeling output suggests that spruce will decrease in distribution while moving up in elevation. Spruce may become established in high-elevation locations where snow precluded

conifer regeneration historically (Schauer et al. 1998), particularly because it has the genetic capacity to adapt to large swings in climate (Jump and Peñuelas 2005). With good seed dispersal and tall stature, spruce is able to establish in previously non-forested areas. Paleoclimatic studies indicate that spruce regeneration was highest during the warmest (low snow) periods of the past several centuries. Because spruce is not resistant to wildfire, major declines could occur if projected increases in fire reach forests where spruce is dominant. Spruce beetle will be an ongoing stressor.

*5.2.12 Mountain Hemlock (*Tsuga mertensiana*)*

Mountain hemlock is found in cold, snowy upper subalpine sites where it grows slowly and can live to be more than 800 years old. It is commonly associated with subalpine fir and lodgepole pine, often restricted to north slopes. Hemlock is very shade tolerant, competes well with other species, and is usually considered a late-seral species (Minore 1979; Means 1990). It is not fire resistant, because although it has thick bark when mature, it retains low branches and has shallow roots (Dickman and Cook 1989). Hemlock is susceptible to laminated root rot (*Phellius weirii*) which can rapidly kill large groups of trees (Means 1990). Fire exclusion has probably facilitated an increase in hemlock in some locations in the Northern Rockies.

In a warmer climate, mountain hemlock forests are expected to increase in productivity at the highest elevations, but could experience some drought stress at lower elevations (Peterson and Peterson 2001). This potential for increasing productivity at high elevation may buffer mountain hemlock from a warmer climate for many decades. Higher temperatures and less snowpack would also facilitate increased regeneration (Woodward et al. 1995). The potential effects of fire are a big uncertainty. Mountain hemlock has a limited range in the Northern Rockies, so if warming and drying facilitate increased spread of fire into higher subalpine habitats, then hemlock could be threatened.

*5.2.13 Alpine Larch (*Larix lyallii*)*

Alpine larch is a deciduous conifer that occupies the highest treeline environments in the Northern Rockies, including the Bitterroot, Anaconda-Pintler, Whitefish, and Cabinet Ranges of western Montana. Larch grows in cold, snowy, and generally moist climates, often in pure stands but also associated with whitebark pine, subalpine fir, and Engelmann spruce. Because this species relies on subsurface water in summer, it has very low drought tolerance (Arno 1990). This shade intolerant conifer has a high capacity to survive wind, ice, and desiccation damage during winter when needles are off the trees, and seedlings are very cold tolerant. Fuel loadings are typically low in the subalpine zone, so large fires are infrequent, killing or

injuring larch when they do occur. Alpine larch populations have been relatively constant historically, although the species may be increasing in ribbon forest glades and high-elevation areas where snowpack has been low during the past 20 years.

Alpine larch is expected to be susceptible to climatic shifts that result in increasing drought and fire. As a shade- and drought-intolerant species, alpine larch is not expected to be competitive in increasingly drier soils (Arno and Habeck 1972). Low water availability would be especially damaging to larch in the southern portions of its range, and other subalpine species may be more competitive under stressful conditions. Larch is not well-adapted to survive wildfire (Arno 1990), and if fires become more frequent, this species would experience considerable mortality. However, alpine larch is a prolific seeder and may be able to take advantage of new seedbeds at treeline that were historically covered with snow most of the year. In addition, like other subalpine species, larch may grow faster in a warmer climate. Where alpine larch is able to genetically intergrade with western larch, hybrids may be more tolerant of drought and competition (Carlson et al. 1990).

*5.2.14 Whitebark Pine (*Pinus albicaulis*)*

Whitebark pine is an important component of upper subalpine forests in the Northern Rockies—a keystone species that supports high community diversity (Tomback et al. 2001). It is associated with subalpine fir, Engelmann spruce, and mountain hemlock. This species is slow growing, long lived and moderately shade tolerant (Minore 1979), surviving extended drought, strong winds, thunderstorms, and blizzards (Callaway et al. 1998). It occurs as krummholz and small tree islands at exposed treeline sites. Whitebark pine regeneration benefits from the Clark's nutcracker burying thousands of pine seeds in "seed caches" across diverse forest terrain (Keane et al. 2012). Whitebark pine fire regimes are complex and variable in space and time, creating diversity in age, stand structure, and habitat characteristics (Keane et al. 1994). Mountain pine beetle is the most damaging insect in mature stands, often spreading upward from lodgepole pine forests. A severe epidemic caused high mortality in whitebark pine in the Northern Rockies between 1909 and 1940, and mortality has been high in the Greater Yellowstone Area in recent years. White pine blister rust has killed large numbers of whitebark pine in the Northern Rockies, with mortality exceeding 80% in some locations (Keane et al. 2012). Efforts to propagate rust-resistant pines have led to recent plantings of resistant nursery stock in areas that have burned or were declining.

The fate of whitebark pine in a warmer climate will largely depend on local changes in disturbance regimes and their interactions (Keane et al. 2015a). Although this species has a limited range, it was able to persist through many climatic cycles in the past (Whitlock and Bartlein 1993; Whitlock et al. 2003). The predominant stress of blister rust, which precludes regeneration in burned areas, is the greatest cause for concern individually and in combination with climate change. Recent mortality from blister rust and mountain pine beetle have been widespread in the

Northern Rockies (Keane and Parsons 2010). A warmer climate is expected to exacerbate this decline because (1) pine is confined to upper subalpine environments (2) its populations are low, and (3) its regeneration is limited. The only realistic pathway for maintaining viable populations of whitebark pine in the future is for rust-resistant individuals to survive, supplemented by restoration efforts, in order to propagate stands that are resilient enough to survive to reproduction age.

5.2.15 *Quaking Aspen (*Populus tremuloides*)*

The most widely distributed tree species in North America, quaking aspen is abundant in the mountains of western and southwestern Montana and northern Idaho. Aspen is a short-lived, shade-intolerant, disturbance-maintained seral species. It sprouts aggressively following any disturbance (usually fire) that kills most of the live stems, thus stimulating vegetative propagation (Bartos 1978). Parent trees produce stems, resulting in a clone of genetically identical stems, a reproductive strategy that allows aspen to establish quickly on disturbed sites and out-compete conifers (Mitton and Grant 1996; Romme et al. 1997). 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 (Frey et al. 2004).

Quaking aspen may experience both gains and losses in a warmer climate, depending on local site conditions. Aspen on warmer, drier sites could experience high mortality because of increasing water deficit (Ireland et al. 2014). Sudden aspen decline has been associated with prolonged drought, particularly in aspen stands that are on the edge of its distribution (Frey et al. 2004). Stress complexes with extreme weather (drought, freeze-thaw events), insect defoliation, and pathogens may be particularly damaging (Brandt et al. 2003; Marchetti et al. 2011), and areas with high ungulate herbivory may have little regeneration. Increased fire frequency, particularly on moist sites, will favor aspen regeneration in the future by removing conifers. However, if future fires are severe, they may kill the shallow root systems and eliminate aspen. Areas with mountain pine beetle-caused conifer mortality may release aspen regeneration once the conifer canopy is thinned or removed, assuming sufficient soil moisture is available.

5.2.16 *Cottonwood (*Populus spp.*)*

Black cottonwood (*Populus trichocarpa*) and narrowleaf cottonwood (*P. angustifolia*) grow primarily on seasonally wet to moist, open-canopy sites, typically in riparian areas in the western portion of the Northern Rockies. Plains cottonwood (*P. deltoides*) occupies similar habitat in eastern Montana and the Dakotas. Cottonwood is very shade intolerant, and shade-tolerant conifers can encroach and

become dominant in upland cottonwood forests (e.g., river and stream terraces). It is also drought intolerant, and requires reliable access to the water table during the growing season (Rood et al. 2003). Plains cottonwood is probably more resilient to drought than the other species. High streamflows and deposition of alluvial sediments are required for seedling establishment, and all cottonwood species are prolific producers of windborne seed. Cottonwood is mildly fire tolerant owing to thick bark and high branches, but is a weak sprouter (Brown 1996). Although several insects attack cottonwood, tent caterpillars (*Malacosoma* spp.) are the only important foliar feeders. Many fungal species can cause decay. Black cottonwood is less common today than it was historically.

In a warmer climate, lower snowpacks will alter streamflows, which may in turn affect germination and establishment of young cottonwoods (Whited et al. 2007). Any alteration of hydrologic flow regime will affect both floodplain interaction and available water (Beschta and Ripple 2005). Effects on regeneration could be positive or negative, depending on the frequency and magnitude of flooding and alluvial deposition. Higher human demands for water could also affect water supplies in riparian areas. Some streamflow-floodplain interactions could result in a conversion of streamside vegetation from cottonwood to upland species (Beschta and Ripple 2005). Plains cottonwood, which currently grows in more arid locations, may be more persistent in a warmer climate because it tends to grow in finer-textured soils that retain sufficient water.

*5.2.17 Green Ash (*Fraxinus pennsylvanica*)*

Green ash is restricted to the northern Great Plains, which is the northwestern edge of its distribution. Typically found in riparian areas and floodplains, it is well-adapted to climatic extremes and has been widely planted in the Plains states and Canada. In the northern Great Plains, it grows best on moist, well-drained alluvial soils, but is found in other topographic positions where some subsurface water is available. Green ash is moderately shade tolerant in woody draws and is considered an early-seral species. It can propagate vegetatively through stump sprouting, which provides resilience to both mechanical damage from flooding and to occasional wildfire (Lesica 2009). The species is relatively drought tolerant, although prolonged drought inhibits regeneration. Green ash stems are easily killed by fire, but stumps of most size classes of green ash sprout readily after fire (Lesica 2009). Some green ash communities on the western fringe of the northern Great Plains may be declining compared to historical levels (Lesica 2001).

Green ash has a broad ecological amplitude and can survive low soil moisture, but grows optimally on moist sites. In a warmer climate, marginal sites may become less favorable for regeneration and survival of young trees. Higher temperatures may increase ash growth, as long as sufficient water is available. Increased fire frequency would reduce reproduction by seedlings although most mature trees would persist through sprouting. Browsing pressure on green ash communities may increase with increased drought, as upland grasses and forbs desiccate and senesce

earlier, or are replaced by invasive, less palatable species. The biggest threat to ash may be the non-native emerald ash borer (Agrilus planipennis), which is spreading westward across North America and may reach the Northern Rockies within the next decade.

5.3 Effects of Climate Change on Broader Vegetation Patterns

The assessment of species vulnerabilities discussed above can be aggregated to assess the vulnerability of broader vegetation assemblages to climate change (Figs. 5.2 and 5.3). Understanding climate change response at this higher level is critical because vegetation assemblages (groups of species) are the focus of most forest management and restoration. The assessment below focuses on dominant vegetation types used by the U.S. Forest Service (USFS) Northern Region.

Dry ponderosa pine/Douglas-fir forests, already located in dry regions, are expected to have significant effects in a warmer climate. Some or all of the tree species may expand into the mixed mesic forest type (next section), especially on south slopes, as drought increases. Fire exclusion has resulted in forest densification and accumulation of surface fuels that will likely support high-severity fires in future decades (Keane et al. 2002). With increasing fire, much of this vegetation type could see losses of Douglas-fir and increases in ponderosa pine. Dry Douglas-fir communities that are currently too cool to support ponderosa pine may support more ponderosa pine in the future.

Western larch mixed conifer forests, found in northern Idaho and northwestern Montana, have been greatly altered from their historical structure. 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 fuels, increasing risk of high-severity fire. Western larch is not susceptible to the many insects and diseases common in associated tree species, and is very fire tolerant. However, this species dominates cooler, wetter topographic positions, and a warming climate may constrain the distribution of larch to only north aspects and other microhabitats capable of retaining sufficient water during the growing season (Rehfeldt and Jaquish 2010). This vegetation type will be susceptible to climate-induced increases in area burned by wildfire for the foreseeable future, unless stand structure and landscape pattern can be managed to improve resilience to higher temperatures and higher levels of disturbance.

Mixed mesic western white pine, cedar, hemlock, grand fir forests provide an important context for assessing the effects of climate change. As moist forests experience climate change, competition among species will be dynamic at both small and large spatial scales. A logical approach is to identify specific landscape components that may respond in a coherent manner—north slopes vs. south slopes, side slopes vs. valley bottoms, etc.—and how environmental niches change over time and space. Species that require high soil moisture (hemlock, redcedar) may over

time become less common, with drought tolerant species such as Douglas-fir and ponderosa pine becoming more common (Graham 1990). The frequency and magnitude of disturbance, especially fire, will determine composition and structure in these forests. In the short term, we expect more and larger crown fires. In the long term, we expect that frequent fire will favor fire-resistant tree species, maintain a more open forest structure, and maintain younger age classes.

Lodgepole pine mixed subalpine forests, located at high elevations along and east of the Continental Divide, are expected to have relatively low vulnerability to climate change, depending on if and how disturbance is altered. Productivity of subalpine species may increase in a warmer climate, provided that sufficient water is available during the growing season. Species composition may shift slightly, but lodgepole pine, subalpine fir, and quaking aspen will probably still dominate high mountain landscapes for the foreseeable future. Mountain pine beetle outbreaks may become increasingly chronic in a warmer climate, at least in the next few decades, and may continue to reduce the dominance of old lodgepole pine stands. If wildfire is not excluded from this forest type, composition and structure will generally be more resilient to climate change.

Whitebark pine mixed upper subalpine forests will respond more to whitebark pine mortality from white pine blister rust than to climate change, with significant changes in forest composition and structure. 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 blister rust, allowing subalpine fir to become dominant. Recent fires in the upper subalpine zone have reset succession to early-seral stages of shrub and herbaceous communities, but whitebark pine regeneration levels are low because of low population levels (Retzlaff et al. 2016), keeping burned areas in the shrub/herb stage for long periods. We expect that species dominance will continue to shift to subalpine fir, Engelmann spruce, and lodgepole pine. Many Northern Rockies wilderness areas have lands above the elevations at which this forest type occurs, so there are potential areas for range expansion. Some wildfire is needed to create conditions in which whitebark pine can become established and grow to maturity, but if fires are too severe, they will kill the pines needed to provide seeds for regeneration. Planting with rust-resistant trees will be needed to ensure the persistence of whitebark pine in this forest type.

5.4 Natural Resource Issues and Management

5.4.1 Landscape Heterogeneity

High landscape heterogeneity creates diverse biological structure and composition that are considered more resilient and resistant to disturbances (Cohn et al. 2015). 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, 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 fuel accumulations (Keane et al. 2002), although some areas with frequent disturbance are also homogeneous compared to pre-settlement forests. Many forests currently in late-seral conditions have low vigor and high fuel accumulations, making them susceptible to insects and disease and to the risk of severe wildfire.

Many lower-elevation forests in the Northern Rockies have less ability to buffer potential climate change effects because of high stand densities and dominance by shade-tolerant species. However, many higher-elevation forests, especially in the subalpine zone, have species composition and structure similar to what they were historically. Although recent wildfires, restoration activities (thinning, prescribed burning), and timber harvest have helped return some heterogeneity, most landscapes are outside their historical range and variability (HRV) in landscape structure (Box 5.1). This is a significant impediment to improving resilience to the stresses expected from climate change.

Landscape heterogeneity may increase if climate-mediated changes in disturbance regimes increase (Funk and Saunders 2014). Wildfire area burned and mountain pine beetle outbreaks have increased over the past 20 years, in some cases replacing late-seral forests with younger forests with more diverse structure. Continued increases in disturbances (Marlon et al. 2009; Bentz et al. 2010) might balance any loss of biodiversity with gains in landscape heterogeneity (Kappelle et al. 1999).

Large wildfires that will inevitably burn Northern Rockies landscapes may create large patches of homogeneous post-burn conditions (Flannigan et al. 2005, 2009), or may result in semi-permanent shrublands and grasslands in areas too dry for rapid conifer establishment (Fulé et al. 2004). However, post-fire heterogeneity varies considerably across large landscapes (Keane et al. 2008), and the extent of wildfires will almost certainly overwhelm any patterns created by land management.

Using HRV of landscape characteristics is a straightforward approach for making most forests more resilient to climate change (Keane et al. 2009; Keane 2013) (Box 5.1). Although HRV may not represent future conditions, it represents landscape conditions that have proven durable for centuries to millennia (Landres et al. 1999). HRV can be initially used as a reference for restoration (Keane et al. 2015a), then ecological models can be used to project future range of variability for a particular forest location (Keane 2012).

Box 5.1 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 identify a reference or benchmark to represent the conditions that describe fully functional ecosystems. Contemporary conditions can be evaluated against this reference to determine status, trend, and magnitude of change, and to design treatments that provide ecosystem services while returning declining ecosystems to a more sustainable condition. Reference conditions are assumed to represent the dynamic character of ecosystems and landscapes, varying across time and space.

The concept of **historical range and variability (HRV)** was introduced in the 1990s to describe past spatial and temporal variability of ecosystems, thus providing a foundation for planning and management. HRV has sometimes been equated with "target" conditions, although targets can be subjective and somewhat arbitrary, representing only one possible situation from a range of potential conditions.

HRV represents a 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, 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 transition to disequilibrium; (3) historical conditions can serve as a proxy for ecosystem health; (4) time and space domains that define HRV are sufficient to quantify observed variation; and (5) ecological characteristics assessed for ecosystems or landscapes match the management objective.

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, particularly altered disturbance processes, shifts in plant species distribution, and hydrologic dynamics. However, a critical evaluation of possible alternatives suggests that HRV is still a viable approach in the near term, because it has relatively lower uncertainty than methods that predict future ranges of variability.

An alternative to HRV is projecting **future range and variability (FRV)** for landscapes under changing climates, using empirical and mechanistic models. However, the range of projections for future climate from global climate models may be greater than the variability of climate over the past three centuries. This uncertainty increases when projected responses to climate change through technological advances, behavioral adaptations, and population growth are included. Moreover, variability of climate extremes, which will drive most ecosystem response to climate-mediated disturbance and plant dynamics, is difficult to project. Uncertainty will increase as climate projections are extrapolated to the finer scales and longer time periods needed to quantify FRV for landscapes.

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 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.

5.4.2 Timber Production

Approximately 22,000 km^2 of forested lands are currently managed for timber in the USFS Northern Region, reflecting a large decrease over the past 30 years. Species composition of timber harvests has fluctuated, with harvest following tree mortality caused by disturbance agents such as mountain pine beetle (lodgepole pine), spruce beetle (Engelmann spruce), white pine blister rust (western white pine), root disease (Douglas-fir, grand fir), and wildfire (several species). The current amount of land in each of the major species in lands suitable for timber production is ponderosa pine (6%), dry Douglas-fir (13%), lodgepole pine (27%), western larch (6%), subalpine fir and Engelmann spruce (12%), and mixed western white pine, grand fir, western hemlock, moist Douglas-fir and western redcedar (35%).

Recent harvests in mixed mesic forest 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. In eastern Montana and the Greater Yellowstone Area, harvesting has focused on beetle-killed lodgepole pine and ponderosa pine. Commercial and restoration thinning in ponderosa pine and dry Douglas-fir is also common.

A large amount of forested lands suitable for timber harvest is in mesic montane and subalpine forests, where productivity may increase in a warmer climate (Aston 2010), potentially leading to higher timber value. However, these forests could also become denser, less productive for timber, and more susceptible to insects and disease, especially in the absence of fire or active management (Joyce et al. 2008). In the future, harvesting timber from mature stands might be a race against losses from disturbance agents (Kirilenko and Sedjo 2007). Simply having more fire and smoke across the landscape in the future will limit access and opportunities for timber harvest.

It is essential that ecological principles be used to design harvest treatments to ensure that future forests are resilient to a warmer climate while continuing to provide a sustainable source of wood. Multiple resources and ecosystem services will

need to be considered as well, including residual fuel loadings, soil fertility, water quality, wildlife habitat, and fisheries habitat. Some practices that confer resilience for a particular resource may conflict with other objectives, requiring interdisciplinary planning to find optimal solutions.

5.4.3 Carbon Sequestration

Storage of carbon in (living and dead) biomass and in soils to reduce and defer carbon emissions into the atmosphere is an increasingly important consideration in forest management. Forests in the United States currently offset about 15% of annual U.S. carbon emissions. Size and persistence of forest carbon sinks depend on land management, vegetation composition and structure, and disturbance processes. Although long intervals between disturbance events allow carbon to accumulate for long periods of time, probability of disturbance increases with time (Loehman et al. 2014). Disturbance-prone forests will eventually emit stored carbon, regardless of management intervention, and net carbon balance is near zero over long time periods and large landscapes—unless changes in ecosystem structure and function occur.

This means that (1) disturbance-prone systems cannot be managed to increase stored carbon over historical amounts without limiting disturbance, and (2) shifts in vegetation abundance and distribution will alter spatial patterns of carbon storage. Therefore, expectations for carbon storage need to be developed in the context of climate change effects on vegetation, disturbance, and their interactions.

In general, expected increases in wildfire and other disturbances in the Northern Rockies will make it extremely difficult to maintain forest carbon storage at or above historical levels. Potential for future carbon storage can be assessed as follows:

- Is it reasonable to expect the system to accumulate carbon over historical levels, if the frequency, severity, and magnitude of disturbance events increases?
- What are appropriate temporal and spatial scales over which to measure carbon storage?
- Can potential future disturbance events be managed? Will it be possible to suppress or exclude wildfires, and at what economic or ecological costs?
- Can the effects of additional stressors (drought, invasive species, etc.) be mitigated to help maintain existing vegetation?
- Are future climatic conditions conducive to persistence of forests, or will conditions be inhospitable for current species?
- Do carbon accounting methods assess benefits of natural disturbance processes in carbon-equivalent units that can be weighed against carbon losses?

These are challenging questions that need to be informed by empirical data, ecosystem modeling, and future monitoring to incrementally improve our understanding of ecological drivers and responses to disturbance (Loehman et al. 2014).

Monitoring data can also be used to calibrate, validate, and provide input to models. Models can be used to simulate emergent environmental patterns, compare effects of potential treatments, and identify vulnerable landscapes or ecosystem components.

5.5 Adapting Forest Vegetation and Management to Climate Change

Adaptation to climate change can be defined as initiatives and measures to reduce the vulnerability of natural and human systems against actual or expected climate change effects (IPCC 2007). Most land managers have the tools, knowledge, and resources to begin to address climate change, which requires considering new issues, spatial scales, timing, and prioritization of efforts beyond a steady-state worldview (Swanston and Janowiak 2012).

Risk management is a key component of adaptation, prioritizing actions based on the magnitude and likelihood of climate change effects on resource vulnerability. *Adaptive management* provides a decision-making framework that maintains flexibility and incorporates new knowledge and experience over time. *No-regrets actions* focus on low-risk implementation of projects that could produce multiple benefits, regardless of climate change implications (e.g., removal of invasive species). *Triage* is sometimes needed in situations where vulnerability is high and immediate action is needed (e.g., a species facing extirpation). *Accomplishing multiple objectives* is often possible where an adaptation action also provides benefits for other resource objectives (e.g., riparian restoration, fuel treatments). *Addressing uncertainty* is a necessary component for adaptation, as for most resource planning, guiding the scope and timing of implementation.

A workshop process was used to identify adaptation options for all resources in the Northern Rockies, including vegetation. Teams of resource specialists and scientists reviewed climate change scenarios and a recent scientific assessment of the effects of climate change on vegetation. In response, they developed adaptation strategies (overarching, general) and adaptation tactics (specific, on the ground) within each strategy. These strategies and tactics, intended to guide both short- and long-term planning and management, were required to be feasible with respect to budget and level of effort, and to be acceptable within current policies.

5.5.1 Adaptation Strategies and Tactics

Of the many adaptation options identified for forest vegetation in the Northern Rockies (Halofsky et al. 2017), the major ones are summarized in Table 5.3. Many of the adaptation options are focused on protecting forests from and building

Table 5.3 Climate change adaptation options and restoration potential for tree species in the Northern Rockies

Species	Primary adaptive tactics	Restoration potential	Additional management recommendations
Ponderosa pine	Restore fire to historically fire-dominated stands; reduce fuel loadings to mitigate uncharacteristic fire severities; use HRV to guide restoration treatments.	Moderate to high. Reintroduce fire in fire-excluded stands as the first step; then identify where to plant in the future.	Reduce Douglas-fir in fire-excluded stands; remove competition with thinning and prescribed burns; monitor lower treeline in SW Montana and central Idaho.
Douglas-fir	Reduce competition and increase vigor; maintain low stem density; replace Douglas-fir with other species where root disease is a concern; emphasize ponderosa pine in low-elevation dry forests.	Moderate to high. Mitigate effects of fire exclusion era as the first step; reintroduce fire if possible (difficult in cool, dry environment).	Change species composition on sites where root disease and soil moisture deficits will increase; focus planting on higher elevation, mesic sites.
Western larch	Restore declining larch stands; prioritize treatments on north aspects and ash-cap soils; reduce competition; manage larch intensively on xeric sites; reduce stand density.	Moderate to high in Western Rockies. Moderate in Central Rockies.	Remove shade-tolerant species using group selection and thinning; prioritize planting options on north slopes; use genetic stock with best adaptive traits for drought and moisture stress.
Western white pine	Promote propagation of genotypes with resistance to white pine blister rust	Moderate in Western Rockies. Low to moderate in Central Rockies.	Increase planting of genotypes that have resistance to blister rust; thin dense stands to increase vigor of young pines
Grand fir	Ensure landscape heterogeneity; ensure age-class structure is near HRV.	High in Western and Central Rockies.	Invest in restoration only if the species is declining locally.
Western redcedar	Ensure landscape heterogeneity; maintain age-class diversity.	High in Western and Central Rockies.	Invest in restoration only if the species is declining locally.
Western hemlock	Ensure landscape heterogeneity; maintain age-class diversity.	High in Western and Central Rockies.	Invest in restoration only if the species is declining locally.
Lodgepole pine	Manage for mixed age classes and successional stages that approximate HRV.	Moderate to high.	Allow wildfires to burn where possible.

(continued)

Table 5.3 (continued)

Species	Primary adaptive tactics	Restoration potential	Additional management recommendations
Limber pine	Promote white pine blister rust resistance while preserving genetic diversity; monitor mortality rates and distribution; determine effects of fire exclusion.	Low to moderate. Most actions should increase rust resistance in native populations.	Implement rust-resistance programs; identify superior genotypes; collect cones and determine rust resistance; map limber pine populations to identify stands established before and after fire exclusion.
Subalpine fir	Use wildfire suppression to reduce species loss locally.	High.	Invest in restoration only if the species is declining locally.
Engelmann spruce	Use wildfire suppression to reduce species loss locally; plant selectively where populations are declining.	Mostly high, but moderate in low-elevation wet sites.	Invest in restoration only if the species is declining locally.
Mountain hemlock	Use wildfire suppression to reduce species loss locally.	Moderate to high in Western and Central Rockies.	Monitor to ensure the species is not locally extirpated.
Alpine larch	Preserve genetic diversity by collecting and storing seed.	Low to moderate.	Monitor changes in alpine larch populations.
Whitebark pine	Follow strategies in Keane et al. (2012); promote propagation of genotypes with resistance to white pine blister rust; conserve genetic diversity; prioritize treatments at high elevation.	Low to moderate because of stress imposed by blister rust.	Protect rust-resistant, high-vigor trees; implement prescribed fire and mechanical cuttings to reduce competition; plant and direct-seed rust-resistant seedlings on burns and treated areas; use hardy, drought-tolerant seedlings.
Quaking aspen	Restore quasi-historical fire regimes; prioritize areas where aspen already exists, even if at lower than historical levels.	Moderate.	Plant aspen where now absent but once existed; ensure diversity of age classes and seral stages across landscapes.
Cottonwood	Encourage high variability in streamflows to increase seedling establishment; reduce competition.	Moderate to high.	Prioritize the most mesic sites first; allow fire to burn in areas that are not too dense; remove competing conifers.
Green ash	Reduce grazing; use fire suppression and planting to promote ash populations in areas with low populations.	High in Eastern Rockies and Grassland.	Plant ash in recently burned areas where it recently existed.

resilience to severe disturbance, primarily wildfire. For example, promoting disturbance-resilient forest structure and species is a key adaptation strategy that guides management of vegetation and other resource areas in the Northern Rockies. Thinning and prescribed fire can be used to reduce forest density and promote disturbance-resilient species. Disturbance-resilient species can also be planted. Managers recognize 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 dry sites, and lodgepole pine on sites that are difficult to regenerate.

Preparing for disturbance will also be important in a changing climate. Tree regeneration after severe fire may be more limited in the future if drought frequency increases. 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.

Promoting species diversity, genetic diversity, and landscape diversity is also a critical adaptation strategy. Increasing diversity is a "hedge your bets" strategy that reduces risk of major forest loss. Areas with low species and genetic diversity will probably be more susceptible to stressors associated with climate change, so promoting species and genetic diversity, through plantings and in thinning treatments, will increase forest resilience to changing climate. Promoting landscape heterogeneity, in terms of species and structure, will also increase resilience to wildfire, insects, and disease.

Managers identified several ways to increase knowledge and manage in the face of uncertainty. Implementation of an adaptive management framework can help address 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 how 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 needed in implementation of the adaptive management framework to ensure that management approaches do not conflict (e.g., which effects will a particular thinning treatment have on wildlife?).

Managers also identified adaptation strategies and tactics to maintain particular species or community types of concern. For example, climate change will probably lead to increased whitebark pine mortality through white pine blister rust, mountain pine beetle activity, and wildfire. To promote resilient whitebark pine communities, managers may want to focus restoration efforts on sites less likely to be affected by climate change (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.

Because stressors associated with climate change will be spatially pervasive, it will be important for agencies to 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 will be more frequently impacted by fire and flood events under changing climate.

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